

**Evaluating eelgrass (*Zostera marina*) as a juvenile habitat for rockfishes:
contributions of site characteristics and larval supply to juvenile
abundance.**

by

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A THESIS SUBMITTED IN PARTIAL FULFILLMENT
OF THE REQUIREMENTS FOR THE DEGREE OF
MASTER OF SCIENCE
in
THE FACULTY OF GRADUATE STUDIES
(Botany)

University of British Columbia

February 2008

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ABSTRACT

Seagrass beds are highly valuable ecosystems with a potential nursery function for rockfishes in British Columbia. Understanding how the nursery value of seagrass habitats differs for rockfish species, and what factors create this variability in habitat value, is important for effective conservation planning. In this study, the use of *Zostera marina* beds in Barkley Sound by juvenile rockfishes (*Sebastes* spp.) was investigated. Rockfish have been observed in *Z. marina* beds by many researchers, however, their abundance and distribution between beds has been shown to be variable. The abundance of juvenile rockfishes in five beds was assessed in 2005 and 2006 using visual surveys. Environmental and biological variables that could potentially influence this abundance were measured concurrently. These included supply of settling fish (hereafter “settlers”), biotic and abiotic habitat characteristics.

The eelgrass beds that I studied were found to differ significantly in their habitat characteristics, the abundance of settlers arriving to them, as well as the abundance of juvenile recruits in the beds. *Sebastes melanops* and *S. caurinus* were found at the sites between June and September. There was no significant relationship between the abundance of settlers arriving at a bed, and the abundance of juveniles using it. However, many of the eleven habitat characteristics investigated were significantly correlated with the abundance of either *S. melanops*, or *S. caurinus*. It is likely that these relationships arose from multiple causes which acted both pre- and post-settlement. Habitat features that were identified as correlates with high juvenile abundance value included lower epiphyte biomass, nearness to kelp habitats, lower water temperature, lower shoot density, and higher leaf width.

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ACKNOWLEDGEMENTS

I have been supported in this endeavour from the moment it became an idea in the back of my head by many fantastic people whom I am grateful for the opportunity to acknowledge here. I am extraordinarily grateful to my supervisors Robert DeWreede and Tomas Tomascik for their support, patience, and financial assistance. I am also very grateful for the faith that they had in me which allowed them to send me unsupervised into the wilds of Barkley Sound in search of eelgrass and tiny rockfishes. I would also like to thank the members of my supervising committee for their contributions to the planning and writing of this thesis.

I owe a great deal of thanks to the people and volunteers at the Bamfield Marine Sciences Center who were so supportive of my project, and gave up many hours of their time to dive, boat tend and tromp around mucky eelgrass beds with me. I could never have completed the many hours of field work required for this project without their help.

To my cheerful and positive field assistant, Stephanie Avery-Gomm; thank-you for all of your help with the fieldwork, and your company during early mornings and long dives. To Ramona DeGraff and Cynthia Durance and so many others who helped me develop my ideas of what would make a meaningful thesis, I am very grateful for your thoughts and encouragement. I am also grateful for my previous coworkers in whose company I gained a love of fieldwork and many of the skills necessary to complete this project.

To my family and friends who have provided undying encouragement, support and patience from the moment I started applying for graduate school, I thank you. Your technical support, patience with my preoccupation, and even feigned interest in eelgrass has meant a lot to me for these past few years, and my experience in grad school would not have been the same without it.

Finally, I am grateful for the financial assistance of Parks Canada (Western Northern Service Center Research Grant), NSERC (PGS M grant to S. Jeffery, and grant # 9872-04 to R. DeWreede), and finally to the Bamfield Marine Science Center for a Sigurd-Tveit scholarship.

To my family who have always supported me in my endeavours, and to Caesar who was a constant companion through the completion of this work.

Chapter 1: Introduction

1.1 SEAGRASSES

1.1.1 Seagrass ecology

The term seagrass is used to describe a polyphyletic group of plants which grow submerged or partially submerged in marine or estuarine waters. More specifically, seagrasses are clonal marine angiosperms that have the unique ability to complete sexual reproduction while immersed in a saline environment (Scagel 1971). It is thought that the first seagrasses evolved from flowering land plants approximately 100 million years ago during the Cretaceous period (McCoy and Heck 1976; Hemminga and Duarte 2000). Presently, seagrasses are ubiquitous in both temperate and tropical areas, forming extensive meadows in shallow subtidal and intertidal areas of every continent except Antarctica (Green and Short 2003). They often form monospecific stands, but can also grow as multi-species beds, especially in tropical locations.

There are more than 60 species of seagrasses in the world (Green and Short 2003). Most of these occur in the tropical regions of the Indo-Pacific where countries like Indonesia are home to up to twelve species (Tomascik *et al.* 1997). In the temperate waters off the coast of British Columbia there are only five species of seagrasses belonging to the genera *Zostera* and *Phyllospadix*, in the family Potamogetonaceae (Gabrielson *et al.* 2000).

Being angiosperms, seagrasses can reproduce sexually by means of flowers. However, they also reproduce vegetatively by horizontal rhizome extension; in fact, many populations persist exclusively through such asexual reproduction (Rasheed 2004). The balance between sexual and asexual reproduction in a meadow varies between locations and species (Alexandre *et al.* 2006). For the temperate *Zostera marina*, this balance seems to be determined by levels of

physical stress experienced by the plants. For example, in the southern extent of the species range where summer temperatures are high, entire meadows die on an annual basis and re-grow from seeds every year, whereas in areas lacking thermal stress flowering seldom occurs (Phillips *et al.* 1983).

Architecturally, seagrasses are modular plants composed of repeated units called 'ramets'. Each ramet contains a section of rhizome, a root system, and leaves enclosed in a sheath (Hemminga and Duarte 2000)(Figure 1.1). Together, rhizomes and roots systems are responsible for anchoring, nutrient uptake and carbohydrate storage (Bertness *et al.* 2001; Harrison 1990). Seagrass leaves are the photosynthetic component of the plant, but are also capable of nutrient and water uptake (Kuo and den Hartog 2006). Morphologically, leaves fall into two basic types; strap-like, such as *Zostera*, and leaf-like, such as *Halophila* (Kuo and den Hartog 2006). Although the repertoire of body forms for seagrasses is low, individual plants exhibit much plasticity in response to abiotic conditions which allows them to persist in a variety of conditions (Hemminga and Duarte 2000).

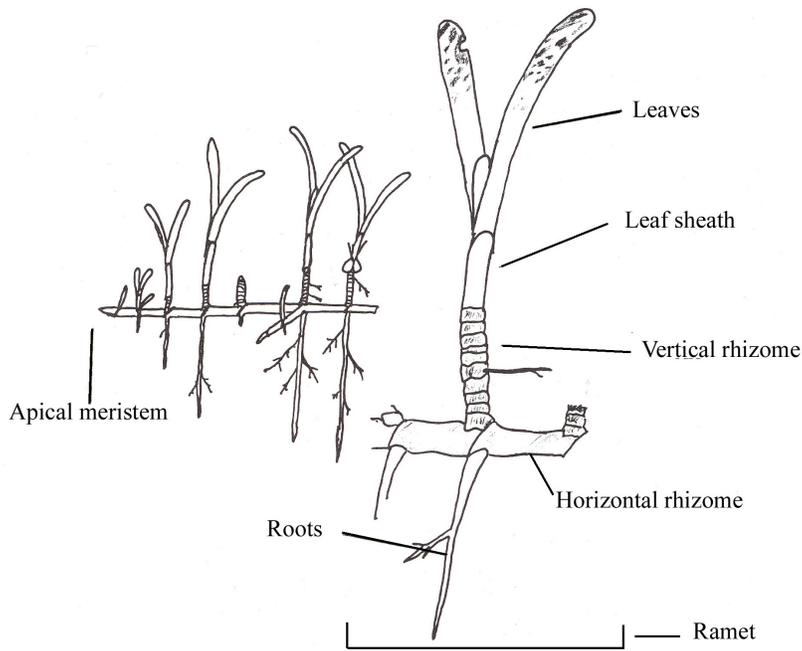


Figure 1.1. Schematic of a seagrass plant with strap-like leaves showing components of an individual ramet, and the modular structure of the plant. (Adapted from Hemminga and Duarte 2000- pg. 29).

1.1.2 Importance of seagrass ecosystems

Seagrasses are recognised as a valuable habitat worldwide because of the many ecosystem services they provide. Firstly, their above and below-ground structures alter the surrounding marine environment in a number of ecologically beneficial ways. For instance, the extensive rhizome network of seagrasses binds the soft sediments in which they anchor. This helps to prevent erosion (Duarte 2002), which is especially important in the shallow coastal areas where they are found. Also, the three-dimensional structure of seagrass leaves creates friction as currents and waves pass over them, baffling water motion and further reducing erosion (Peterson *et al.* 2004). The slowing of water due to friction within seagrass beds is also beneficial because it increases deposition of sediments and other particles which increases the clarity of the water that passes through it (Green and Short 2003; Hemminga *et al.* 1991).

Secondly, seagrass beds are highly productive ecosystems. They have an average biomass turnover rate of $2.6 \pm 0.3\%$ per day, and their yearly production is estimated at 1000 gC/m²/year; rivalling that of even the most productive vegetated habitats on earth (Duarte and Chiscano 1999). For example, the renowned productivity of *Macrocystis pyrifera* beds in California ranges between 350-1500 gC/m²/year (Dayton 1985). The fate of the energy sequestered by seagrasses varies between species and geographic location. In warmer climates, a portion of the seagrass material produced becomes food for large herbivores such as dugongs, manatees, and sea turtles who rely almost exclusively on seagrasses as food (Thayer *et al.* 1984). However, in many areas, the majority of seagrass material is not consumed because it is either not very palatable (Thayer *et al.* 1984), or is below-ground, and therefore unavailable to most consumers. Thus, a large percentage of the energy sequestered by seagrasses is either recycled internally within a bed (Harrison 1989), or exported to adjacent ecosystems. In fact, it is estimated that an average of 24.3% of energy produced in a seagrass bed is transported to other ecosystems (Duarte & Cebrian 1996). The effect of this transferred energy on other ecosystems is largely unstudied, but the fact that seagrass material has been found in habitats as far removed as the deep sea suggests they may be far-reaching, and potentially important to the overall functioning of the world's oceans.

Finally, the habitat that seagrasses create is important for many faunal species, and has long been considered a nursery for many due to the large numbers of juvenile fishes that are often found within it (McNeill *et al.* 1992a; Nagelkerken *et al.* 2000d; Beck *et al.* 2001; Beck *et al.* 2001; Nakamura and Sano 2004a; Jenkins *et al.* 1997c). Many commercially and ecologically important species with pelagic larval stages seem to rely on seagrass habitat to bridge their early planktonic existence with later adult habitats (Beck *et al.* 2001; Smith and Sinerchia 2004). To date, seagrasses have been found to function as nurseries for many reef fish

in the Caribbean (Nagelkerken *et al.* 2002), the blue crab, *Callinectes sapidus* in Virginia, USA (Perkins-Visser *et al.* 1996), and black rockfish (*Sebastes inermis*) in Japan (Guido *et al.* 2004b). The benefits reportedly provided by seagrasses that make them such important juvenile habitats are: provision of protection from predation (Nakamura and Sano 2004a; Rooker *et al.* 1998b), enhanced growth due to an abundance of food (Nagelkerken *et al.* 2000c), and greater interception of larvae due to friction within the beds (Parrish 1989).

1.1.3 Seagrass nursery hypothesis

The role of seagrasses as a nursery for fish and invertebrates has been widely cited and accepted in the literature in past decades. However, a critical evaluation of them in such a capacity has not been conducted for the majority of seagrass species or their inhabitants despite the fact that such beds are referred to as nurseries throughout the literature. In order to successfully study a habitat as a nursery, scientists need a working definition of the word to act as a framework for their research objectives. Until recently, the term ‘nursery’ has lacked such a definition; a shortcoming which has made testing the nursery function of habitats like seagrasses problematic (Beck *et al.* 2001), and results difficult to compare across studies. In 2001, Beck *et al.* wrote an influential paper that addressed this ambiguity. They defined the term nursery, and outlined a hypothesis and predictions for testing the nursery function of habitats. Since its publication, it has been cited in the majority of papers addressing this issue (Heck *et al.* 2003; Kraus and Secor 2005; Uhrin and Holmquist 2003), and has influenced the experimental design of many. In their paper, Beck *et al.* state that “a habitat is a nursery for juveniles of a particular species if its contribution *per unit area* to the production of individuals that recruit to adult populations is greater, on average, than production from other habitats in which juveniles occur”. Testing this hypothesis involves evaluating all habitats in which juveniles of a

particular species occur, and demonstrating movement of juveniles from the nursery to adult habitats. This requires knowledge of where juveniles of a species are found, and also limits nursery tests to species with disjunction in habitat use between adults and juveniles.

Until the 2001 paper was published, many studies simply assessed the abundance of juvenile animals in seagrass. The high abundance of juveniles in these beds prompted the notion that they act as nurseries, and this notion became commonly accepted in the scientific community (Nagelkerken *et al.* 2000b; Heck *et al.* 1997). Studies such as these assumed that the higher numbers of juveniles found in seagrass habitat would recruit into adult habitats in proportionately high numbers (Nagelkerken *et al.* 2000d; Costa *et al.* 1994). This assumption was rarely tested, however, and would not hold true if the habitat was not well placed for movement to adult habitats, or had a higher amount of predation. It has been shown, for example, that high numbers of juveniles actually attract opportunistic predators in some cases (Hobson *et al.* 2001). Beck *et al.* (2001) felt that juvenile density was only one of four factors that could contribute to a high nursery value; the other three being growth, survival and movement to adult habitats. A rigorous test of nursery function should test all four of these factors, in all habitats in which juveniles of a given species are known to occur (Beck *et al.* 2001).

1.1.4 Threats to seagrasses

Approximately 50% of the world's human population lives within 100km of a coastline (Coen *et al.* 1997), and this percentage is expected to rise significantly in the near future. As coastal populations grow, human activities will increasingly disturb nearshore habitats such as seagrass meadows (Heck *et al.* 2003; Uhrin and Holmquist 2003). Disturbances caused by these activities include: mechanical damage from dredging, boating activities and the construction of

ports and marinas (Harrison 1990); decreased light levels from higher turbidity and shading from docks (Short and Wyllie-Echeverria 1996; Burdick and Short 1999); eutrophication from agricultural or urban runoff and sewage outfalls (Hauxwell *et al.* 2003); increased water temperatures from the discharge of heated waste water (Homziak *et al.* 1982); and disrupted faunal communities initiated by overfishing (Heck *et al.* 2000). Habitat fragmentation (Uhrin and Holmquist 2003), and increased epiphyte loads (Hauxwell *et al.* 2003; Piazzini *et al.* 2004) often result from these disturbances (Green and Short 2003) and jeopardize the continued survival of seagrass habitat.

Seagrass loss caused by the previously mentioned factors has been significant already. The amount of documented seagrass lost worldwide in the 1990's alone was over 1900 km², and actual amounts were certainly much higher (Short and Wyllie-Echeverria 1996). Long-term declines have been documented in Denmark (Frederiksen *et al.* 2004b), as well as in Puget Sound (Gaeckle *et al.* 2007), and have been observed for most seagrass species around the world (Green and Short 2003). The increasing threats to seagrasses, coupled with their value as nurseries, areas of high primary production, important links between ecosystems, etc., requires an urgent understanding of their role in the oceans to improve conservation efforts.

1.1.5 Ecology of eelgrass (*Zostera marina*)

Eelgrass, *Zostera marina* Linnaeus, is the most common seagrass species on the west coast of British Columbia. It forms extensive beds in shallow, wave protected nearshore areas with mud or sand substrates. The global range of this species is circumboreal, and on the west coast of North America it is found from Alaska to Baja California (Green and Short 2003). Because of its wide distribution, eelgrass and its associated fauna has been studied extensively

in the scientific literature; however, most of these studies have focused on eelgrass habitats on the east coast of North America. Similar ecological studies on the west coast are rare.

A noticeable deficit in our knowledge resulting from this bias is our lack of information about the role eelgrass plays as a juvenile habitat for commercially important species in British Columbia. Juveniles of the following species are commonly documented in eelgrass beds: prawns (*Pandalus platyceros*), Dungeness crabs (*Cancer magister*), salmon (*Oncorhynchus* sp.), cod (*Gadus* sp.), herring (*Clupea harengus pallasii*) and rockfishes (*Sebastes* sp.) (Johnson and Thedinga 2005; Jensen 1995). The importance of this habitat for these species has not been evaluated, however, nor is it known how variability in eelgrass habitat affects its value for the species using it. Given the extent of eelgrass habitat, and the variety and abundance of juveniles that use it, it has been assumed that eelgrass is a highly important nursery habitat, despite the fact that a rigorous test of its nursery function (*sensu* Beck *et al.* 2001) for the above-mentioned species has not been performed to date.

The decline of eelgrass in British Columbia as a result of nearshore development is thought to be significant (Copping *et al.* 2005), especially within the Strait of Georgia. However, estimating this decline is hampered by the lack of baseline data on where eelgrass existed historically, or is located presently. Significant progress in mapping eelgrass habitat province-wide has been made recently¹. If the loss of eelgrass in B.C. is equivalent to the estimated rates of loss worldwide, 2-5%/year (Duarte 2007), it is possible that much of the province's eelgrass habitat would be lost before researchers are able to scientifically evaluate its importance to our fish species, or learn what features of the beds positively influence its habitat value.

¹ The results of these mapping efforts can be seen on the Community Mapping Network. <http://www.shim.bc.ca/atlasses/eelgrass/main.htm>

1.2 ROCKFISHES

1.2.1 Rockfish ecology

The group of fishes referred to as ‘rockfish’ are a diverse group belonging to the genus *Sebastes* and the family Scorpaenidae. Members of this genus are characterized by a dorsal fin with both spiny and soft rays, toxic spines, a suborbital stay (a type of cheek bone), high fecundity, and viviparity (Love *et al.* 2002). There are over 100 species of rockfishes in the world, most of which occur in the north Pacific (Love *et al.* 2002). The area with the highest diversity of rockfishes is the coast of California where up to 60 different species can be found. Rockfishes live in a variety of habitats ranging from midwater to benthic environments; intertidal areas to depths of more than 1000m; and rocky reefs to vegetated areas (Love *et al.* 2002). Many species in this genus are brightly coloured and prized by underwater photographers and live food fishermen alike. As a group, rockfishes are very long-lived with delayed sexual maturity; larger species such as rougheye rockfish (*Sebastes aleutianus*) have been aged at more than 200 years old, and only 50% are sexually mature at 20 years old (Love *et al.* 2002).

1.2.2 Reproduction

Rockfish reproduction involves internal fertilization, internal development of larvae, and extrusion of a large number of developed larvae into the plankton (Reilly *et al.* 1992). This mode of reproduction is termed viviparity and is characterized by the transfer of energy from mother to embryos during development, and the release of live young. During the mating season female rockfish can retain sperm from multiple donors and delay fertilization of eggs until environmental conditions favour parturition (Wyllie-Echeverria 1987). In British

Columbia fertilization occurs between January and June, although the species considered in this study (*S. melanops*, *S. melanops*, *S. flavidus*, and *S. paucispinus*) are not reported to release larvae later than April (Wyllie-Echeverria 1987). Female rockfish can release over two million larvae, with larger, older females releasing both the larger amount, and the most viable (Berkeley *et al.* 2004). Rockfish are so fecund, in fact, that their larvae consistently rank as the fourth most abundant group of fish larvae in the plankton (Moser and Boehlert 1991). The timing of larval release usually coincides with maximal upwelling (Carr 1991), and highest ocean productivity, which ensures maximal larval survival in the plankton (Yoklavich *et al.* 1994). Newly released larvae are between 3 and 7 mm long, and grow at rates up to 0.17 mm/day (Love *et al.* 2002).

Development in the planktonic environment lasts several months for most species (Anderson 1983), and comprises larval and early juvenile stages of development (Love *et al.* 2002). The two stages of planktonic development are termed the pelagic larval stage, and the pelagic juvenile stage. During the first stage, larvae undergo flexion (upward bending of the notochord for caudal fin formation), and develop head spines and fin rays. By the time fish are pelagic juveniles, their meristics have reached adult counts. The pelagic juvenile stage ends once the fish begin to associate with the benthos (Love and Haldorson 1991).

1.2.3 Juvenile settlement and recruitment

Although the terminology used in the literature is confusing, the process of leaving the plankton and associating with a benthic habitat is generally termed ‘settlement’, while recruitment is the survival of post-settlement fish for an indeterminate amount of time (*sensu* Jenkins *et al.* 1998). The timing of settlement varies between rockfish species but usually happens between May and July, when the fish are 3-9 cm long (Love *et al.* 2002). The depth

and habitat used by post-settlement fishes are varied, and differ between species, but juveniles generally settle into shallower waters than adult conspecifics, resulting in a high degree of habitat separation between life stages (Love and Halderson 1991). This is thought to be the result of ontogenetic changes in resource needs, temperature preferences and foraging abilities (Halderson and Richards 1987; Love *et al.* 2002). A consequence of this habitat separation is the possibility of evaluating nursery habitats for juveniles as defined by Beck *et al.* (2001).

1.2.4 Threats to rockfishes

Rockfish populations in B.C. have declined significantly over the past few decades, with population levels at less than 10% of historic values for some species (Love *et al.* 2002), and one species in particular (*S. paucispinus*) is currently listed as threatened by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). It has been suggested that the causes for these declines are largely overfishing and poor recruitment due to adverse oceanographic conditions (Moser *et al.* 2000; Love *et al.* 1998).

Approximately 28 species of rockfish are targeted commercially in B.C., primarily by the hook and line fishery (Anon 2006). The commercial catch of rockfishes by this method in B.C. was 1780 tonnes in 1998, while recreational fishers annually remove 84,000 individuals from the Strait of Georgia alone (DFO 2000). Large numbers of rockfishes are also caught as bycatch in other fisheries, and so actual numbers of rockfish being removed from British Columbian waters are largely unknown. In a DFO research document, Yamanaka and Lacko (2001b) stated that

“The available data for stock assessment is insufficient to recommend species specific catch quotas for the five management areas on the Pacific coast of British Columbia. Given the biology of inshore rockfish, together with the inability to assess total catch and the evidence of excessive fishing mortality, it is recommended that a portion of the rockfish population in each management area be completely protected from

harvest through spatial management measures such as harvest refugia (areas closed to fishing) as a precautionary measure to promote conservation”.

This statement, along with declining rockfish populations, led to the development of the Rockfish/Lingcod Conservation Strategy in 2002 (DFO 2002). Under this strategy, 89 areas in British Columbia have been closed to hook and line fishing to protect inshore rockfish populations².

Changing oceanic conditions may be reducing the survival of rockfish larvae and resulting in poor recruitment to the fishery. Many studies have shown that warmer waters, such as those experienced during El Nino events, adversely affect larval survival and subsequent year-class strength of rockfishes (Lenarz *et al.* 1995; Ralston and Howard 1995; Moser *et al.* 2000). These warmer temperatures may adversely affect larval survival in a number of ways. Increased water temperature is associated with reduced upwelling and lower productivity which can be related to higher starvation rates in larvae (Ralston and Howard 1995). Similarly, increased onshore advection caused by reduced upwelling results in changes to the dispersal patterns of pelagic larvae and may increase mortality by bringing larvae into areas that they are not adapted to (Lenarz *et al.* 1995). Coupled with high fishing pressures, poor environmental conditions for larvae and subsequent reductions in recruitment may drive rockfish populations to unrecoverable levels if measures are not taken to adequately protect them.

1.3 THESIS OUTLINE

1.3.1 Thesis objectives

The purpose of this investigation was to evaluate eelgrass as a juvenile habitat for rockfish (*Sebastes* spp.) in the context of habitat management and marine reserve design. The

² http://www.pac.dfo-mpo.gc.ca/recfish/Restricted_Areas/RCA/default_e.htm

information gained from this study will broaden our understanding of juvenile rockfish use of eelgrass, and help us to recognize important habitat features which may aid fisheries and habitat managers with the selection of valuable reserve areas.

This study is timely given the recent decline in rockfish populations and the increased conservation effort being afforded to them, since the preservation of habitats for all life stages of rockfish is of utmost importance (Love *et al.* 2002), and the most effective preservation will require information on which habitats are used by juveniles and what habitat characteristics are related to increased habitat use.

The two questions addressed in this thesis were:

1. Are all eelgrass beds used to the same extent by juvenile rockfishes?
2. What influences the abundance of rockfish between eelgrass beds?

1.3.2 Hypotheses

It was hypothesized that habitat use by juvenile rockfishes would vary between sites. Differences in habitat use may involve the timing and length of residency in eelgrass, the abundance of juveniles within a bed, or their species composition. It was hypothesized that this use, most notably the abundance of juveniles, would be related to a combination of habitat variables, and to some extent by the availability of pelagic juveniles to the site.

Chapter 2: Environmental and biological characteristics of five eelgrass beds

2.1 INTRODUCTION

Despite superficial similarities, monospecific seagrass beds often differ from one another in a number of important ways. Such differences include, but are not limited to, the density and structure of the plants that make up the bed, the composition and abundance of epiphytes living on the blades, and the environmental conditions within and surrounding the bed. These interbed differences have been reported for many different seagrass species, in various parts of the world. For example, species richness in mixed-species seagrass beds can vary from 1-7 species of seagrasses in tropical locations (Hemminga and Duarte 2000). In terms of abiotic conditions, Fonseca & Bell (1998) recorded maximal tidal currents through mixed *Zostera marina* and *Halodule wrightii* beds in N. Carolina ranging from 5-37 cm/s, and Robinson & Martel (2007) reported nitrate levels ranging from undetectable to 1.40 mg/L in beds within the relatively small and geographically similar area of Barkley Sound on the west coast of Vancouver Island, British Columbia.

Bed characteristics are not homogenous within a given bed either. Many variables differ between depths, and between edge and interior regions of a bed. For example, shoot density has been shown to decrease with increasing depth (Collier *et al.* 2007), and water motion and nutrients are also known to decline from the edge to interior of a bed due to friction with seagrass leaves (Bartleson 2004).

Many of these differences have been correlated with differences in associated faunal communities. Therefore, the evaluation of differences in biotic and abiotic factors within and among seagrass beds is important for understanding the overall habitat value of seagrass beds,

and how they function as nursery habitats. Examples of these correlations include: interbed differences in relative habitat use by King George whiting (*Sillaginodes punctata*) were negatively correlated with canopy height in Australian seagrass beds (Jenkins and Hamer 2001); epiphyte load was shown to influence differences in decapod crustacean densities between beds in the English channel (Jackson *et al.* 1991); and the species composition of seagrasses affected the abundance of juvenile fishes in a Texas lagoon where juveniles were more abundant in beds of *Halodule wrightii* than the other 3 seagrasses evaluated (Tolan *et al.* 1997). Similarly, the abundance and composition of fish species differed between deep and shallow zones of *Posidonia* beds in Australia (Bell *et al.* 1992).

Therefore, to achieve the overall objective of relating rockfish abundance in eelgrass (*Z. marina*) beds to habitat characteristics, I chose to measure the following variables as potential determinants of rockfish species composition and abundance: temperature, salinity, water motion, bed dimensions, proximity to kelp habitats, eelgrass characteristics, eelgrass growth, and epiphyte biomass (Table 2.1). These variables were used to characterize the study sites in this chapter, and were used for analysis with rockfish data in chapter five. The key questions addressed in this chapter are:

1. Do the five study sites differ in terms of biological and environmental variables?
2. Do these variables differ between depths within a site?
3. What combination of variables best distinguishes the sites from one other?

2.2 METHODS

2.2.1 Study sites

The eelgrass beds evaluated in this study are located in Barkley Sound, on the west coast of Vancouver Island, British Columbia (Figure 2.1). The first site, Dodger (Figure 2.1a), is located in Dodger channel adjacent to Haines Island (48° 50.0N, 125° 11.7W). It is a large bed that borders both sides of the channel almost connecting in the center. Only the southernmost side of the bed was evaluated; it is 22,936 m² in area and spans a tidal range of +0.5 m to -5.2 m relative to chart datum (Fisheries and Oceans Canada 2006). The eelgrass at this site is distributed continuously, but is much denser in shallow than deeper areas. The bed is affected by strong tidal currents, but its orientation is such that it receives little to no wave exposure (personal observation). The sediment at this site is very fine silt. The bed is bordered on the eastern side by a stand of kelp, *Macrocystis integrifolia*.

The second site, Roque, is located in Roquefeuil Bay on the south eastern side of Barkley Sound (48°51.7N, 125°6.3W) (Figure 2.1b). The bay opens to the southwest, which is the direction of prevailing wave action. The eelgrass at this site occurs at depths ranging from +0.5 m to -2.7 m relative to chart datum. This bed covers 3,220 m² with continuously distributed eelgrass. No obvious ecotype or shoot density changes were observed from the shallow to deep zones of the bed. Rocky reefs containing stands of *M. integrifolia* border the northern and southern edges of the bed, and a rocky island is located approximately 13 meters from the deep edge of the bed. *Sargassum* and *Phyllospadix* are dominant in the higher intertidal. The sediment at this site is coarse sand.

The third site, Fleming, is located in a small bay on the west side of Fleming Island (48°52.9N, 125°8.7W)(Figure 2.1c). The bay opens to the northwest and is protected from some of the wave action in Imperial Eagle channel by this aspect. At this site, eelgrass occurs

only as a fringing band along a narrow depth range close to shore. The shallow edge of the bed is at a similar depth to Dodger and Roque (+0.5 m), but the deepest edge of the bed is much shallower (only 0.6 m below chart datum). This bed is the smallest of all the study sites at 1,828 m², however, it has the highest shoot density. The shoreline of the bay is lined with boulders beyond the edges of the bed, and a small rocky, kelp-lined island is located approximately 80 m from the subtidal edge of the eelgrass bed.

The fourth and fifth sites are situated on either side of Robber's passage, a channel that separates Fleming and Tzartus islands. Robbers South (S) is located in a slight bay on the southern side of Robber's passage, at the northern end of Fleming island (48° 53.58N, 125° 07.12W)(Figure 2.1d). Robbers North (N) is on the northern side of the passage, situated on Tzartus island (48° 53.8N, 125° 07.01W)(Figure 2.1e). Both of these beds have a considerable intertidal portion. The eelgrass at Robbers S is continuously distributed and ranges from the high intertidal to 3.6 m below chart datum. The main portion of the bed is flat with continuous eelgrass. A fringing bed extends from the main bed along the shoreline to the west into an adjacent bay. The aerial coverage of the flat portion of the bed, which was the main study area, is 7,371 m². The bay where the bed is situated opens to the north and is sheltered from almost all wave action and currents. A small yacht club with one long dock is located on the east side of the bay, just beyond the subtidal edge of the eelgrass bed. During July and August boats were regularly moored on the dock. During rain events, a small creek empties into the bay through the eelgrass bed.

The eelgrass at Robbers N also extends into the high intertidal, but at this site the subtidal edge of the bed is very shallow (only 1.0 m below chart datum). The substrate consists of fine white sand. The bed is oriented towards the west into Imperial Eagle channel; the direction of most incoming wave action. Robbers N is partly protected from this wave action by

a small barrier island and a peninsula from Tzartus Island. A small, ephemeral fresh water stream divides the intertidal bed area and empties into the subtidal portion. During summer months (when this bed was studied) the stream usually contained low amounts of water.

In July and August 2005 all bed characteristics were measured at Robbers S and N. In 2006, Dodger, Roque and Fleming were evaluated in detail from May - September, and some variables were re-measured at the 2005 sites. The following sections provide a description of how each variable was measured and analyzed. The eelgrass beds were stratified into intertidal (high, mid and low) and subtidal (shallow and deep) zones. The high intertidal zone was defined as the area higher than 0.6 m above chart datum, the mid intertidal was between 0.3 and 0.6 m, and the low intertidal between 0.3 and 0 m. Subtidal zones were defined as follows: shallow = 0 to 1 m below chart datum, and deep = greater than 1 m below chart datum. Measurements were made in each of the 5 zones for each variable.

2.2.2 Sampling methods

A summary of the variables described below, their units, method of measurement, and scale of replication are presented in Table 2.1.

2.2.2.1 Temperature

A plastic-encased alcohol thermometer calibrated in degrees was taken on every dive and the temperature of the water was recorded in °C during the dive.

2.2.2.2 Salinity

A labelled 20 mL glass vial was taken on almost every dive and filled at depth within the eelgrass bed. These containers were closed tightly and stored until the end of the field season when their salinity was measured in parts per thousand (ppt) using a salinity refractometer that was calibrated with fresh water after every 10-15 samples.

2.2.2.3 Water motion

Water motion was determined at each site using a dissimilar metals technique outlined in McGehee (1998). Basically, this method involved holding two metals that are dissimilar on the Noble scale in contact with one another and measuring the amount of electrolysis that occurred over time as weight loss. The more water motion the metal pair is exposed to, the faster this electrolysis occurs. This method integrates water motion at a site over a relatively long period of time (days-weeks) because the metals in question are not ‘used up’ very quickly (McGehee 1998). Unlike in McGehee (1998), relative rather than absolute water motion was calculated in this study due to the variable nature of water motion between sites (current vs surge).

The dissimilar metals used were bars of zinc and copper which were held together using segments of bicycle inner tubes. Zinc segments were 5.7 cm long with a 0.9 cm diameter and the copper cylinders were 7.5 cm long by 1.2 cm in diameter. Each metal bar was uniquely identified by position and weighed prior to being placed in the field. Station units were assembled by strapping 4 paired metals to a 1/2” PVC pipe such that none of the metal pairs touched one another (Figure 2.2). The PVC pipe was then secured to a brick. Three station units were randomly situated within the subtidal portion of the eelgrass bed at each site and marked at the surface with a float. They were submerged on the same day at every site and left in the field for 12-14 days (except the measurements at Robbers N and S in August 2005 which

were only submerged for 9 days). After this time they were retrieved from the field and returned to the lab where the metal segments were separated and soaked in 10% hydrochloric acid solution for 3 minutes, rinsed with fresh water, dried and weighed. The weight loss of each zinc segment was recorded. This value was divided by the number of days the metal pair had been submerged to yield a value of weight lost per day. The weight of the copper segments was also recorded to check that they had not changed or been mixed up.

Water motion was evaluated at all sites in July and August of 2006. Additional water motion measurements were taken in May 2006 for Roquefeuil, Dodger and Fleming sites. Robbers N and S were also measured in July and August 2005.

2.2.2.4 Water clarity

On three sampling days in August 2006 water clarity measurements were taken using a 20 cm diameter black and white Secchi disc. These measurements were taken at all five sites on the same day. Where water clarity depth exceeded the depth of the bed, measurements were taken in water beyond the subtidal edge of the bed.

2.2.2.5 Bed dimensions

The perimeters of all 5 eelgrass beds in this study were mapped using Garmin GPSMap76 units. Mapping protocols provided by the BC community mapping network were followed (see reports by: Precision Identification Biological Consultants and Environment Canada 2002; Geostreams Consulting 2004). Intertidal edges of each bed were mapped by walking the perimeter at low tide and recording waypoints on the GPS unit every 10-15 paces. The subtidal edges were mapped using SCUBA: divers identified the subtidal edge of the bed

and marked it using a float attached to a small lead weight every 10 fin kicks. At the surface, the boat tender recorded the location of these floats with the GPS unit. All GPS points were subsequently downloaded into a shapefile using Garmin extension v5.1.1 (2001 Minnesota Department of Natural Resources) and uploaded into ArcMap v. 9.1 (2005 ESRI Geographic Information System) which was used to convert the points into polygons and calculate their areas. ArcMap was also used to calculate the dimensions of the beds. Maximum and minimum bed depths were determined using a SCUBA depth gauge and meter stick respectively, and subtracting the calculated tide height at the time of measurement from those values.

2.2.2.6 Proximity to kelp habitats

Kelp beds adjacent to the study sites were delineated using GPS units and uploaded into ArcMap in the same manner as the eelgrass perimeters. The distance from these beds to the eelgrass habitat was measured using ArcMap v. 9.1.

2.2.2.7 Eelgrass characteristics

Several eelgrass characteristics were measured once during the summer field season within a four week period from the end of July to the end of August of the appropriate year. Table 2.2 shows the dates that each depth and site were sampled. All eelgrass characteristics were measured using stratified random sampling along transects in each of the zones present at the sites. All transects were laid parallel to shore within the central portion of the bed to avoid edge effects. In 2005, quadrats (50 cm x 50 cm) were laid at 15 random points along these transects, at random distances (max 5 m) from the transect (except for subtidal transects where low visibility prevented divers from straying from the transect line), and on alternating sides. In 2006 subtidal shoot density counts were taken along the growth transect line at the same seven

randomly placed locations as the growth measurements (Figure 2.3) for a total of seven, rather than 15 quadrats per depth. Below are the methods for the various measurements that were taken within each quadrat (items i-vii):

- i. Shoot density: All above ground shoots originating from within these quadrats were counted, and vegetative and reproductive shoots were enumerated separately.
- ii. Biomass determination: Within each large quadrat, a 10 cm x 10 cm quadrat was placed in the bottom left corner. If no shoots were found in that location the small quadrat was placed in the upper right corner and so on until a corner was found that contained shoots. If no shoots were found in any corner, a biomass sample was not taken for that quadrat location. Once the small quadrat was positioned, all above ground plant material that originated from within the quadrat was removed and placed in an individually labeled bag. This plant material was returned to the lab and refrigerated until it could be measured. Processing involved rinsing the eelgrass shoots with saltwater to remove debris and epiphytes. Excess water was squeezed from the blades and a wet weight of the sample was recorded. The sample was then placed in a foil container and dried in an oven at 60 °C until a constant weight was attained. The dry weight of the sample was recorded and standardized to a unit of 1 square meter (g/m^2).
- iii. Length and width of blades: These measurements were taken for up to three randomly chosen shoots from the biomass sample prior to desiccation. The length of the longest leaf was measured in cm with a tape measure from the end of the leaf sheath to the leaf tip. The leaf width (mm) was measured a few centimetres above the sheath using callipers.
- iv. Sheath length: The length of the sheath was measured from the meristem to the end of the sheath using a tape measure (cm). This measurement was not taken in 2005.
- v. Total plant length: This was calculated as the sheath length + leaf length (not possible for 2005 data)

vi. Number of leaves per shoot: For every shoot that was measured, the number of leaves was also recorded. To do this, the senescing sheath was removed and the remaining leaves were counted and recorded.

vii. Leaf area index (LAI): Calculated as mean plant length (m) X mean leaf width (m) X mean shoot density (shoots/m²).

2.2.2.8 Eelgrass growth

a. Field season 1 (2005)- Two sites were evaluated in this field season- Robbers S and Robbers N. Four transect lines were located at each site: high, mid and low intertidal, plus a general subtidal one. Three growth plots were set up at random locations along each of these transect lines at each site (total = 12 per site). Each growth plot was marked with 60 cm of rebar that was driven into the substrate and labeled with large amounts of flagging tape (for subtidal plots these markers also included a surface float). The locations of the plots were recorded with a waypoint in a GPS unit and the Lat/Long recorded. Fifteen eelgrass shoots were chosen in close proximity to each rebar stake and were marked with approximately 20 cm of flagging tape that was attached below the substrate to the rhizome using a small cable tie. On each of these shoots the newest leaf was marked 5 cm above the sheath with a hole-punch. Care was taken to ensure that the removed area spanned no more than $\frac{1}{3}$ of the leaf.

Every two weeks the growth plots were visited and as many flagged shoots as possible were located. The distance between the leaf sheath and the punched area was measured. The punched leaf was then marked so that it would not be re-measured at subsequent visits, and the newest leaf was punched as above. If less than 15 shoots at a plot were found, or were measurable, new shoots were tagged to make the total number of tagged plants equal to 15. The

distance the leaf grew, minus the original 5 cm, was recorded into a spreadsheet and divided by the number of days growth it represented to yield units of cm/day.

b. Field season 2 (2006)- The method for determining growth at the eelgrass sites was modified after the 2005 field season to address issues of repeated measures on the same plants over time. To ensure that methodology was comparable across all sites, the subtidal areas of the 2005 sites were re-evaluated in 2006 using the new methodology. From July 14-18, 30 m leaded transect lines were laid at random locations within each subtidal zone of all five study sites (for a total of eight transects). They were held in place at each end with lead cannonballs and marked at the surface with a float. Depths of each transect line are shown in Table 2.3. Seven growth plots were randomly located on each transect line and marked with flagging tape.

Underwater, divers located the first marked location on each transect and positioned a 1 m x 1 m quadrat so that it was to the right and below the flagging tape, thus marking out the first quarter of the area to be measured over subsequent weeks (Figure 2.3). The rhizome of one plant in this area was marked with flagging tape. Using a hole punch, all leaves of approximately 10 plants in close proximity to the marker were punched 5 cm above the sheath. The quadrat was then moved to the second marked location on the growth transect and the process repeated. Seven plots were set up in this manner on each transect line.

Every two weeks these growth plots were sampled and setup again. On these visits, five plants punched on the previous visit were located at each plot, removed, and placed in a labelled Ziploc bag. A total of 35 plants were collected in this manner along each transect line. In the lab the bags were drained of water and frozen until they could be processed. At each plot a new quadrat was also set up; on the second sampling date these quadrats were all located to the right

of the marker tape and above the previous plot, and on the third sampling day they were located diagonally opposite the first plot (see Figure 2.3).

In September 2006, all frozen samples were thawed and processed. The leaf length, width, and sheath length were measured for each plant. The distance between the plant sheath and the punch mark was measured on each leaf, and the total length of all new leaves also recorded. Total growth for the plant was recorded as the sum of the distance between the sheath and the punch mark minus 5 cm for each leaf, plus the length of all new, un-punched leaves. This value was divided by the number of days of growth it represented to yield a growth measurement in units of cm/day. It represents the total amount of plant material produced by the plant per day.

2.2.2.9 Epiphyte biomass

a. Field season 1 (2005): Ten plants were haphazardly chosen from each of three intertidal depths at Robbers S and N on August 24, 2005 and placed in labelled plastic bags. Samples were frozen upon return from the field until they could be processed. Processing involved removing the end 30 cm from the three oldest leaves of each plant and measuring their width. These leaves were then placed in a shallow pan, wetted with seawater and scraped gently with the side of a razor blade to remove all of the attached epiphytes and fauna. The epiphytes from each plant were filtered through a Whatmann filter and dried at 60 °C. The weight of the dried epiphyte material on the filter paper was later recorded. This weight was standardized by dividing by the area of the leaf it was scraped from (determined by the length and width of the blade) to yield epiphyte biomass in units of $\text{g}\cdot\text{dwt}/\text{cm}^2$. It should be noted that since epiphyte density on most plants decreased from leaf tip to base, and the top 30 cm contained the majority of epiphytes, these measures might overestimate the abundance of epiphytes over the entire leaf.

b. Field season 2 (2006): The abundance of epiphytes on eelgrass leaves was measured concurrently with growth in 2006 for subtidal areas of all 5 sites. The methods were slightly changed from 2005 in order to relate these measurements with growth and eelgrass characteristic measurements and are therefore not directly comparable with 2005 data. At three of the seven growth plots along a transect, three of the five plants collected for growth measurements were carefully placed in separate, labelled bags (total = 9 plants per transect). These samples were refrigerated and processed within 48 hours of collection. Processing involved measuring the length and width of the two oldest leaves, scraping the epiphytes from them, and drying and calculating biomass as per 2005.

2.2.3 Data analysis

Variables were checked for a normal distribution and equal variance prior to all analyses. If these assumptions were met (with transformations where necessary), parametric tests were used, and where they were not met, non-parametric equivalents were used instead. All factors in the analyses were fixed. An alpha level of 0.05 was used to indicate a significant effect except where multiple comparisons were made; for these cases, the alpha level was adjusted using Bonferroni corrections for multiple comparisons (Townend 2002: 118). All univariate analyses were performed in JMP IN version 4.0.4, and multivariate analyses in SAS version 9.1 unless otherwise indicated.

2.2.3.1 Temperature

Mean temperature and associated standard errors were calculated for each site. A Kruskal-Wallis test was used to look for differences in temperature between sites followed by

multiple Mann-Whitney U tests on pairwise comparisons when significant overall differences were found.

2.2.3.2 Salinity

Mean salinity and associated standard errors were calculated for each site. A Kruskal-Wallis test was used to look for differences in salinity between sites followed by multiple Mann-Whitney U tests on pairwise comparisons when significant overall differences were found.

2.2.3.3 Water motion

These data were analyzed using non-parametric tests due to heteroskedasticity and small sample sizes. The variability of water motion at the sites was compared across the 2 years (2005/2006) and 3 months (May, July, August). Wilcoxon rank-sum tests were used to look for differences between Robbers N and S in July and August of both years. Water motion was also compared between months and sites in 2006 using a Kruskal-Wallis test followed by multiple Mann-Whitney U tests on pairwise comparisons when significant overall differences were found.

2.2.3.4 Eelgrass characteristics

2.2.3.4.1 *Univariate analyses*

These data were analyzed using non-parametric tests due to heteroskedasticity and small sample sizes. Eelgrass characteristics were compared individually across sites using Kruskal-Wallis tests, followed by multiple Mann-Whitney U tests on pairwise comparisons when significant overall differences were found. Deep and shallow transects were analyzed

individually. Differences between shallow and deep transects at Dodger, Robbers S and Roque were compared with Wilcoxon rank-sum tests.

2.2.3.4.2 Multivariate analysis

Beds with an intertidal component were analyzed with a two-way MANOVA (Factor 1= site- Robbers S and N; and factor 2= tidal height- high, mid, low, subtidal). The eelgrass variables used in the analysis were: square-root transformed shoot density (shoots/m²), log transformed biomass (g/m²), leaf length and width. Although transformations were used, the assumptions of parametric tests were not strictly met, therefore the results were interpreted with caution.

A one-way MANOVA was also used to look for differences between the five sites using eelgrass characteristics from the shallow subtidal. Variables used in this analysis were: leaf width, plant length, log transformed biomass and shoot density. Posthoc comparisons were performed subsequently using an adjusted alpha value to look for differences between all site pairs.

2.2.3.5 Eelgrass growth

Parametric analyses were used to analyze measurements of growth, because homoskedasticity was achieved through a log transformation of the data, and sample sizes were large. Two-factor ANOVAs were used to look for an effect of site and sampling date on growth measurements at five study sites at both subtidal depths in 2006. T-tests were also used to look for differences in growth between deep and shallow zones of each site (sampling periods were originally analyzed separately to ensure that trends were consistent across the dates, and then data were pooled to simplify analyses).

Because methods used in 2005 and 2006 were different, growth could not be compared between the two years. Growth at Robbers N & S in 2005 was analyzed separately with a two-way ANOVA (factors= site and tidal height). Data from two sampling dates in August were pooled for this analysis (no significant differences were found between the dates) and the low intertidal zone was omitted because plants were never recovered at one site.

2.2.3.6 Epiphyte biomass

All epiphyte data were square-root transformed to meet assumptions of normality and equal variance for parametric tests. Epiphyte data were collected with slightly different methods in 2005 compared to 2006 and thus, the data from the two years were analyzed separately. Data from 2005 were available for three intertidal heights at Robbers N and S. They were analyzed with a two-factor ANOVA to look for effects of site and tidal height.

Epiphyte data from the three sampling dates in 2006 were compared with an ANOVA to look for a significant effect of date at each site and depth. Since a significant difference between the sampling dates was found for only 2/8 comparisons (Roque-deep: $p=0.013$ & Fleming: $p=0.035$), data from the three sampling periods were combined in further analyses.

The combined epiphyte data were then analyzed with a one-way ANOVA to look for differences between sites at each subtidal depth, followed by a Tukey's multiple comparison test when significant differences were found. These data were also analyzed with a two sample t-test to look for differences between depths at the three study sites with a deep zone.

The mean epiphyte biomass in August 2006 averaged over all depths was regressed with Secchi depths from August 2006 to determine whether there was an effect of water clarity on epiphyte abundance.

2.2.3.7 Multivariate analysis of selected habitat variables

To address the question of which variables best distinguish the sites from one another, a multiple discriminant analysis (MDA) was performed. It was not possible to use all variables discussed in this chapter because replication scales were not equivalent for all of them; therefore, only variables that were measured at the plant level in 2006 were included in this analysis. They included: growth, leaf width, plant length, sheath length, plant weight, and epiphyte abundance. The last two variables were log transformed to meet assumptions of univariate normality. Correlations between the original variables and the discriminating functions were calculated to determine which were most responsible for the separation of the sites seen in the MDA.

2.3 RESULTS

2.3.1 Temperature

Temperature ranged from 12 to 18 °C in 2006 (Figure 2.4, Table 2.4). There were significant differences between sites for this variable ($X^2=28.2$, $df=4$, $p<0.0001$); Fleming was significantly higher than all sites except Robbers N.

2.3.2 Salinity

The range of salinity measurements in 2006 was from 27.5 to 37 ppt (Figure 2.5, Table 2.4). There were significant differences in salinity between sites ($X^2=10.4$, $df=4$, $p=0.0339$), but multiple comparison tests revealed that only two sites differed significantly from one another; Dodger had significantly higher salinity than Fleming.

2.3.3 Water motion

Water motion was variable over time and space (Table 2.5). Despite the integrative nature of the method used, significant differences in water motion were found between measurements in the same month of different years, and between summer months of the same year for some sites (Figure 2.6). At Robbers S, water motion was significantly different between August 2005 and 2006 ($X^2=16.56$, $df=1$, $p=0<.0001$), but not in July for the same years ($X^2= 0.3424$, $df 1$, $p=0.5585$). Mean water motion at Robbers N was also higher in August 2005 than August 2006, however, this difference was not significant ($X^2= 0.285$, $df=1$, $p=0.0917$). In all comparisons between 2005 and 2006 water motion was both higher and more variable in 2005.

For repeated measurements in 2006, significant differences were found between months for two sites (Figure 2.7). Water motion at Fleming was significantly higher in July than in May or August ($X^2=12.41$, $df=2$, $p=0.002$), and water motion at Roque was also significantly higher in July than in August or May ($X^2=10.88$, $df=2$, $p=0.0083$). Water motion did not differ between summer months for the other three sites.

There were also differences in water motion between the study sites. In 2005, combined measurements at Robbers N were significantly higher than at Robbers S ($X^2=5.07$, $df=1$, $p=0.0243$). When all monthly measurements in 2006 were pooled within a site, significant differences between sites were found ($X^2=98.77$, $df=4$, $p=<0.0001$) (pooled data not shown), Roque was significantly higher than all other sites except Fleming. All other pairwise comparisons were significant except between Fleming and Robbers N.

2.3.4 Water clarity

Secchi depths averaged 3.9 m, 2.6 m, 2.6 m, 3.0 m, and 3.2 m at Dodger, Fleming, Robbers N, Robbers S and Roque respectively. Nine measurements were taken at every site except Roque and Robbers N where high wind precluded measurements on one sampling day; there were six measurements at these sites.

2.3.5 Bed dimensions

The size of eelgrass beds in this study ranged from to 1,828 m² to over 22,000 m². The maximum depth of the beds ranged from 0.6 m at Fleming, to 5.2 m at Dodger. The area, length, width and depth distribution of each bed are shown in Table 2.6.

2.3.6 Proximity to kelp habitats

Kelp habitats were close to many of the eelgrass beds in this study. Distance from the center of the eelgrass bed to the nearest kelp bed ranged from 30 m at Roque to 364 m at Robbers S. Distance from the kelp beds to the closest edge of the eelgrass ranged from 0 m at Roque and Dodger, to 326 m at Robbers S (Table 2.7).

2.3.7 Eelgrass characteristics

2.3.7.1 Univariate analyses

Eelgrass characteristics were variable both between depths, and between sites (Figure 2.8 a-f).

a. Shoot density- Shoot densities ranged from 16 shoots/m² at Dodger-deep, to 369 shoots/m² at Robbers N. Comparisons of densities in deep strata showed significant differences between all three sites; Roque > Robbers S > Dodger (n=7or15, X²=17.30, df=2, p=0.0002). Significant differences were also found between sites along shallow transects (n=7 or 15, X²=30.75, df=4, p=<0.0001). At this depth, densities at Dodger were significantly lower than all other sites.

Table 2.8 indicates where the other pairwise differences occurred. Shoot density was higher on shallow transects than deep transects at all sites with deep portions (Dodger n=7, X²=9.12, df=1, p=0.0025; Roque n=7, X²=9.06, df=1, p=0.0026; Robbers S n=15, X²=21.45, df=1, p=<0.0001)(Figure 2.8a).

b. Above-ground biomass- Biomass in the deep region of the beds ranged from 134 g/m² to 508 g/m² (Figure 2.8b). It was significantly higher at Roque compared to both Dodger and Robbers S (n=7or15, X²=7.74, df=2, p=0.0208). There were also significant differences between the shallow portions of the five eelgrass beds (n=7 or 15, X²=23.70, df=4, p=<0.0001). Fleming had the overall highest amount of above ground biomass (500.0 g/m²) and Robbers S-shallow had the lowest (87.5 g/m²). Post hoc analyses indicated that Fleming and Robbers N were significantly higher than Dodger and Robbers S. There were no significant differences between shallow and deep areas of any site (Dodger n=7, X²=1.05, df=1, p=0.3056; Roque n=7, X²=0.33, df=1, p=0.5653; Robbers S n=15, X²=0.53, p=0.4677) however, at Roque it was substantially higher in the deep zone.

c. Leaf width- Mean leaf width ranged from 4.24 mm at Robbers N to 7.96 mm in the deep portion of Dodger (Figure 2.8c). In the deep zone there were no significant differences in leaf width between the three sites (n=7or15, X²=1.02, df=2, p=0.6001); however, in the shallow subtidal zones of the five sites, significant differences in leaf width were found (n=7 or 15, X²=24.75, df=4, p=<0.0001). Table 2.9 indicates where these differences occurred. Although

leaf widths only differed significantly between shallow and deep zones at Robbers S (Robbers S: $n=15$, $X^2=15.31$, $p<0.0001$; Dodger: $n=7$, $X^2=0.08$, $df=1$, $p=0.7745$; Roque: $n=7$, $X^2=2.98$, $df=1$, $p=0.0845$), mean leaf width was greater with depth at all sites.

d. # leaves/shoot- The number of leaves per shoot ranged from 2.64 at Dodger-shallow to 4.90 at Roque-deep. There were no data from Robbers S. There were significant differences in number of leaves between sites on deep transects ($n=7$, $X^2=8.72$, $df=1$, $p=0.0031$) as well as for shallow transects ($n=7$ or 15 , $X^2=10.90$, $df=4$, $p=0.0123$). On the shallow transect plants at Dodger had significantly fewer leaves than at Roque and Robbers N however, this difference may have been due to the fact that Dodger was sampled the latest in the summer when plants had begun to shed their outer leaves that were highly epiphytized (personal observation). The mean # leaves per shoot was generally greater on deep transects than shallow, but these differences were not significant for Dodger ($n=7$, $X^2=0.006$, $df=1$, $p=0.938$), or for Roque ($n=7$, $X^2=3.49$, $df=1$, $p=0.0616$) (all data shown in Figure 2.8d).

e. Plant length- Mean plant lengths are shown in Figure 2.8e. Plants were longest at Fleming (166.6 cm), and shortest at Robbers S (60.3 cm). They did not vary significantly between the deep subtidal zones of the study sites ($n=7$ or 15 , $X^2=1.07$, $df=2$, $p=0.5855$), but differences were significant between the shallow subtidal zones ($n=7$ or 15 , $X^2=22.22$, $df=4$, $p=0.0002$). Plants at Robbers S were significantly shorter than plants at Robbers N and Fleming. Shallow versus deep comparisons were only significant at Robbers S ($n=15$, $X^2=17.38$, $df=1$, $p<0.0001$) however, plants at Roque were also considerably longer in the deeper than shallow zones (106.8 cm versus 76.9 cm).

f. Leaf area index (LAI)- These data are shown in Figure 2.8f. In this study, a comparison of shallow LAI across sites revealed significant differences ($n=7$ or 15 , $X^2=24.28$, $df=4$, $p<0.0001$). There was a lot of overlap in LAI measurements, but the site with the highest LAI,

Fleming, was significantly higher than the lowest, Robbers S. Table 2.10 shows all other pairwise comparisons of LAI in the shallow zone. Of the three sites where a deep transect was measured, Roque had the highest LAI (0.596), followed by Robbers S (0.356), and finally Dodger (0.134). The differences between all three of these sites were significant ($n=7$ or 15 , $X^2=15.09$, $df=2$, $p=0.0005$). At all of these sites LAI was lower on deep than on shallow transects, but this difference was only significant at Dodger ($n=7$, $X^2=8.16$, $df=1$, $p=0.0043$).

2.3.7.2 Multivariate analyses

A two-way multivariate ANOVA on 2005 data showed a highly significant effect of site and tidal height when shoot density, biomass, leaf length and leaf width measurements were considered (site: $df=1$, $F=166.30$, $p<0.0001$; tidal height: $df=3$, $F=6.0$, $p<0.0001$). The interaction effect of site and tidal height was also highly significant ($df=3$, $F=6.5$, $p<0.0001$) indicating that the effect of tidal height on the four variables was not consistent among sites (Figure 2.9 a-d). A MANOVA using 2006 data revealed more differences between sites than the univariate tests alone (Wilks Lambda test result: $F=12.17$, $p<0.0001$). Posthoc comparisons indicated that all sites were significantly different from one another except Fleming and Robbers N ($p=0.029$), Fleming and Roque ($p=0.082$), or Dodger and Roque ($p=0.012$).

2.3.8 Eelgrass growth

Figures 2.10 and 2.11 show the amount of eelgrass growth on shallow and deep transects respectively, at all sites, and on each of the sampling dates in 2006 (end July, mid August and late August), plus their associated standard errors. Growth ranged from 1.17 cm/day at Dodger-shallow, to 2.19 cm/day at Roque-shallow at the end of August 2006. In the shallow zones,

Robbers N and S had very similar growth trends across the three months. Other sites had differing growth patterns from month to month (as indicated by the significant interaction term in the 2-way ANOVA). Growth at Roque increased throughout the summer, whereas growth at Dodger declined steadily.

Two-factor ANOVAs indicated that the effects of site and sampling date were different on shallow versus deep transects in 2006. On the shallow transect there was a significant effect of site (F-ratio=26.17, df=4, $p < 0.0001$), but not of date (F-ratio=2.20, df=2, $p = 0.1120$), whereas on deep transects both factors were significant (site: F-ratio=23.49, df=2; date: F-ratio=35.71, df=2, $p < 0.0001$). There was also a significant 'site' x 'sampling date' interaction for both depths indicating that eelgrass growth at the sites was not consistent across time (shallow: F-ratio=10.4, df=8, $p < 0.0001$; deep: F-ratio=13.67, df=4, $p < 0.0001$).

T-tests revealed that the effect of depth on growth was highly variable. At Dodger, growth was significantly higher in the deep zone (F=5.2, df=1, $p = 0.0241$), whereas at Robbers S, growth was actually higher in the shallow zone (F=9.68, df=1, $p = 0.0021$) and at Roque there was no significant difference between the depths (F=0.82, df=1, $p = 0.3657$).

Analysis of data from 2005 (when a different, and less accurate growth measurement method was used) revealed significant effects of tidal height (df=2, F=15.09, $p < 0.0001$) and site (df=2, F=44.57, $p < 0.0001$), plus a significant interaction term (df=2, F=12.03, $p < 0.0001$) when a two-factor ANOVA was used on data from Robbers N and S (Figure 2.12). At all tidal heights, eelgrass growth was greater at Robbers N than S. This difference was statistically significant for each height except the subtidal. Eelgrass growth trends were similar between sites in the high intertidal and subtidal zones, but in the mid intertidal at Robbers N, eelgrass growth was higher than any other site or height in the analysis (which lead to the significant interaction term). Growth at these sites (as measured by the 2005 method and therefore not

comparable to 2006 data) ranged from 1.46 cm/day in the high intertidal at Robbers S, to 3.23 cm/day in the mid intertidal at Robbers N.

2.3.9 Epiphyte biomass

A two-factor ANOVA on square-root transformed epiphyte data from 2005 revealed a significant effect of site ($df=1$, $F=35.99$, $p<0.0001$), but not of tidal height ($df=2$, $F=1.65$, $p=0.2026$) on epiphyte biomass. Epiphyte biomass was much higher at Robbers S than at Robbers N (Figure 2.13). There was no significant interaction between the two factors ($n=2$, $F=0.58$, $p=0.5638$). Epiphyte biomass in subtidal zones in 2006 ranged from 0.0058 g·dwt/cm² at Roque-shallow to 0.0324 g·dwt/cm² at Robbers S-shallow (Figure 2.14). Significant differences were found between the shallow zones of the 5 study sites ($df=4$, $F=28.62$, $p<0.0001$), but not between the deep zones ($df=2$, $F=0.155$, $p=0.8572$). Multiple comparison tests revealed that, of the shallow zones, Robbers S had a significantly higher epiphyte biomass than all other sites, and that Fleming and Robbers N were significantly higher than the remaining two sites. T-tests revealed that there were significant differences between shallow and deep zones at all three study sites compared (Dodger: $df=1$, $F=4.93$, $p=0.0320$; Robbers S: $df=1$, $F=30.33$, $p<0.0001$; Roque: $df=1$, $F=14.08$, $p=0.0007$). At two of these sites (Dodger and Roque), epiphyte biomass was higher in the deeper zones, and at Robbers S, it was higher in the shallow zone.

No relationship was found between epiphyte biomass and water clarity (Secchi depths) ($R^2= 0.34$, $p= 0.31$).

2.3.10 Multivariate analyses of selected habitat variables

An MDA provided insight into which variables are most important for distinguishing the sites. It revealed that the five study sites could be discriminated from one another using two discriminant functions (Wilk's Lambda test result: $p < 0.0001$) (Figure 2.15). A correlation analysis between the six original variables and these two functions shows that the first discriminating function is highly correlated with leaf width ($r = 0.84$), and the second with plant weight ($r = 0.56$) and growth ($r = 0.57$) (Table 2.11).

2.4 DISCUSSION

Ultimately in this thesis, the data presented in this chapter will be used in a correlation analysis with juvenile rockfish abundance to assess which variables influenced the habitat value of the beds. However, these data are also of scientific interest in their own right and worth discussing in detail. Eelgrass beds are generally well studied, but the majority of our knowledge comes from the Atlantic coast of North America. Studies that compare multiple *Z. marina* beds on the west coast of North America are scarce in the published scientific literature (although much long-term monitoring has been done at multiple locations on the coast by Parks Canada in British Columbia (Robinson and Martel 2007) and by Washington Dept. National Resources in Puget Sound (Gaeckle *et al.* 2007)). Within the seagrass literature, studies that evaluate beds in terms of multiple variables are also rare. Most focus on one to several variables of direct interest, but none could be found that described eelgrass beds in terms of 16 variables (listed in Table 2.1), as was done in this study.

2.4.1 Temperature

Ambient temperatures in eelgrass beds can effect the growth and survival of eelgrass plants, their response to epiphyte loads, respiration rates, and possibly the nursery value of the bed. In this study, temperatures occurred over a narrow range (12-18 °C) compared to the tolerance range of the plants (0-30 °C), due to the latitude of the sites, and the limited number of months that measurements were taken.

Eelgrass growth rates are known to increase with increasing water temperature up to 21 °C, after which eelgrass growth declines (Olesen and Sand-Jensen 1993). Very high temperatures can kill eelgrass plants. In fact, at the southern extreme of their range, eelgrass populations are essentially annual, because high summer temperatures kill all eelgrass plants on a yearly basis (Phillips *et al.* 1983). Temperatures in this study, despite being collected during summer months, were never greater than 18 °C, suggesting that growth and survival was not limited by high temperatures during the study period. Exceptions to this may have occurred for brief periods in shallow areas during low tides, but these events were never recorded as temperatures were always taken within the eelgrass canopy during diving operations and therefore may not reflect the more extreme temperature events occurring in areas too shallow for SCUBA.

At the other end of the range, low temperatures have been found to retard growth and increase the amount of light required for compensation (Olesen and Sand-Jensen 1993). On the eastern coast of North America extremely low temperatures can indirectly effect eelgrass when ice sheets form and scour intertidal plants. This phenomenon creates an annual population that regrows from seeds after the ice events (Robertson and Mann 1984). In this study, growth was only measured during summer months and was therefore not limited by low temperatures,

although it is likely that study sites with intertidal regions such as Robbers N and S, would experience temperatures low enough to retard growth during the winter.

In warmer latitudes, temperature affects growth in other ways: the germination of *Z. marina* seeds is thought to be stimulated when water temperatures decline to 15 °C (Moore *et al.* 1993). In this region, however, no effect of temperature has been found on seed germination (Harrison 1991; Phillips *et al.* 1983). High water temperatures can also increase the effect of algal epiphytes on the growth of *Z. marina* plants, leading to decreased growth compared with plants with similar epiphyte loads and lower temperatures (Neckles *et al.* 1993). Given the similarity of water temperatures between sites during the months of maximum epiphyte productivity (summer), it is unlikely that temperature caused a difference in the effect of epiphyte load during this study, however.

The most important implication of temperature differences found in this study may be on the value of eelgrass beds for juvenile fish, rather than on the growth or survival of the plants themselves. The impetus for juvenile rockfish movement into deeper water with age has been attributed to ontogenetic changes in temperature preferences (Love *et al.* 1991). Because of this, fish at warmer sites may leave eelgrass habitat sooner than at cooler sites and not fully benefit from the increased growth and protection offered by eelgrass. If an earlier emigration stimulated by warm temperature affects the survival or future fitness of the juveniles it could be argued that higher temperatures negatively impact nursery value.

2.4.2 Salinity

Eelgrass can persist in salinities ranging from 10-39 ppt. In this study, the average salinity was 33.6 ppt (SE=0.31) during the summer of 2006, and significant differences were found between two sites only- Dodger and Fleming (Figure 2.5). These measurements were

only taken over a 4 month period. Three sites, however, have no freshwater inputs and are not likely to vary considerably throughout the rest of the year either (Dodger, Fleming and Roque). Robbers S and N, on the other hand, had small streams that emptied directly into the eelgrass beds and the potential for much lower salinities during rain events. At Robbers N low volumes of water flowed through the stream throughout the summer, whereas at Robbers S, water was only observed in the stream after one rainstorm. Had salinity measurements been taken later in the year, and when maximal stream flows would be encountered, there may have been more differences among sites.

Reduced salinities have been shown to stimulate seed germination in eelgrass plants (Harrison 1991; Phillips *et al.* 1983), although the salinities required for this would likely only be encountered by seeds in intertidal areas (Moore *et al.* 1993). This is supported by Phillips *et al.* (1983), who stated that in the central portion of its range, eelgrass reproduces largely by vegetative growth in the subtidal region where salinity fluctuations are minimal. It is likely that any salinity differences found in this study were not an influential factor on seed germination in the subtidal areas of the 5 sites. At the two sites with extensive intertidal eelgrass, and ephemeral fresh water streams, however, low salinity events may play a role and stimulate germination of the seed bank.

2.4.3 Water motion

The dissimilar metal technique used to measure water motion in this study records water motion over a longer period of time than most other ‘snapshot’ methods (e.g. plaster blocks or ‘lifesavers’ methods), incorporating small temporal fluctuations in water motion (McGehee 1998). This method is preferable to other methods for this study as it yields a more relevant picture of the amount of water motion experienced by plants and animals that permanently

inhabit a seagrass bed. Relative rather than absolute measures of water motion were used in this study, because the study sites experienced a range of water motion types from surge to tidal current.

In my study, the water motion units were left submerged at the study sites for 8-14 days. Despite the fact that measurements were integrated over a period of more than a week, the amount of zinc weight loss per day was significantly different from year to year, and between summer months, indicating that water motion is highly variable in time. These differences further justify taking measurements over a long period of time because snapshot measurements of the sites would have been even more variable.

Water motion was also significantly different between sites. Zinc weight loss averaged over the sampling months in 2006 indicated that Roque had the highest overall water motion, followed by Fleming, Robbers N, Dodger, and finally Robbers S. These between-site differences may lead (either solely, or in combination) to differences in eelgrass characteristics and fish communities. It has been shown, for example, that the growth, architecture and morphology of *Z. noltii* is significantly affected by differing levels of water motion (Peralta *et al.* 2007), and that seagrass percent cover is influenced by the degree of water motion (Fonseca and Bell 1998). On a larger scale, Frederiksen *et al.* (2004a) demonstrated that the distribution and shape of eelgrass beds is influenced by wave action. Currents and wave action are also known to influence the dispersal and burial of eelgrass seeds, thereby facilitating bed expansion by several meters annually in sexually reproducing populations (Orth *et al.* 1994). In terms of fish communities, McGehee (1994) found that fish assemblages in Puerto Rico were influenced by water motion more than any other environmental factor tested. For these reasons it was thought that water motion would be an influential factor in determining the species composition and abundance of juvenile rockfishes; a relationship that will be tested in Chapter 5.

Measurements of water motion using a dissimilar metal technique ranked the five study sites in terms of wave action as would be expected by considering the aspect of each of the sites, with significant differences found between most sites. However, water motion such as tidal currents seemed to cause less electrolysis of the zinc rods than I expected based on personal experience at the sites. Unlike the other four sites that experienced varying degrees of wave action, but had no noticeable current, Dodger experienced high current flows without any wave action (personal observation). Despite these high currents, Dodger was ranked as though there was very little water motion (Figure 2.7). In a comparative study of gypsum dissolution rates between different flow regimes, Porter *et al.* (2000) found that dissolution rates of gypsum cards differed between water motion types and cautioned against using a dissolution method to compare between flow regimes. The dissimilar metal technique used in this study incorporates water motion using similar principles, and may have the same limitations. Based on this, characterizing Dodger using the low relative amount of water motion that we measured with the dissimilar metals technique may be an inaccurate representation of the site compared to others in the study. It may also lead to an underestimation of the value of that site for fish as higher currents can provide higher amounts of recruits to a site (Carr and Reed 1993b; Jenkins *et al.* 1997a), as well as continuous feeding opportunities to the fish that have settled there (Wankowski and Thorpe 1979; O'Brien and Showalter 1993). However, it is also possible that the lower levels of water motion recorded at Dodger were a function of the low temperature at that site compared to the others since it can lead to slower electrolysis rates and subsequently lowered amounts of weight loss.

2.4.4 Bed dimensions

In my study, multiple aspects of eelgrass bed morphology were measured including: bed dimensions, maximum depth limit of the plants, and bed size; many of which were significantly different between sites. Factors that are known to influence these seascape scale measurements include the availability of substrate (Koch 2001); water depth and light availability (Duarte 1991; Dennison 1987); and the direction, magnitude and type of water motion (Fonseca *et al.* 2007; Frederiksen *et al.* 2004a). The size, shape and depth of an eelgrass bed have implications for its habitat value (Bell *et al.* 1992; Jackson *et al.* 2006), and for other ecosystem functions such as wave baffling, and improving water quality (Bartleson 2004).

Bed dimensions are largely influenced by water motion. High amounts of current or wave action tend to create long beds parallel to the direction of prevailing currents and waves (Fonseca *et al.* 2007)(as seen at Dodger, Robbers N and Roque -Figure 2.1). This orientation has implications for the ecosystem functioning of the bed: longer beds attenuate water flow within the bed more fully, resulting in greater deposition of larvae and sediments, higher water clarity, and a greater decrease in dissolved organics and nutrients (Bartleson 2004). In my study, Fleming was the one site with considerable wave action that proved an exception; the greatest width of the bed was actually perpendicular to the direction of incoming waves.

The maximum depth limit of a seagrass bed is thought to be controlled by light availability (Duarte 1991; Olesen and Sand-Jensen 1993; Greve and Krause-Jensen 2005) which in turn is influenced by water depth, water clarity, weather and surface conditions (Hemminga and Duarte 2000). At two sites in this study (Robbers N and Fleming) the maximum depth of the bed was shallower than would be expected by light limitations alone; despite ample substrate, eelgrass did not grow deeper than 1.0 m (Table 2.6). It is speculated that eelgrass did not penetrate deeper at these sites because of high amounts of wave action and the shallow

nature of the depth contours. Wave energy entering an eelgrass bed is dissipated over shallow areas, and can cause uprooting of plants where edges and patches occur. This action has been shown to erode away the subtidal edge of shallow beds, effectively making them smaller and preventing them from expanding into deeper areas (Fonseca and Bell 1998). This was in fact observed at Robbers N; after winter storm events at the start of 2006 the subtidal zone was smaller, and the edge more patchy compared to the same month in 2005.

Restricted maximum depths, such as those at Robbers N and Fleming can limit the size of eelgrass beds, which will have negative consequences for ecosystem functioning. For example, bed size has been positively correlated with fish abundance (Jelbart *et al.* 2006), although results from other studies have been equivocal (McNeill and Fairweather 1993; Johnson M.W. and Heck 2006). This may be due to the fact that small beds are less able to entrain larvae and will have limited recruitment compared to larger beds (Bartleson 2004). Smaller beds are also less resilient to physical disturbances and less persistent over time (Fonseca and Bell 1998). A recent report on the status of eelgrass in Puget Sound attributed large seagrass declines in Hood Canal to the small size of beds in the region and their reduced resistance to hydrodynamic forces (Gaeckle *et al.* 2007). Beds that are unstable in size and over time are a less reliable habitat for macrofauna, and may have reduced value as a nursery if juveniles persist for more than one season. In this study, Fleming (at only 47 m wide and 1828 m² in area) was the most likely site to be limited in size by wave action due to this reduced capacity for 'self-protection', and is most at risk of declining in size in time.

Smaller beds also have limited capacity for improving water clarity and quality. Bartleson (2004) found that narrow beds (<100 m wide with respect to direction of water motion) did not alter the environmental conditions within the bed compared to the surrounding area because they did not adequately reduce water velocity through the bed. This resulted in

less nutrient reduction and improvements to water clarity (via sedimentation) in the narrower beds. In my study, both Roque and Fleming were less than 100 m wide, although Fleming was the most narrow at only 23 m suggesting that it may have reduced ecosystem functioning compared to the other sites. Of the sites in this study, it seems that Fleming may be the least valuable for a number of reasons due to its restricted size and dimensions, and the impacts these features may have on nursery value for fishes, and the provision of ecosystem services.

2.4.5 Proximity to kelp habitats

The proximity of canopy forming kelp to an eelgrass bed may have direct effects on the health and distribution of eelgrass plants, as kelp is a fast-growing alga that may outcompete *Z. marina* for light and nutrients. Eelgrass decline has been attributed to increased amounts of macroalgae within beds in several studies (Hauxwell *et al.* 2001; Hauxwell *et al.* 2003). Kelp adjacent to eelgrass beds could prohibit or limit the growth of nearby eelgrass, or restrain the size of a bed by shading nearby plants. At Roque and Dodger, a kelp bed was in direct contact with the edge of the eelgrass bed, and potentially limited the light available to nearby plants. Furthermore, at Dodger kelp was attached to rocks within a sandy area, rather than on bedrock thereby taking up habitat that could otherwise have supported eelgrass growth and thus, may have limited the size of the eelgrass bed.

Close proximity to kelp habitats, however, may also be of some value to the eelgrass plants that are not close enough to be shaded by the kelp canopy, because kelp forests are known to baffle wave energy at the water surface (Mork 1996), and reduce currents through the bed (Rosman *et al.* 2007). Because of this, if fortuitously placed, kelp beds could limit the overall water motion experienced by eelgrass plants and thus protect them from being eroded or torn up (Koch 2001). Adjacent kelp habitat at Roque was positioned in such a way; a large perennial

bed was located between the eelgrass and incoming waves and may be a contributing factor to the persistence of a uniform bed of eelgrass despite high levels of incoming wave action.

Some studies have also found that the position of a seagrass bed relative to the surrounding landscape may be the largest factor influencing fish assemblages within the bed (Dorenbosch *et al.* 2006; Hemminga and Duarte 2000). Therefore, the proximity of an eelgrass bed to a vegetated rocky reef where adult rockfish are present may be an important factor in determining the number of juveniles that recruit to it. Sites like Dodger and Roque that were in direct contact with kelp beds may have higher levels of recruitment due to the proximity of adult spawners or the migration of juveniles between the habitats. The differences in proximity to kelp habitat could therefore be a contributing factor to differences in juvenile rockfish settlement patterns that will be discussed in future chapters.

2.4.6 Eelgrass characteristics

Eelgrass plants vary from site to site and between depths of the same site because their morphology is plastic and able to respond to differing abiotic conditions (Hemminga and Duarte 2000; Rhode and Duffy 2004). The plasticity of *Z. marina* plants was noticed on the west coast of North America as early as 1970 (den Hartog 1970) and led to the distinction of eelgrass ecotypes. Presently, five ecotypes of the species *Z. marina* are recognized on our coast, the first three of which can be found in BC. They include: *typica*, *phillipsi*, *latifolia*, *ata'm* and *izembek* (Backman 1985). Based on leaf width and habitat, the intertidal eelgrass at Robbers S and all eelgrass at Robbers N is most likely of the ecotype *typica*, whereas all subtidal eelgrass at the remaining four sites is likely the ecotype *phillipsi*.

Widely varying morphological characteristics have been reported in the literature for many seagrass species (Hemminga and Duarte 2000; Rhode and Duffy 2004), and as we saw in

this study, *Z. marina* is no exception. All of the eelgrass characteristics measured in this study were found to be significantly different between sites (at least in the shallow subtidal zones), and, with the exception of biomass, between subtidal depths. The most predictable differences in seagrass morphology are those that occur between depths. For instance, the length and width of eelgrass plants are known to increase with water depth (Precision Identification Biological Consultants and Environment Canada 2002). Indeed, in this study plant length was longer at two of the three study sites with a deep zone, and leaves were wider at all sites at depth (although only some of these differences were significant). However, this trend is not conserved throughout all seagrass species, as decreases in leaf width or length with water depth have been reported for some tropical species (Collier *et al.* 2007; Lee and Dunton 1997).

Conversely, eelgrass shoot density, LAI and biomass are commonly reported to decrease with depth (Dennison and Alberte 1986; Duarte 1991; Enriquez and Pantoja-Reyes 2005; Peralta *et al.* 2002). Significantly lower shoot densities, and to some degree LAI, were found in the deeper subtidal zones of all sites in this study (Figure 2.8a,f). It has been suggested by many researchers that these trends are an adaptation by plants to reduce self-shading at depth (Enriquez and Pantoja-Reyes 2005; Collier *et al.* 2007; Zimmerman 2003) and this could certainly be the case in the present study. However, unlike many other studies, no significant differences in biomass were found between depths in this study. Measures of biomass in shallow and deep zones were almost the same at two sites, and at Roque, biomass was actually higher in the deeper zone (likely due to an increase in the # leaves with depth). At these study sites overall biomass seemed to be maintained by the increased width, length, and to some degree # leaves, of the remaining plants as the number of shoots decreased with depth.

It is likely that these intra and especially inter-bed differences will have consequences for the ecosystem functions provided by eelgrass beds. For instance, it has been shown that

increased canopy height and shoot density reduce current flows, thereby increasing deposition of larvae and sediments (Peterson *et al.* 2004), that increased leaf width offers more habitat for epifauna (Hemminga and Duarte 2000), and that longer sheath lengths are positively correlated with growth (Gaeckle *et al.* 2006).

2.4.7 Eelgrass growth

In my study, plant growth is reported in units of cm/day, which is generally referred to as 'leaf elongation' in the literature (Peralta *et al.* 2002). Most seagrass growth studies, however, report growth as g·dwt/day which makes it difficult to compare the results of this study to most of what has been reported in the literature. However, trends that are seen using traditional measures may also hold for leaf elongation, therefore general comparisons may be made between the present study and others that report in g·dwt/day (although Olesen & Sand-Jensen (1993) found that eelgrass in Denmark maintained leaf elongation at light levels which were inadequate for biomass gain). For this reason, this discussion will be limited to comparisons of trends, rather than absolute values.

Eelgrass growth is largely controlled by light availability (Dennison 1987). Light levels in subtidal regions of an eelgrass bed can be very low due rapid attenuation of light in nearshore waters. Eelgrass plants however, have morphological and physiological adaptations that allow them to live at considerable depths and still constitute one of the most productive benthic ecosystems in the ocean (Hemminga and Duarte 2000). Despite these adaptations, most studies do find that growth rates decline to some extent with depth. For example, in 2002, Peralta *et al.* found that leaf elongation in *Z. nolteii* was significantly and negatively affected by water depth.

The present study, however, does not entirely support those findings. At Roque there were no significant differences in growth between depths, and at Dodger higher growth was

recorded in the deep zone on all three of the sampling dates. Robbers S was the one site in this study that did have significantly lower growth in deeper zones. A similar study on *Posidonia sinuosa* in Australia also failed to find a difference in growth between shallow and deep zones despite a 70% reduction in light, and attributed the maintenance of growth rates at depth to reduced shoot density (Collier *et al.* 2007). This is a possibility in the present study, especially at Dodger where reduced self-shading was likely because the shoot density in the deep zone was only 17% of shallow density. Neither water depth nor Secchi depth were significantly correlated with eelgrass growth, therefore these factors are not useful covariates to explain these differing relationships.

2.4.8 Epiphyte biomass

Seagrass epiphytes are considered to be any plant or animal that lives in close association with seagrass plants (Borowitzka *et al.* 2006). Significant differences in epiphyte biomass were found between sites in this study. Four out of ten pairwise comparisons were significant. These differences can result in varying levels of plant productivity and growth, habitat complexity, as well differences in food abundance and quality for juvenile fishes, and may ultimately contribute to differences in overall habitat quality between the sites.

The fact that significant differences in epiphyte communities were found within the spatial scale of this study (max distance between sites <10 km) is not uncommon; Saunders *et al.* (2003) found that epiphyte assemblages differed significantly at sites separated by as little as 10's of meters. In this study, epiphyte biomass was also compared between subtidal depths of beds, and significant differences were found for the three sites examined, although the pattern of higher abundance was not consistent between the sites. It was expected that epiphytes would decrease with depth due to decreased light levels as this has been documented by many other

studies (e.g. Borowitzka *et al.* 2006). At Dodger and Roque, however, epiphytes were more abundant on deep transects, and only at Robbers S were they more abundant in the shallow zone. Measurements of water clarity did not help to explain these patterns as no significant effect of Secchi depth on epiphyte biomass was found. Conflicting effects of depth were likely found because floral and faunal types were not evaluated separately in this study. Studies on epiphyte composition have shown that the community structure of epiphytes changes with depth (Pinckney and Micheli 1998), and it is likely that depth affected functional groups differently at the three study sites and produced inconsistent patterns. Perhaps if algal epiphytes only had been evaluated more consistent results would have been obtained.

Increased epiphyte abundance usually leads to reduced plant growth because of lowered light availability to the leaves (Penhale 1977; Brush and Nixon 2002). More detailed studies on seagrass epiphytes however, have demonstrated that the relationship is actually more complicated due to the antagonistic effects of floral versus faunal epiphytes. It has been shown that faunal epiphytes in the form of grazers help to control the amount of floral epiphytes on eelgrass leaves, and can even mediate expected negative effects of nutrient inputs and increased temperature on eelgrass growth (Neckles *et al.* 1993; Heck *et al.* 2000). In this study, floral and faunal epiphytes were not distinguished and it is likely because of this that there was only a very weak negative relationship between epiphyte abundance and growth. Another contributing factor to the lack of a strong relationship between growth and epiphyte abundance may be the type of floral epiphytes. Bladed epiphytes may actually break up boundary layers along leaves allowing increased diffusion rates of photosynthetic gasses and nutrients resulting in increased growth (Hemminga and Duarte 2000).

As mentioned, differences in epiphyte communities, including overall abundance, may affect the habitat value of a site for fishes. Epiphytes can positively influence fish communities

by offering larger surface areas for hiding (in the case of foliose epiphytes like *Smithora* sp.), or by acting as a food source (in the case of faunal epiphytes). For instance, Jelbart *et al.* (2006) found a positive correlation between juvenile fish abundance and epiphyte abundance, although they only considered floral epiphytes. The reason for this relationship was speculated to be increased habitat complexity. In the present study, some faunal epiphytes were found in the stomachs of juvenile rockfishes (Figure 2.16) suggesting a role for them as a food source for macrofauna in eelgrass beds. Also, other studies have found increased fish abundance in artificial seagrass units conditioned with epiphytes than without (Bologna and Heck 1999).

2.4.9 Multivariate analyses

In my study, it was determined through an MDA that leaf width was the primary metric for distinguishing sites from one another, followed by plant weight and growth. The interpretation of these results is hampered by the fact that no similar analyses could be found in the literature thus precluding comparisons. For example, Carty (2003) did report large differences in leaf width between sites in the same geographic area, confirming the variability that was detected in the analysis, but did not test whether this variation was the most important factor for distinguishing the sites.

It was hypothesized at the beginning of this chapter that differences in abiotic and biotic site characteristics will correspond to differences in habitat value between the study sites. If this is the case, then the most important variables for distinguishing sites should also be the most important for influencing habitat value. Eelgrass is reportedly of high value to juvenile fishes because it provides protection from predation and a high food supply. Leaf width was the most important variable for distinguishing sites in this study, and may influence habitat value by increasing the potential of a plant to hide juvenile fish.

However, epiphytes are also potential sources of shelter, as well as food, for juvenile fishes, and have been linked to differences in nursery value in other studies (Bologna and Heck 1999). If this is the case in Barkley Sound it would be expected that epiphytes would be one of the more important variables in defining the sites. However, they played a small role in distinguishing sites in this study (see low r values in Table 2.11). It is possible, however, that since I only measured the abundance of epiphytes without considering the species composition I may have underestimated the role that epiphytes play in distinguishing seagrass sites. More detailed studies on epiphyte composition in Barkley Sound, and their effect on habitat value are needed to fully address this question.

2.5 CONCLUSIONS

In summary, I found that eelgrass beds were highly variable in terms of the variables measured, despite their superficial similarities. As discussed, this is a common conclusion in many seagrass studies, and may be generalized across continents and species, but it is an interesting finding nonetheless due to the fact that the sites were located within a small and geographically similar area. In this chapter, abiotic and biotic variables also differed between tidal heights and zones of the subtidal leading to intra-bed differences. There is less information in the literature regarding how common intra-bed differences are and what the implications of them may be.

It was hypothesized that observed variation in abiotic factors (water motion, water depth, temperature and salinity) between sites led to differences in the biotic variables, as many have been reported to influence eelgrass morphology, productivity and epiphyte abundance in the literature (For example: Fourqurean *et al.* 2003; Hackney and Durako 2004). These relationships were discussed for the variables measured.

Similarly, since differences in biotic variables can in turn affect the ecosystem functions provided by the bed, as well as the macrofaunal communities that reside there (Rooker *et al.* 1998a; Attrill *et al.* 2000), it is expected that factors such as ‘nursery value’ will also be significantly different between sites, and that the variables most responsible for causing inter-site differences will play an important role in affecting the ‘nursery value’. These relationships will be discussed in detail in the following chapters.

Finally, I found that a combination of eelgrass morphometrics (plant weight, leaf width), as well as leaf growth, were the most influential variables in distinguishing the sites from one another. These are the variables that should be the focus of future studies in the area that seek to quantify differences between eelgrass beds, since they best defined the beds in this study.

2.6 TABLES AND FIGURES

Table 2.1. Summary of variables presented in this chapter, and the method used to measure them.

Variable	Units	Method of measurement	Replication level
<i>Environmental</i>			
Temperature	° C	Spirit thermometer	Bed
Salinity	ppt	Refractometer	Bed
Water motion	wt·loss/day	Dissimilar metals	Bed
<i>Seascape characteristics</i>			
Bed size	m ²	GIS	Bed
Maximum depth	m	Depth guage	Bed
Distance to kelp habitats	m	GIS	Bed
<i>Biological</i>			
Shoot density	Shoots/m ²	SCUBA collections	Quadrat
Biomass	g/ m ²	SCUBA collections	Quadrat
Plant length [†]	cm	SCUBA collections	Plant
Leaf width [†]	mm	SCUBA collections	Plant
Sheath length [†]	cm	SCUBA collections	Plant
# leaves/shoot		SCUBA collections	Plant
LAI		Calculation	Quadrat
Plant weight [†]	g	SCUBA collections	Plant
Growth [†]	cm/day	SCUBA collections	Plant
Epiphyte biomass [†]	g/cm ²	SCUBA collections	Plant

[†] Entered into a Multiple Discriminant Analysis.

Table 2.2. Dates that eelgrass characteristics were measured at each site and depth. Intertidal heights include High, Mid and Low; subtidal depths include Shallow and Deep. N/A indicates that plants did not occur at that depth.

Site	High	Mid	Low	Shallow	Deep
Dodger	N/A	N/A	N/A	Aug 23, 2006	Aug 23, 2006
Roque	N/A	N/A	N/A	Aug 4, 2006	Aug 4, 2006
Fleming	N/A	N/A	N/A	Aug 14, 2006	N/A
Robbers S	July 23, 2005 June 15, 2006	July 23, 2005 June 15, 2006	July 23, 2005 June 15, 2006	July 23, 2005	Aug 3, 2005
Robbers N	July 22, 2005 June 15, 2006	July 22, 2005 June 15, 2006	July 22, 2005 June 15, 2006	Aug 2, 2005	N/A

Table 2.3. Depth (meters below chart datum) of growth transect lines placed at five study sites.

Zone	Dodger	Fleming	Robbers S	Robbers N	Roque
Shallow	0.9	0.4	0.9	0.6	0.9
Deep	2.7	N/A	3.2	N/A	1.6

Table 2.4. Mean temperature and salinity measurements at 5 study sites throughout the summer of 2006. Standard errors are shown in brackets.

Site	Mean Temp (°C)	N	Mean Salinity (ppt)	N
Dodger	12.2 (0.313)	15	35.1 (0.273)	10
Fleming	16.0 (0.325)	15	32.5 (0.568)	14
Robbers N	15.0 (0.577)	4	33.6 (1.140)	4
Robbers S	12.8 (0.333)	6	34.6 (0.245)	5
Roque	13.9 (0.450)	17	33.4 (0.681)	15

Table 2.5. Mean relative water motion values measured on multiple dates throughout the study (g·lost/day). Standard errors in brackets.

		Dodger	Fleming	Robbers N	Robbers S	Roque
2005	July	not measured	not measured		0.041 (0.0033)	not measured
	August	not measured	not measured	0.179 (0.0446)	0.079 (0.0123)	not measured
2006	May	0.053 (0.0024)	0.073 (0.0036)	not measured	not measured	0.0919 (0.0049)
	July	0.054 (0.0046)	0.106 (0.0112)	0.079 (0.0042)	0.039 (0.0024)	0.1085 (0.0024)
	August	0.050 (0.0039)	0.074 (0.0029)	0.069 (0.0099)	0.038 (0.0010)	0.0818 (0.0063)

Table 2.6. Spatial characteristics of eelgrass beds. Measurements include: area, mean width (longshore), and mean length (perpendicular to shore), and maximum depth of the eelgrass.

Site	Area (m²)	Width (m)	Length (m)	Max Depth (m)
Dodger	22,936	350	90	5.2
Fleming	1,828	47	63	0.6
Robbers N	16,513	95	193	1.0
Robbers S	7,371	85	96	3.6
Roque	3,220	47	62	2.7

Table 2.7. Distances from closest adjacent kelp habitats to two positions in each eelgrass bed: bed center, and nearest edge (m).

Site	Distance to center of bed (m)	Distance to nearest edge (m)
Dodger	167	0
Fleming	103	85
Robbers N	164	64
Robbers S	364	326
Roque	30	0

Table 2.8. Table of p-values corresponding to pairwise comparisons of shoot density measurements in shallow zones. Bold values indicate comparisons that are significant at $\alpha = 0.05$ after Bonferroni corrections for multiple comparisons.

	Dodger	Fleming	Robbers N	Robbers S	Roque
Dodger	1.0				
Fleming	0.0017	1.0			
Robbers N	0.0010	0.5489	1.0		
Robbers S	0.0024	0.0038	<.0001	1.0	
Roque	0.0026	0.0180	0.0022	0.7239	1.0

Table 2.9. Table of p-values corresponding to pairwise comparisons of leaf width measurements using multiple Mann-Whitney U tests. Bold values indicate comparisons that are significant at $\alpha = 0.05$ after Bonferroni corrections for multiple comparisons.

	Dodger	Fleming	Robbers N	Robbers S	Roque
Dodger	1.0				
Fleming	0.0178	1.0			
Robbers N	0.0003	0.0001	1.0		
Robbers S	0.0005	0.0081	0.1294	1.0	
Roque	0.0127	0.5644	0.0893	0.3407	1.0

Table 2.10. Table of p-values corresponding to pairwise comparisons of LAI measurements using multiple Mann-Whitney U tests. Bold values indicate comparisons that are significant at $\alpha = 0.05$ after Bonferroni corrections for multiple comparisons.

	Dodger	Fleming	Robbers N	Robbers S	Roque
Dodger	1.0				
Fleming	0.0017	1.0			
Robbers N	0.0150	0.3782	1.0		
Robbers S	0.7511	0.0002	0.0017	1.0	
Roque	0.9491	0.0088	0.0376	0.3319	1.0

Table 2.11. Correlations (R-values) between eelgrass variables used in the MDA and the first two discriminant functions (DF). Strong correlations constituting the major influence on the functions are highlighted in bold.

	DF1	DF2
Growth/day	-0.30366	0.566289
Plant length	-0.01246	-0.11621
Sheath length	0.254821	0.386943
Leaf width	0.843188	0.250276
Plant weight	0.244392	0.562882
Epiphytes	-0.40638	-0.06902

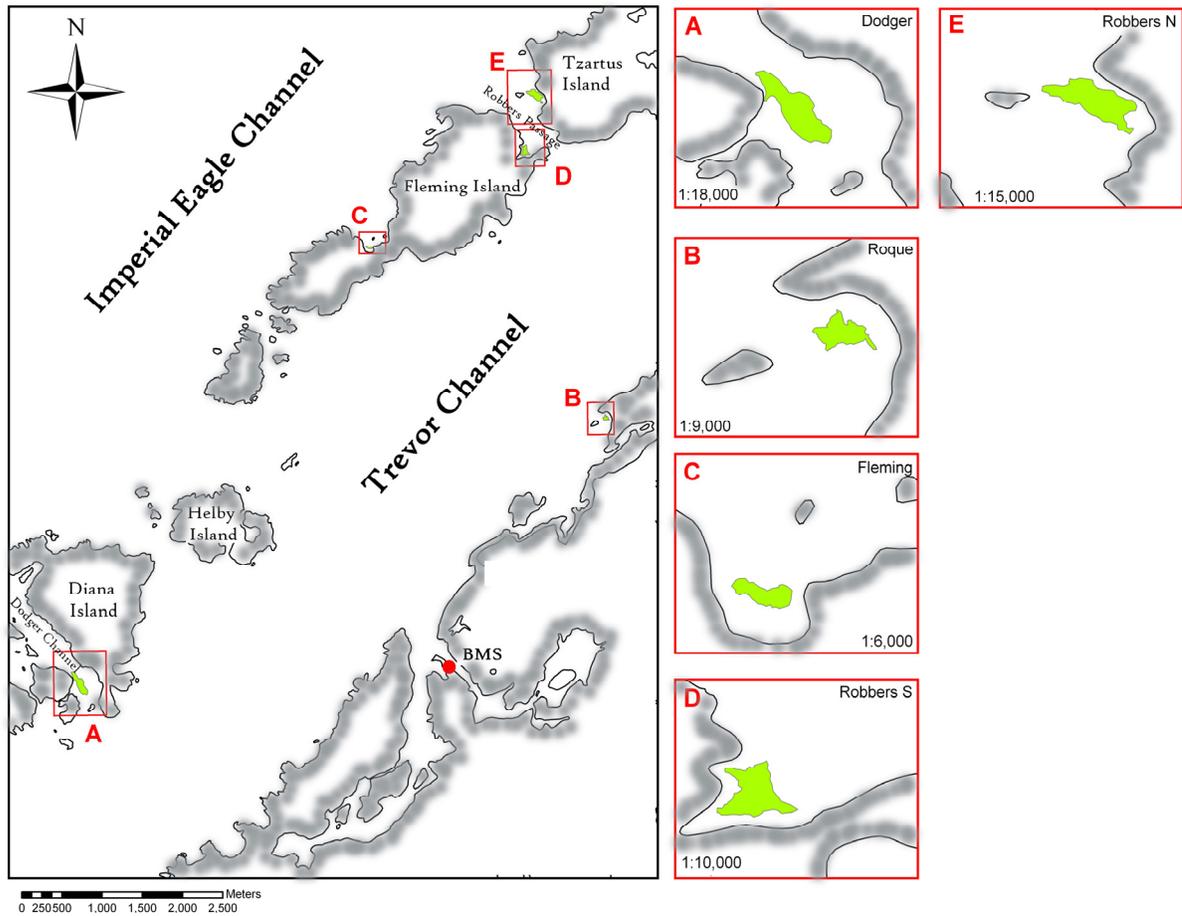


Figure 2.1. Map of study sites in Barkley Sound. Small maps to the right are more detailed versions of the correspondingly labelled extent rectangles in the larger map. Green areas represent the eelgrass beds.

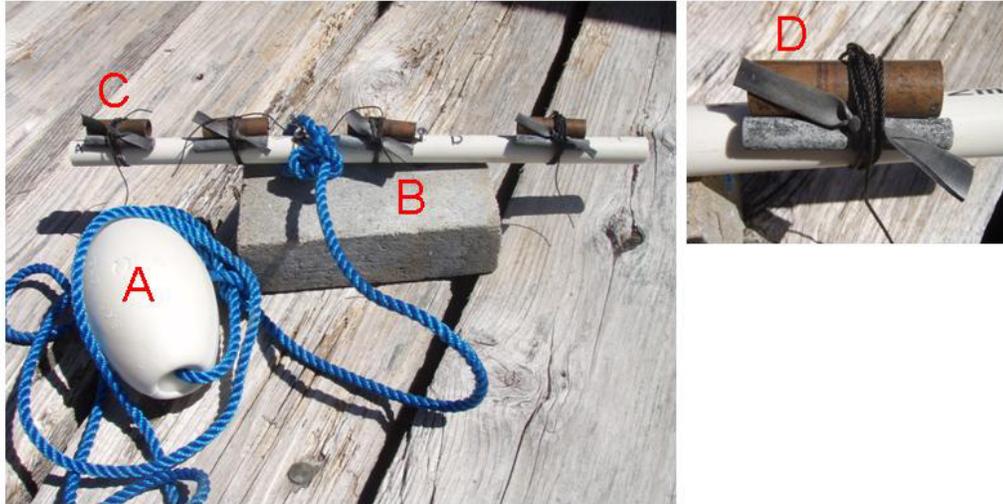


Figure 2.2. Image of a complete station unit for measuring relative water motion. A= surface float, B= anchor, C= metal pair, D= close-up of metal pair.

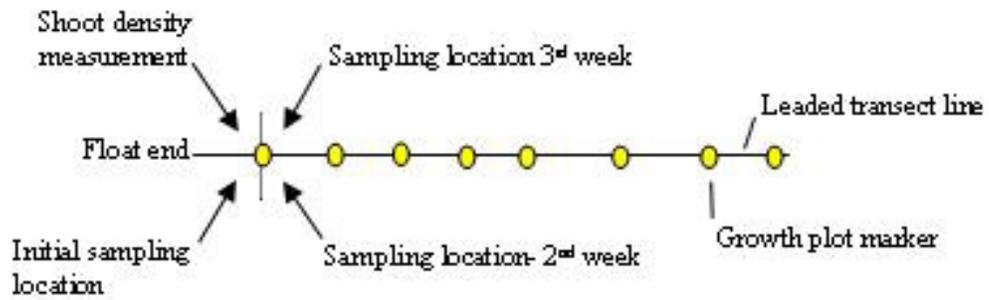


Figure 2.3. Diagram of growth transect set up showing sampling locations over 3 sampling dates.

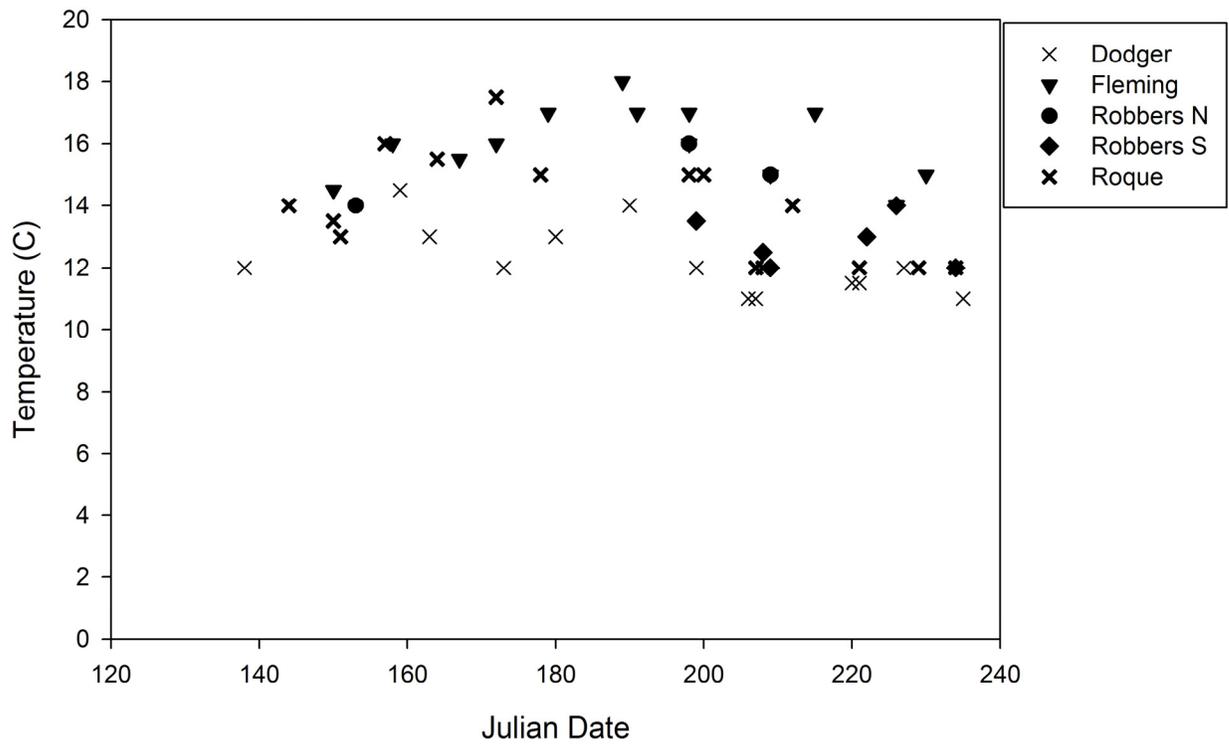


Figure 2.4. Temperature data from 5 study sites for May-Sept 2006.

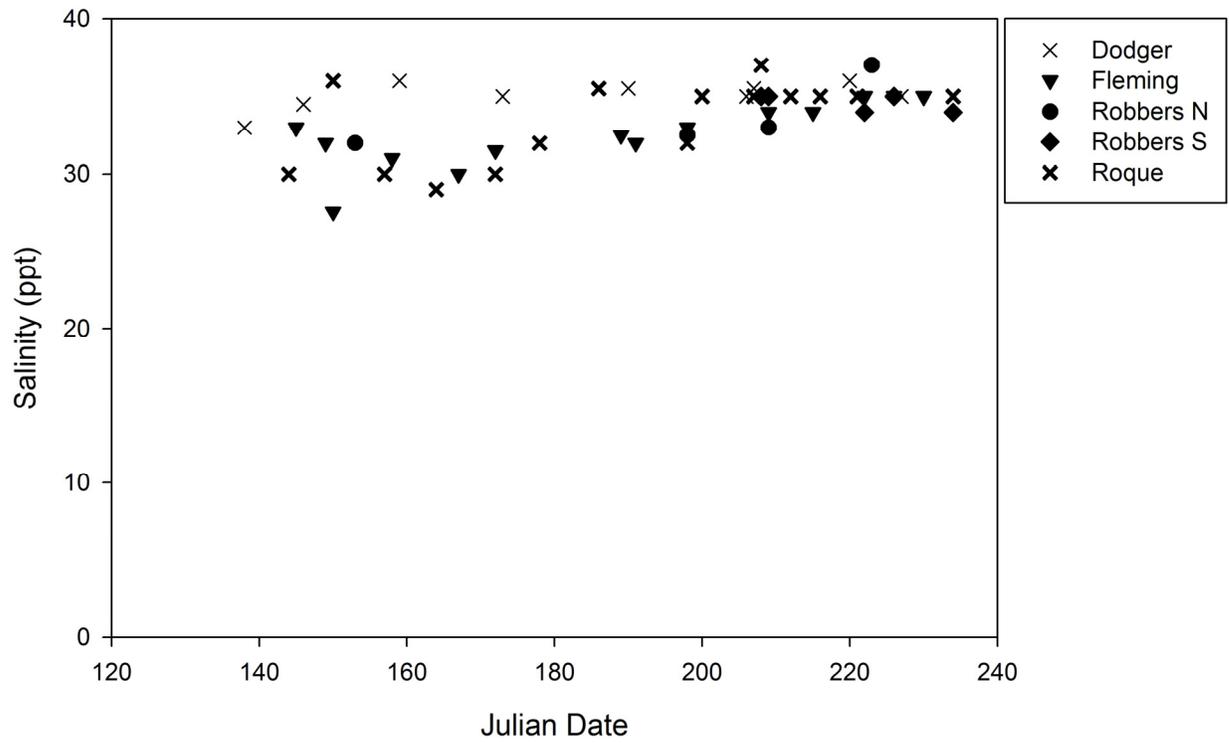


Figure 2.5. Salinity data from 5 study sites for May-Sept 2006.

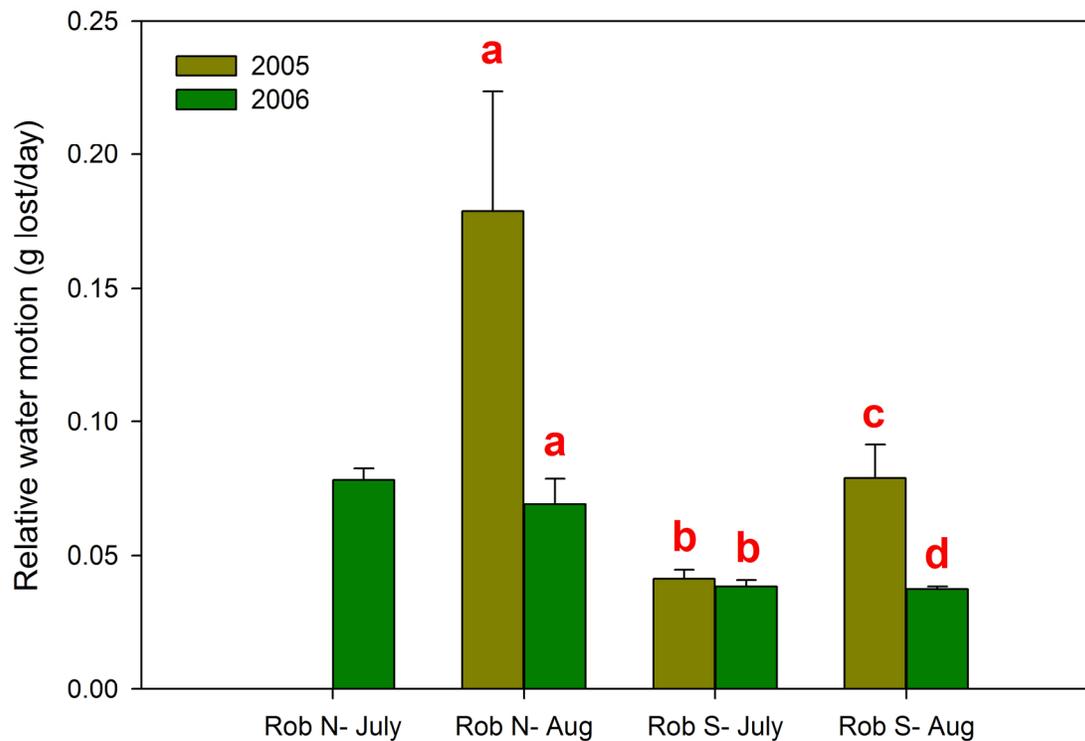


Figure 2.6. A comparison of relative water motion measurements in 2005 & 2006 at Robbers N and S. Values shown are mean values based on N=11-12, and error bars indicate standard errors. Paired bars with identical letters are not significantly different from one another and those with different letters are significantly different at $\alpha= 0.05$.

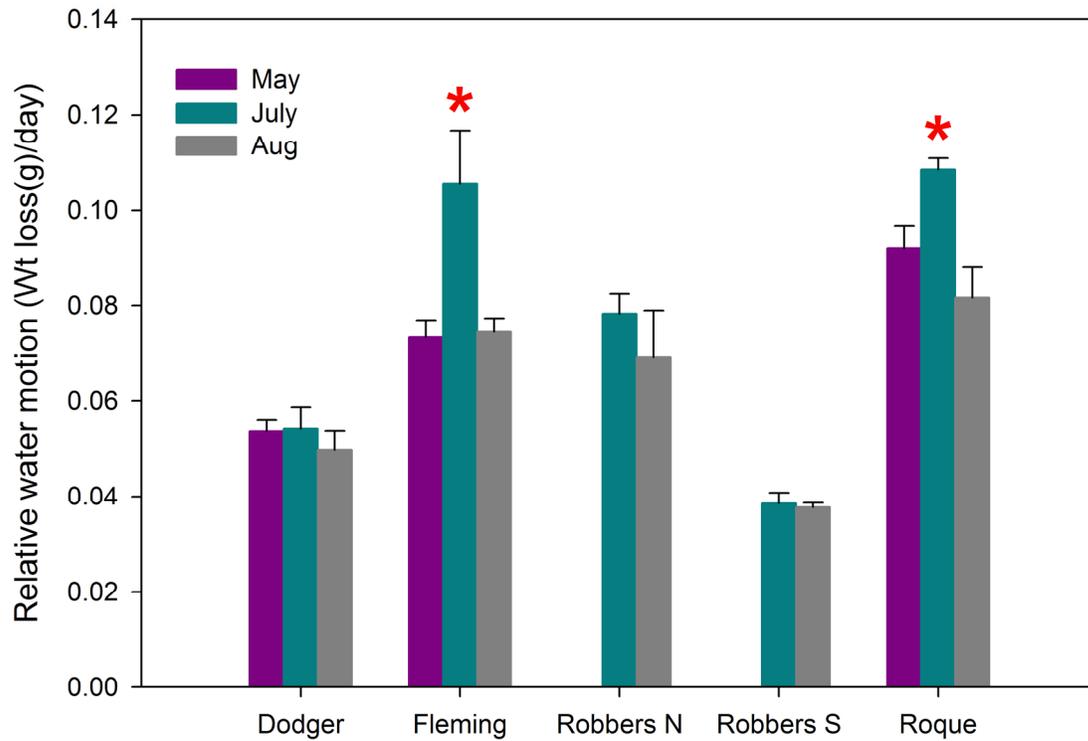


Figure 2.7. A comparison of relative water motion at 5 sites between three months in 2006. Values shown are mean values based on N=6-16 replicates, and error bars indicate standard errors. Months where water motion measures differed significantly from other months are marked with *.

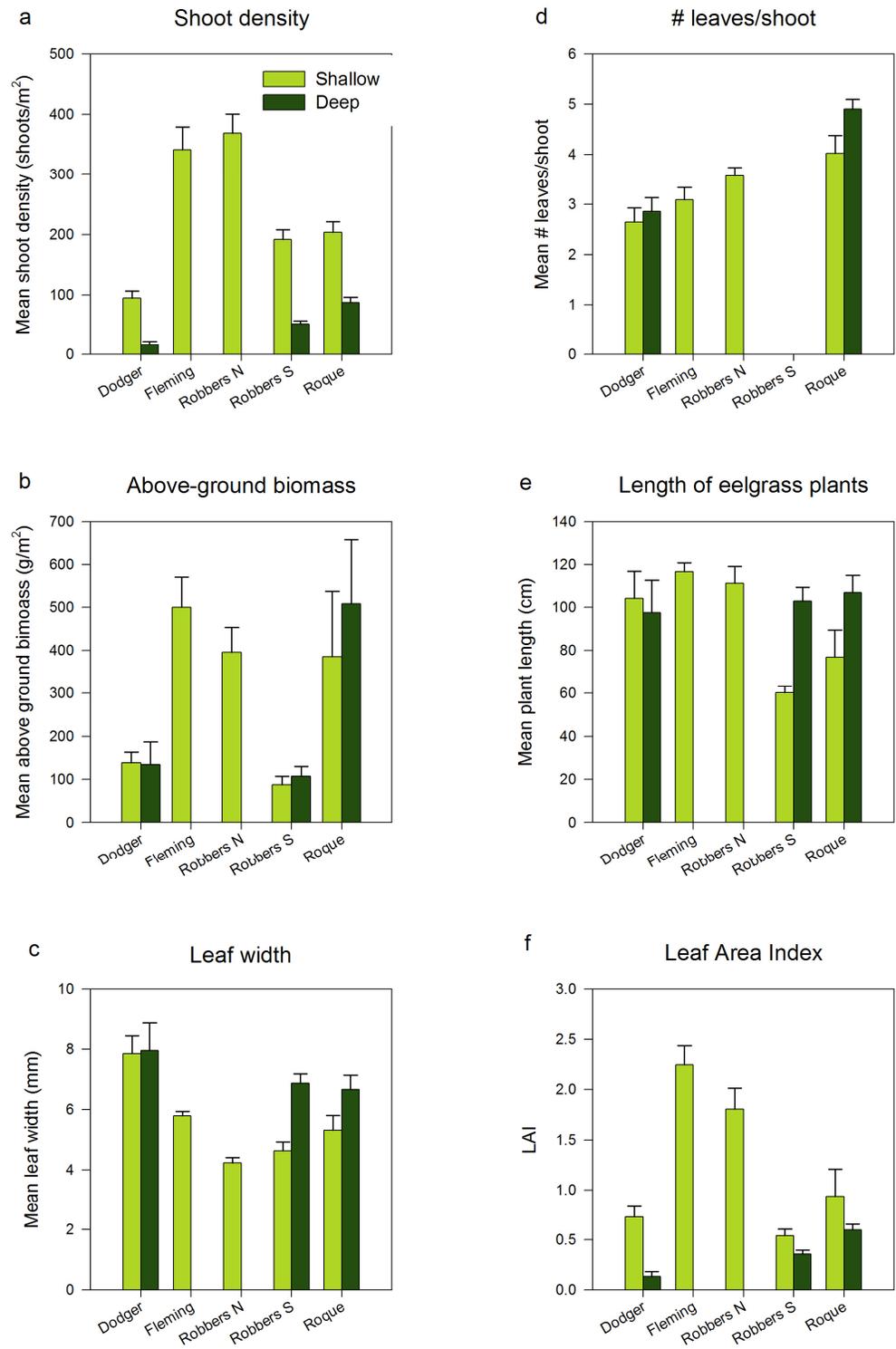


Figure 2.8a-f. Characteristics of eelgrass plants in quadrats at shallow and deep portions of 5 eelgrass sites. Data were recorded in late July or August. Robbers N and S were measured in 2005 (N=15), all other sites in 2006 (N= 7). Errors bars indicate the standard error.

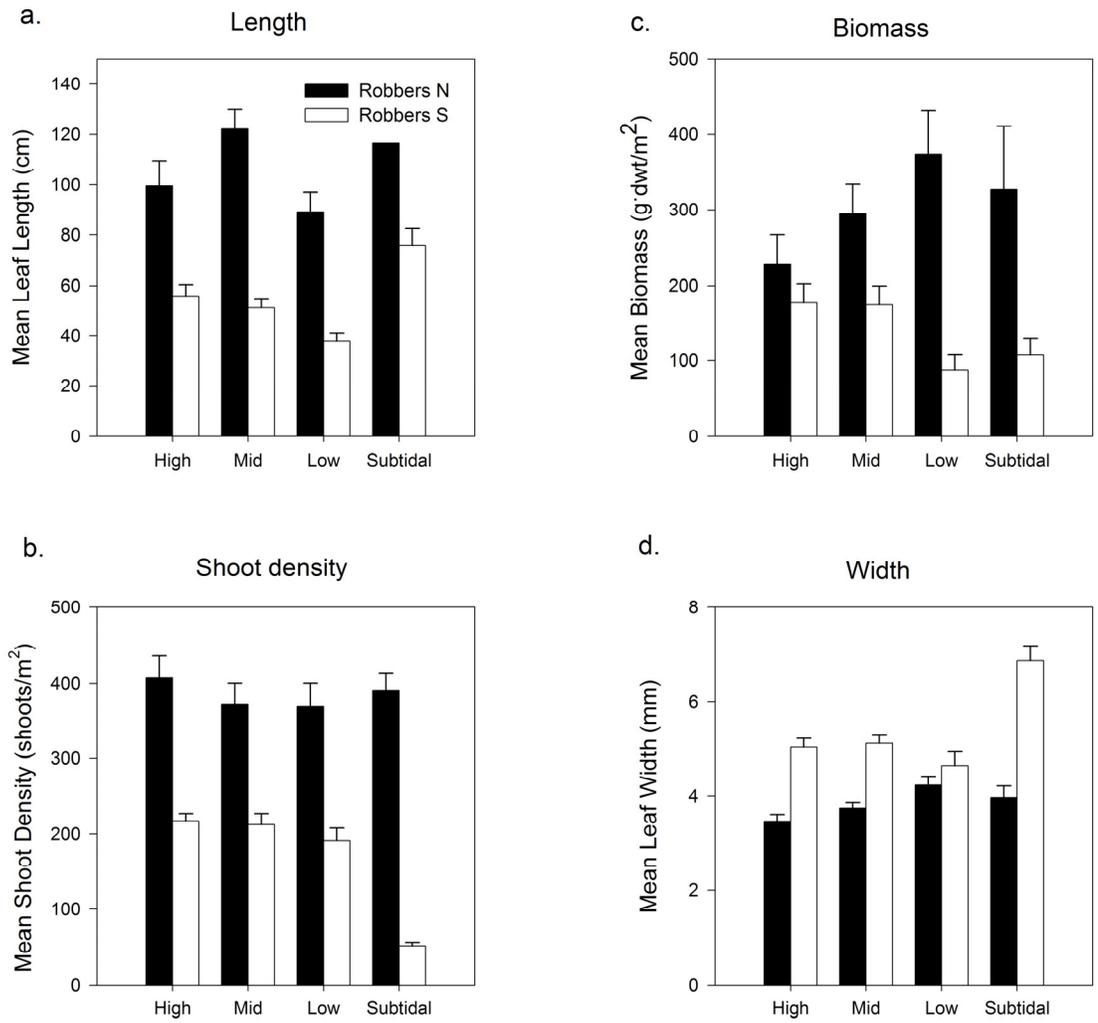


Figure 2.9a-d. Characteristics of eelgrass plants in quadrats at Robbers N and S measured in 2005. These data were used in a MANOVA. Error bars indicate the standard error.

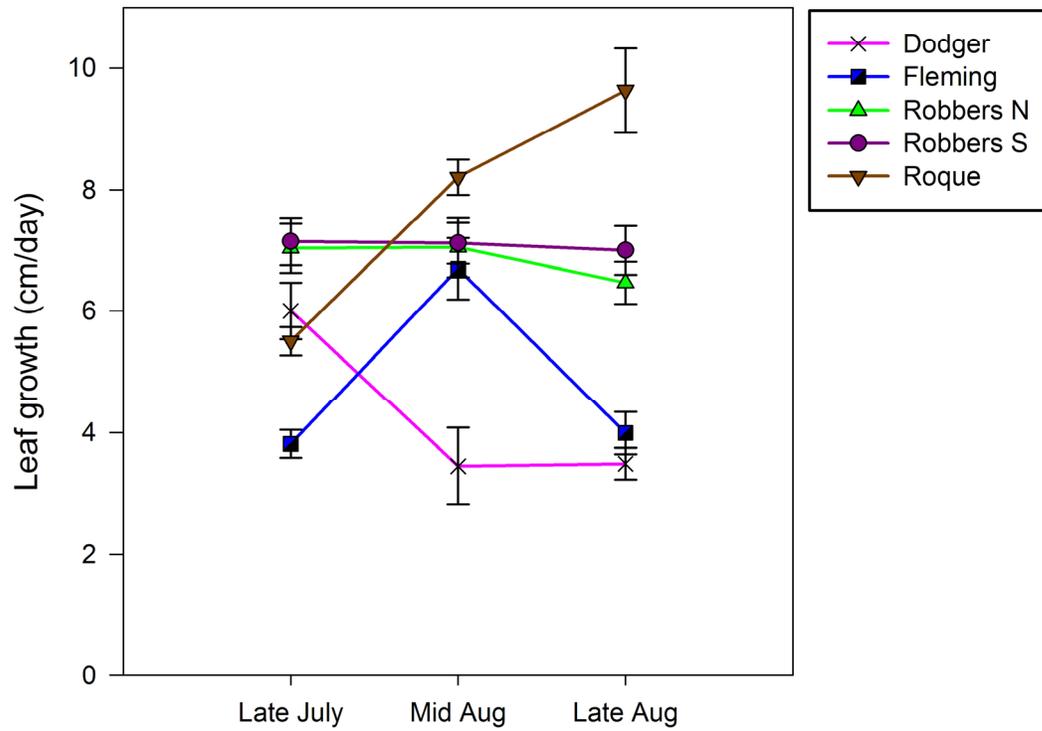


Figure 2.10. Comparison of growth in the shallow zones of 5 study sites measured in late July, mid and late Aug of 2006 (cm/day). Errors bars indicate the standard error.

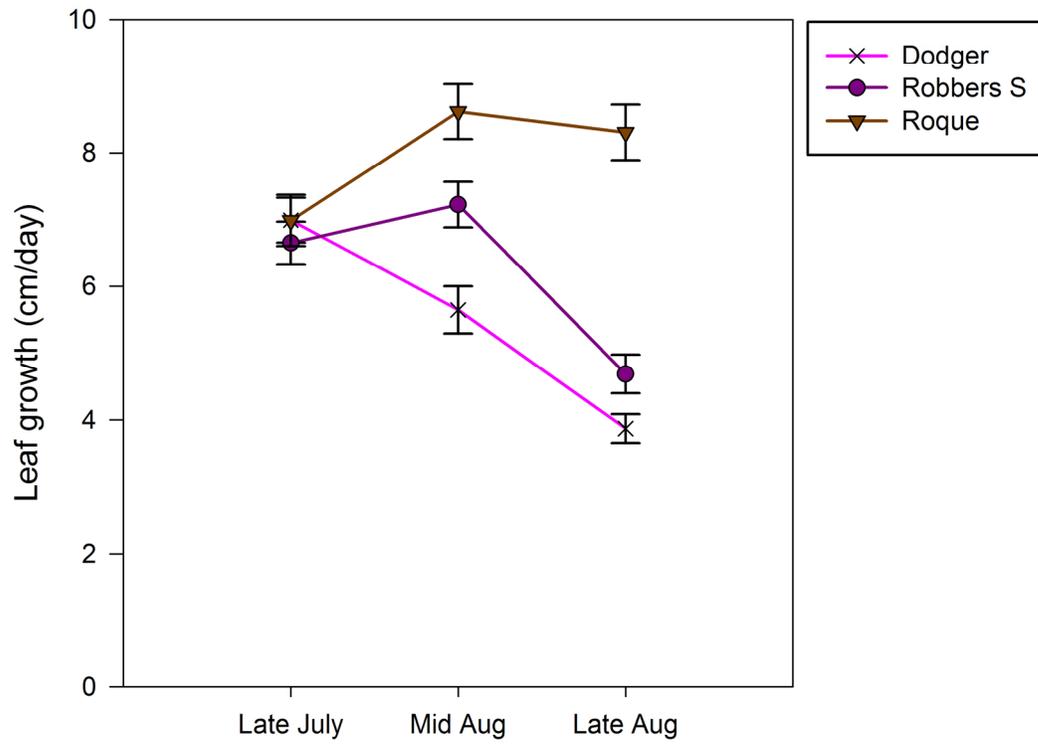


Figure 2.11. Comparison of eelgrass leaf growth in the deep zones of 3 study sites measured in late July, mid and late Aug of 2006 (cm/day). Errors bars indicate the standard error.

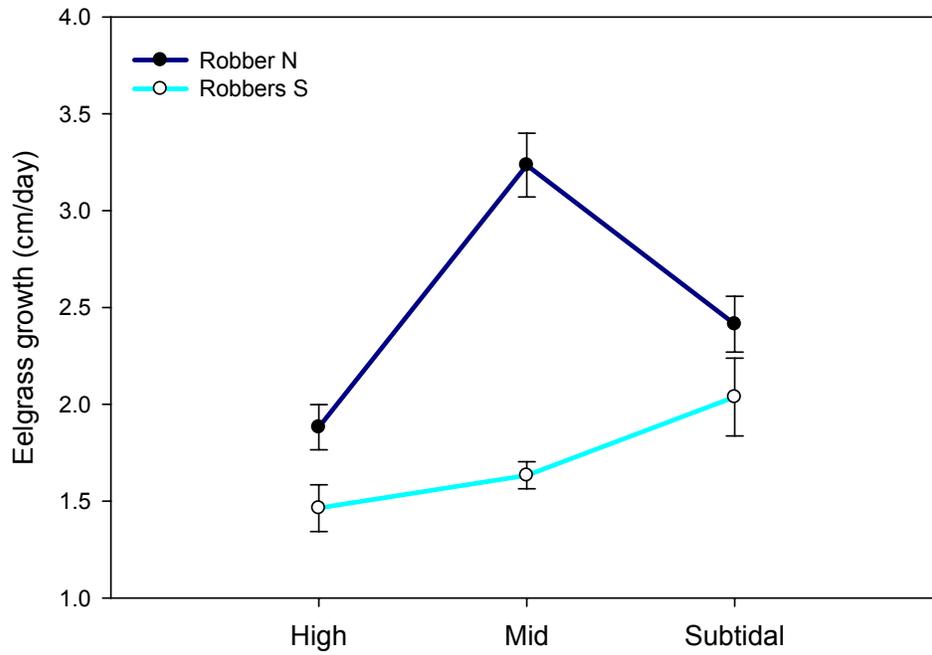


Figure 2.12. Comparison of eelgrass leaf growth in two intertidal and one subtidal zone of two sites in 2005. Errors bars indicate the standard error. (Note, the method used to collect these data was different than the data shown in Figure 2.10 and 2.11.

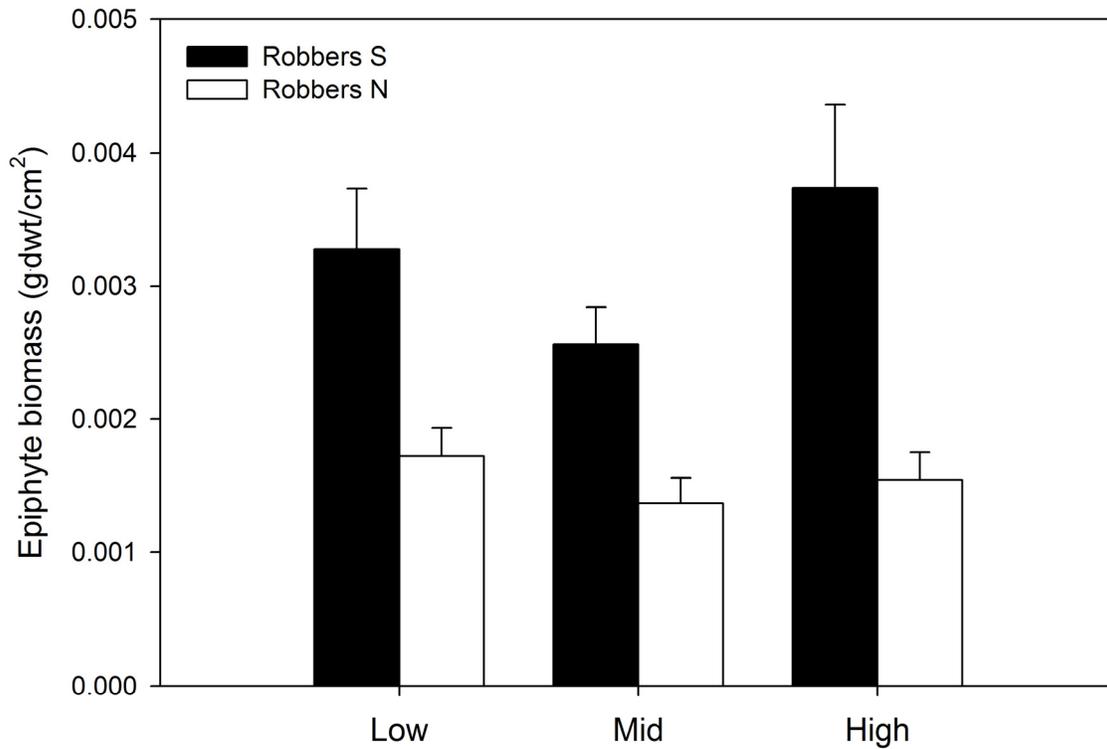


Figure 2.13. Comparison of epiphyte biomass at Robbers N and S across three intertidal heights (g·dwt/cm²) (N=10). Samples were taken at the end of August 2005. Errors bars indicate the standard error. Significant differences were found between sites at all depths, but tidal heights did not differ significantly from one another.

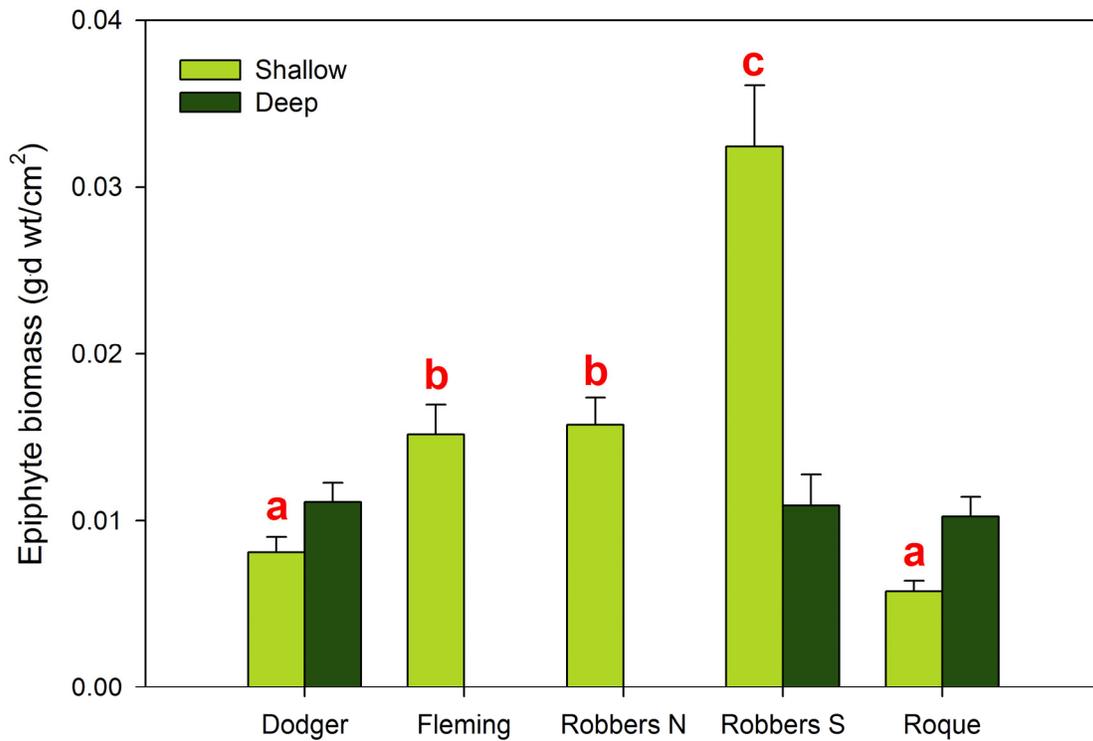


Figure 2.14. Comparison of epiphyte biomass (g·dwt/cm²) in deep and shallow subtidal zones of 5 study sites. Data shown are combined from 3 sampling periods in July and August 2006. Errors bars indicate the standard error. No significant difference was found between sites for deep zones. Identical letters indicate no significant difference between shallow zones.

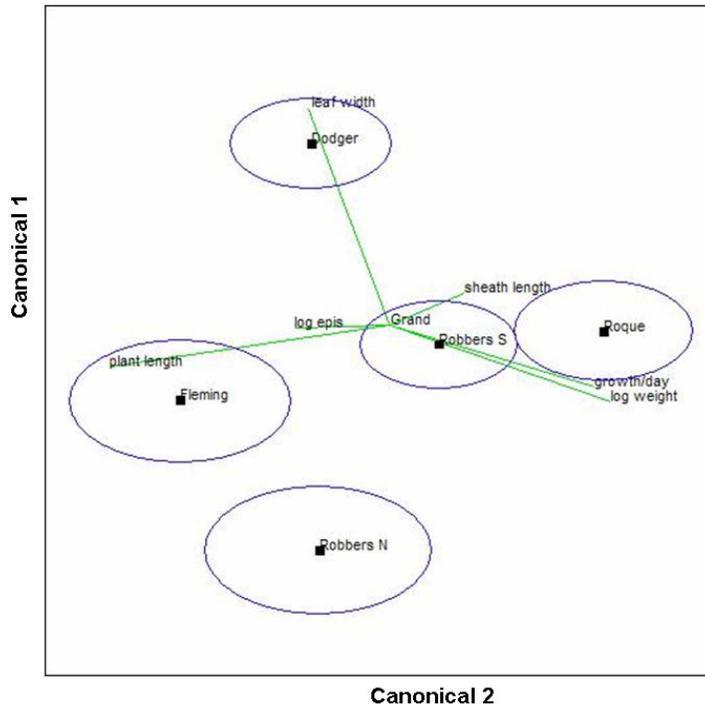


Figure 2.15. Multivariate means of sites with respect to the first 2 discriminant functions. The circles around these means indicate 95% confidence ellipses. Variable vectors are overlaid to show relations between the discriminant functions and the original variables.

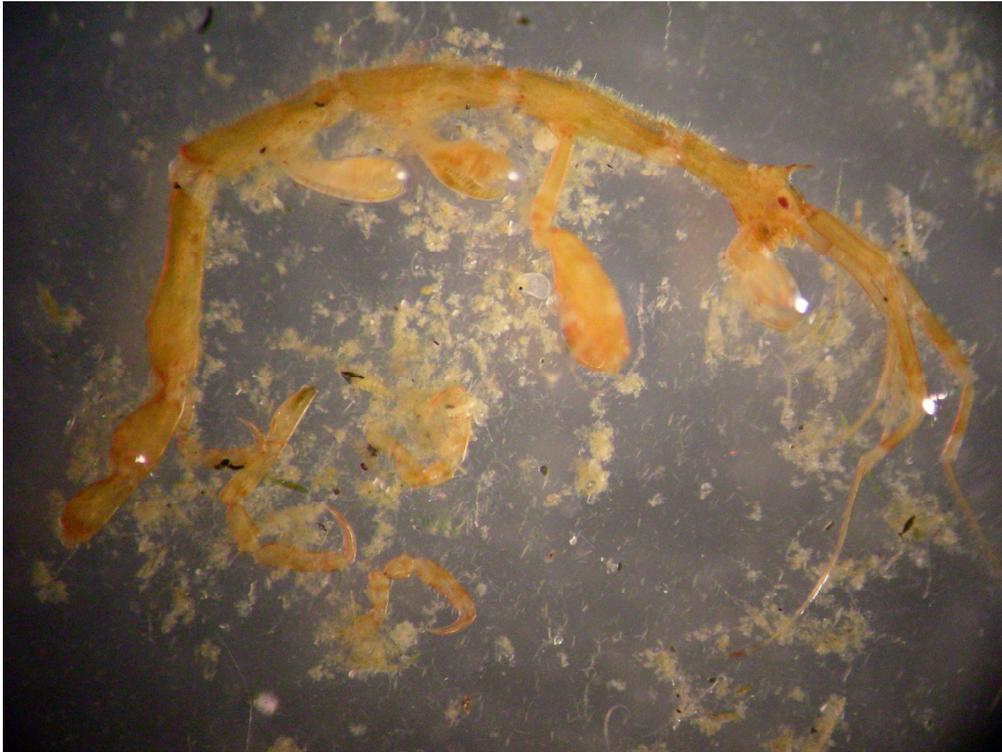


Figure 2.16. Image of a caprellid shrimp dissected from the stomach of a juvenile rockfish collected at Roque.

Chapter 3: Juvenile rockfish habitat use

3.1 INTRODUCTION

The conservation of habitats and establishment of marine protected areas is becoming an important part of fisheries management. However, the evaluation and conservation of habitats used by juveniles of economically important species, such as rockfishes, has been neglected, despite the importance of this stage in determining future stock size (take, for example, the recent establishment of rockfish conservation areas (RCAs) that target mainly adult habitats (DFO 2006)). To effectively enhance rockfish populations, juvenile as well as adult habitats will need to be conserved. Before efficient conservation can occur, however, important habitats need to be identified and evaluated for the species and life stage in question. In this chapter, the importance of eelgrass beds to juvenile rockfishes, and the variability of their abundance and habitat use, is investigated. This information will help determine whether eelgrass beds should be conserved to protect rockfish populations, which species would benefit from that protection, and when it needs to be in place. In Chapter five this selection of juvenile habitats will be improved by identifying site characteristics which increase juvenile abundance and habitat value, thereby allowing the identification of priority beds within the specific habitat type of eelgrass.

In this chapter, the species composition of juvenile rockfish in eelgrass beds within Barkley Sound on the west coast of Vancouver Island is documented, in addition to their habitat utilization (i.e. timing of residency, their size and position in the bed), and abundance.

3.1.1 Juvenile rockfish habitats

Rockfish settle into juvenile habitats following larval development in the plankton. For most species the planktonic stage lasts several months, after which they become at least somewhat associated with the seafloor. Habitats reportedly used by newly settled rockfish are varied and include: rocky reefs, soft bottoms, sponge oscula, corals and vegetated habitats such as *Alaria* sp., *Macrocystis* sp. and *Zostera marina* (Freese and Wing 2003; Carr 1991; Guido *et al.* 2002; Johnson *et al.* 2001; West *et al.* 1994). How rockfish are distributed within and between habitats is not a random event, but rather a function of larval and resource availability, as well as post-settlement mortality and redistribution (Love and Haldorson 1991). There is also considerable evidence for habitat selection by juveniles prior to, and especially after settlement (Carr 1991; Almada and Faria 2000; Marliave 1977; Laurel *et al.* 2003).

The period of rockfish life history corresponding to post-settlement is probably the least studied portion of the rockfish life cycle (Love and Haldorson 1991). This is most unfortunate because it has been suggested by many studies that the strength of the year class of many species can be determined by the abundance of recently settled juveniles (Mearns *et al.* 1980; Mason 1998; Laidig *et al.* 2007) (but see Ralston and Howard 1995). A better understanding of temporal and spatial patterns of habitat use by juvenile rockfishes would, therefore, be beneficial to fisheries management by facilitating the identification of ‘essential fish habitat’ (Levin and Stunz 2005), and improving marine reserve placement (Johnson *et al.* 2001; Carr and Reed 1993a). It would also be useful for evaluating the relative nursery value of a habitat for a particular species (Adams *et al.* 2006).

3.1.2 Eelgrass as a juvenile habitat

Eelgrass beds are considered an important juvenile habitat for many species. They may provide: food chain support, provision of refuge from predation, foraging and spawning habitat, resting sites and a transition habitat for anadromous and catadromous species (Blackmon *et al.* 2006). In fact, according to Blackmon *et al.* (2006), 75% of commercially important, and 80-90% of recreationally important, marine fish in the United States rely on estuarine habitats such as eelgrass beds at some life stage- usually the juvenile stage. Two main reasons are often cited to explain why eelgrass beds are so important to the juvenile stage of marine fishes. Firstly, eelgrass beds are highly productive, providing an abundant food supply in the form of associated epibenthos, and plant detritus (Blackmon *et al.* 2006). A study on growth of copper rockfish (*Sebastes caurinus*) for example, demonstrated greater growth rates in eelgrass habitat compared with another vegetated habitat, *Agarum* (Halderson and Richards 1987). Secondly, the physical structure that eelgrass forms may provide effective protection from predation for vulnerable newly settled juveniles (Carr and Reed 1993a). A meta-analysis of studies on the survival of juvenile fishes in seagrasses provided support for this hypothesis, as fish survival was significantly higher in seagrass habitats compared with unstructured ones (Heck *et al.* 2003).

3.1.3 Eelgrass and juvenile rockfishes

Juveniles of many rockfish species have been reported in eelgrass habitats throughout the northern hemisphere. These species include: yellowtail (*S. flavidus*), black (*S. melanops*), copper (*S. caurinus*), bocaccio (*S. paucispinus*), dusky (*S. ciliatus*), quillback (*S. maliger*), brown (*S. auriculatus*), Japanese black rockfish (*S. inermis*) and blue (*S. mystinus*) (Guido *et al.*

2002; Guido *et al.* 2004b; Guido *et al.* 2004a; Love *et al.* 2002; Robinson and Martel 2007; Mathews 1990; Halderson and Richards 1987).

Despite the presence of juveniles in the habitat however, there is a paucity of information (especially from B.C.) on the abundance, timing/duration of residency, and size of juvenile rockfish in eelgrass beds (i.e. their habitat use). The literature that is available on this topic illustrates that juvenile rockfish are abundant in some, but not necessarily all, eelgrass beds. Rockfish inhabit these beds during late spring, summer and early fall months, but the timing of recruitment and length of residency is highly variable in space and time. Factors that affect this variability include: bed location, habitat quality, weather and the timing of parturition.

In 1987, Halderson and Richards published one of the few studies on the habitat use of juvenile rockfish in B.C., and found that *S. caurinus* in the Strait of Georgia were initially abundant in kelp canopies, then moved into habitats such as *Zostera* and *Agarum* in September and October where they were found at densities of up to 110 fish/100 m². Their size in the eelgrass habitat ranged from 3.9 cm in August to 4.8 cm in October.

In Alaska, Dean *et al.* (2000) used visual censuses to survey fish communities in eelgrass beds within Prince William Sound, and found rockfish at low densities (<1/100 m²). In the same year however, Murphy *et al.* (2000) published a paper on eelgrass beds near Craig, Alaska, reporting high numbers of age-0 rockfish in seine sets during the month of September; 97% of which were found in eelgrass versus other vegetated habitats.

More work has been conducted on this topic on the western side of the Pacific Ocean. In Japan, Guido *et al.* (2002) determined that young-of-the-year (YOY) *S. inermis* inhabited eelgrass beds in high numbers from April to August, although densities were highest earlier in the season. *S. inermis* in *Zostera* beds ranged in size from 21-90 mm and were found at densities as high as 230 fish/seine haul. They also determined that their settlement date was

related to when they were born, and that fish that were born earlier stayed in eelgrass beds longer than those that were born later in the season.

3.1.4 Visual transect methods

Gear selection is an important aspect in the design of ecological studies involving measuring fish abundance (Rozas and Minello 1997). The data collected in this chapter were obtained using underwater visual transects. Similar methods have been used often in the clear waters inhabited by tropical seagrasses (Nagelkerken *et al.* 2000a; Dorenbosch *et al.* 2006; Dorenbosch *et al.* 2005; Dorenbosch *et al.* 2004), and occasionally in the colder habitats created by *Z. marina* (Mathews 1990; Dean *et al.* 2000; Halderson and Richards 1987; West *et al.* 1994). However, there are many other survey methods that would also be well suited to eelgrass beds, some of which are more commonly used, and each of these has associated positive and negative attributes. Fish communities have been successfully assessed with the following methods: video cameras, SCUBA, quadrats, drop traps, trawls, and the most common method: seine nets (Bell *et al.* 1992; Edgar and Shaw 1995; McGehee 1994; Moran *et al.* 2003; Nakamura and Sano 2004b; Uhrin and Holmquist 2003; Richards 1986).

Visual surveys, like all survey methods, have their advantages and disadvantages. The advantages of the visual survey method used in this study include:

1. The behaviour and position of the fish can be noted during the surveys
2. They are non-destructive to both fish and seagrasses
3. They are not restricted to low tide events
4. The exact area being surveyed can be calculated accurately
5. Species can be identified more accurately than with video methods
6. They can be used in multiple habitat types if habitats are to be compared
7. Divers can actively search for fish through vegetation canopies.
8. They are easily conducted in subtidal habitats

Disadvantages of visual surveys include:

1. They are time consuming and costly to conduct

2. There is potential human bias with respect to size estimations and the ability to ‘find’ fish
3. They cannot be conducted in turbid conditions
4. Highly mobile fish may be counted more than once, or missed if their flight distance is greater than the visibility of the site.
5. Comparative studies have shown that they underestimate species richness

However, these pros and cons are context dependent, and can change with the questions being asked, the fish species in question, and the habitat to be surveyed. There is no one method that will be best in all situations. The visual census method was chosen for this study because: (1) the behaviour of the rockfish species of interest was amenable to underwater observations, (2) the visibility at the sites was generally sufficient to observe fish, and (3) visual transects provide additional information (i.e. how fish are distributed at high tide, and where they are located within the bed) that would complement the extensive seine data already collected in the area by Yakimishyn (2003), and a monitoring program established by Parks Canada (Robinson and Martel 2007). The one distinct disadvantage of the visual transect however, was that similar species could not be examined closely enough for an exact identification. For this reason, *S. flavidus* and *S. melanops* were not distinguished from one another; their distinction as juveniles requires morphometric measurements that cannot be conducted underwater (Laroche and Richardson 1980). All fish in this study that resembled these species were therefore counted as one species and all referred to as *S. melanops* hereafter.

3.1.5 Objectives

The objectives of this chapter were to determine the abundance of juvenile rockfishes in eelgrass beds, how they use the habitat, and how these measurements differ between study sites. Additionally, the overall species composition in eelgrass was investigated and compared between sites. The specific questions asked were:

1. Overall fish community:

- What species were present at each site?
- Which species were the most abundant?

2. Juvenile rockfish habitat use:

- What is the species composition of juvenile rockfishes in eelgrass beds in Barkley Sound?
- When does each species recruit to eelgrass beds and how long do they remain?
- What size are juvenile rockfishes while they utilize eelgrass habitat?
- Where do rockfishes occur within each bed (intertidal, shallow or deep subtidal)?
- Does habitat use differ between study sites?

3. Juvenile rockfish abundance:

- How abundant are juvenile rockfishes at each study site?
- Does rockfish abundance differ between sites?

3.2 METHODS

3.2.1 Field methods

Fish assemblages were evaluated using underwater visual transects every two weeks throughout the study period. During each sampling period, surveys were completed at all sites within a span of ≤ 5 days. In 2005 these surveys were conducted at Robbers N and S in late July and August. In 2006, Roque, Fleming and Dodger were evaluated regularly from May-Oct, and the other two sites were evaluated only twice (in June and August). Fleming was not measured in late July due to extremely poor visibility within the bed.

Transect lines were constructed so that they floated approximately 1 m above the substrate and could be easily seen within the canopy. They were positioned parallel to shore and left for at least 15 minutes after deployment to allow fish to settle back into the disturbed

area. During the ‘settling’ period, divers conducted a training transect outside of the eelgrass bed with fake fish of known size as a means of improving the consistency of their size estimates.

In both years, fish surveys were conducted by divers who swam side by side along the transect line, moving as slowly as needed to be able to find all the fish within the survey area, and also to ensure that they did not scare away any fish. For every fish encountered within 1 m either side of the transect, the species and estimated size was recorded. The start and end time of each transect was recorded, as well as the depth. All surveys were conducted between 0900 and 1700 hours when underwater visibility was greater than 2 m.

In 2005, three transects were completed in each sampling period on both high and low tides (low tide transects were done on a maximum tidal height of 1.3 m and high tide transects were conducted on a minimum tidal height of 3 m). One transect was always located at a random position along the subtidal edge of the bed. The tidal zone of each transect was recorded.

In 2006, four transects were completed in each sampling period. They were located haphazardly within the two depth zones of the eelgrass beds (shallow and deep). At Dodger, where the bed was also very wide, it was also stratified alongshore, and transects were placed within randomly chosen strata. No effort was made to sample on both high and low tides in 2006. In this year, the start of every transect line was marked with a waypoint in a GPS unit and mapped over the bed polygons in a GIS.

3.2.2 Data analysis

All analyses in this chapter were performed using JMPIN version 4.0.4 unless otherwise noted. Species abundance was not distributed normally due to the abundance of zero data, therefore non-parametric statistical tests were used in all cases. An alpha level of 0.05 was

taken to indicate a significant effect. Unless otherwise noted, all analyses were performed using combined data from 2005 and 2006. Only fish <80 mm were considered juveniles and thus included in analyses of juvenile rockfishes (as per Laidig *et al.* 2007), except *S. paucispinus* which grow faster (Love *et al.* 2002) and were considered juveniles up to 150 mm.

3.2.2.1 Habitat utilization by juvenile rockfishes

Species composition

The species composition of juvenile rockfishes was examined graphically to determine how many species were present in each month, and if their relative abundance changed over time.

Timing of residency

In 2005, fish communities at Robbers N and S were only assessed from mid-July to mid-August, therefore questions regarding the timing of rockfish residence cannot be answered at these sites. In 2006, rockfish communities were assessed biweekly during a 5 month period beginning at the start of the settlement period, and therefore, this question can be addressed for Dodger, Fleming and Roque. The abundance of juvenile rockfish at these sites was examined graphically to determine when rockfish were first observed at each site, and how long they remained. These results were compared between the three study sites.

Size of juveniles

The size frequency distribution of *S. melanops* and *S. caurinus* at each site from May – September was examined. Only data from 2006 were used in this analysis so that recruitment pulses were not confused between years. The mean fish size was compared between sites for

each month and species using a Kruskal-Wallis test followed by pairwise comparisons on months with significant differences. These comparisons were limited to Dodger, Fleming and Roque due to minimal sampling at Robbers N and S in 2006. *S. paucispinus* were not analyzed due to their extremely low numbers during the study.

Location of juveniles

Transects were identified by their location within the bed (intertidal, shallow or deep subtidal), and the mean abundance of *S. melanops* and *S. caurinus* at each of these locations was summarized for each site. *S. paucispinus* were not included due to their extremely low abundance during the study. The abundance of rockfish in each region was compared for each study site and species using Kruskal-Wallis tests.

Influence of tidal height

A subset of transects that occurred within the subtidal region of the beds on either high (≥ 3 m) or low (≤ 1.3 m) tides was analyzed to determine whether tidal height influenced the abundance of juvenile rockfish in the subtidal region. A Kruskal-Wallis test was used to compare rockfish abundance between tidal heights at all sites. Again, *S. paucispinus* were not analyzed due to their extremely low numbers.

3.2.2.2 Juvenile rockfish abundance

The mean number of fish per transect, plus the associated standard error, was calculated for each site and species throughout the study period. One-factor Kruskal-Wallis tests were used to look for differences in abundance between study sites and sampling periods for *S. caurinus* and *S. melanops*.

3.3 RESULTS

3.3.1 Fish communities in eelgrass beds

More than 4000 fish, from 14 families and 23 species were observed on 122 transects conducted in 2005 and 2006 (Table 3.1). Black rockfish comprised 15.9% of the total fish abundance on these transects, while copper rockfish only accounted for 1.3%. Juvenile rockfish were recorded on 64/122 transects (52%) and were present at all sites. The most abundant fish species were shiner perch (*Cymatogaster aggregata*) which comprised 30.3% of the total number of fish recorded. Shiner perch were recorded on 80/122 transects, and were also present at every site.

3.3.2 Habitat utilization by juvenile rockfishes

Species composition

The relative abundance of each rockfish species from May-September is shown in Figure 3.1. A maximum of three species were present in the beds during the study. The proportion of *S. caurinus* in the fish community was initially very high (100%), but dropped to less than 10% for the remainder of the study until September when the relative abundance increased (11% in early September, and 30% in late September). *S. paucispinus* were only recorded in the beds in August and September, and were a minor constituent of the community (<2%). The percent composition of *S. melanops* ranged from 82-98% in all months other than May. Patterns of relative abundance were similar across sites, except where sample size was very low (e.g. Robbers S in August)(Table 3.2).

Timing of residency

Based on data averaged from 2005 and 2006, it is evident that *S. caurinus* recruited to the study sites slightly earlier (May) than *S. melanops* (June)(Figure 3.2). *S. caurinus* were continuously observed at low numbers throughout the study, and were still present in the beds at the end of September. *S. melanops* were present in higher numbers in the beds throughout the summer, and were also still present when the study was completed at the end of September. The numbers of both species declined at the end of September, and based on graphs of rockfish abundance at individual sites over time it is clear that this pattern is consistent between all sites (Figure 3.3 a-c). Despite the presence of ‘ready to settle’ Bocaccio near some sites at the start of the study period (Figure 4.4), none were observed within eelgrass beds until late August. They were recorded on transects at two of the study sites in August and September, however their numbers were too low to determine whether their arrival at the end of the study period indicated a trend.

Rockfish size

S. caurinus ranged in size from 3-5 cm in May, and from 4-6 cm in June (Figure 3.4). In July, fish were observed in smaller size classes, possibly indicating a pulse of new settlement. They ranged from newly settled (3 cm) to 7 cm. In August, newly settled fish were still observed; in fact fish were recorded in an even smaller size class than in July (2 cm) suggesting either another pulse of settlement, or a continuation of the July pulse. The maximum size observed, however, was only 5 cm. In September the size of *S. caurinus* observed on transects only ranged from 4 to 8 cm. The lack of fish in the smallest size classes this month indicates an end to settlement. *S. melanops* ranged in size from 3-5 cm in June, and 4-6 cm in July. In

August and September they ranged from 4-8 cm. This size frequency distribution does not indicate any pulses of new recruitment after June.

The mean size of *S. caurinus* ranged from 2 cm in May to 6.6 cm in September, and *S. melanops* ranged from 4.1 cm in May to 5.7 cm in September (Figure 3.5). The size of *S. caurinus* was more variable both within and between sites than *S. melanops*. Comparisons of juvenile rockfish size revealed significant differences between sites for *S. melanops* only. At the start of the summer, no significant differences were found, but by August and Sept, fish size did differ between sites (Aug: $X^2=16.55$, $df=2$, $p=0.0003$; Sept: $X^2=18.86$, $df=2$, $p<.0001$). Pairwise comparisons show that in August fish at Roque were significantly larger than at either Dodger or Fleming ($X^2=11.77$, $df=1$, $p=0.0006$ and $X^2=8.82$, $df=1$, $p=0.003$ respectively). In Sept, all pairwise comparisons were significant; fish at Fleming were significantly larger than at Roque ($X^2=7.00$, $df=1$, $p=.008$), and fish at Roque were significantly larger than at Dodger ($X^2=4.03$, $df=1$, $p=0.045$).

Copper rockfish were smaller than blacks at the start of the study, but by the end there was no significant difference in size between the two species ($X^2=0.70$, $df=1$, $p=0.401$).

Rockfish location within the beds

In the deep regions of the study sites, the mean number of *S. caurinus* /transect ranged from 0.11 at Robbers S to 0.94 at Dodger (Figure 3.6a). Abundance in the shallow region ranged from an average of 0.2 fish/transect at Robbers S to 0.8 fish/transect at Dodger. In the intertidal region, the average number of fish/transect was 0 at Robbers S, and 0.5 at Robbers N.

S. melanops were more abundant than *S. caurinus* in all locations (Figure 3.6b). Their average abundance in the deep region of the study sites ranged from 0.11 fish/transect at Robbers S, to 20 fish/transect at Roque. In the shallow region, *S. melanops* abundance was

lower, and ranged from 0 fish/transect at Robbers S to over 11 fish/transect at Dodger. The average abundance in the intertidal region was very low (although sampling effort was also low; N=4 at Robbers N and N=1 at Robbers S); it was 0.25 fish/transect at Robbers N and 0 fish/transect at Robbers S.

When rockfish abundance was compared between locations, there was no difference in *S. caurinus* abundance at any study site. *S. melanops*, however, was significantly more abundant in the deep area at Roque ($X^2=9.04$, $df=1$, $p=0.003$), and the shallow region of Dodger ($X^2=5.89$, $df=1$, $p=0.015$). Results from tests at each site are shown in Table 3.3.

Influence of tidal height

Fourteen transects were conducted within subtidal regions during high tides, and 18 transects were conducted during low tide events. High tide transects were performed at all sites except Roque, and low tide transects were performed at all sites except Fleming (where it was too shallow to use SCUBA during tides less than 1.3 m). Overall, more fish were present in the subtidal region of the beds during high tides (Figure 3.7). The average number of juvenile *S. melanops* observed on high tide transects was 2.57, while the mean on low tide transects was only 2.11. *S. caurinus* averaged 0.36 fish/transect at high tide, and 0.11 fish/transect at low tide. However, neither of these differences were statistically significant (*S. caurinus*: $X^2=0.71$, $df=1$, $p=0.399$; *S. melanops*: $X^2=1.75$, $df=1$, $p=0.187$).

3.3.3 Juvenile rockfish abundance

Almost 700 juvenile rockfish were observed on 122 transects in 2005 and 2006, of which approximately 640 were *S. melanops*. Figure 3.2 shows the average number of each rockfish species during each sampling event for all sites and years combined. The average

abundance of *S. caurinus* and *S. melanops* at individual sites sampled in 2006 is shown in Figure 3.3. *S. caurinus* density was low, but constant, throughout the study period, never averaging more than 2 fish/transect at any site or month. *S. melanops*, on the other hand, were observed in low numbers in June (up to 4 fish/transect at Roque), but by July and August had reached a mean of over 20 fish/transect (at Dodger). They were also more social and were observed schooling together (unlike *S. caurinus* which were always solitary and occurred in lower numbers overall). By late September, the abundance of both species had declined at all sites except Fleming where the abundance of juvenile *S. caurinus* continued to increase. The mean number of *S. caurinus* and *S. melanops*/transect in September was 3.3 and 0.67 respectively.

Comparisons of juvenile rockfish abundance between sites revealed some significant differences. The mean abundance of *S. caurinus* differed between sites when all dates were combined ($X^2=12.2$, $df=4$, $p=0.0159$), as did *S. melanops* abundance ($X^2=20.6$, $df=4$, $p=0.0004$). Table 3.4 shows where these differences occurred. A ranking of sites based on mean abundance of *S. caurinus* ordered them thus:

Dodger > Fleming > Roque > Robbers N > Robbers S (*S. caurinus*)

Abundance at Dodger was significantly greater than at Roque, and marginally greater than at Robbers N & S, while all other pairwise comparisons were not significant. Sites were ranked differently when the mean abundance of *S. melanops* was considered:

Roque > Dodger > Robbers N > Fleming > Robbers S (*S. melanops*)

Here, Robbers S was significantly lower than any other site, except Fleming, and Fleming was significantly lower than Roque. Although abundance data for the two species ranked the sites differently, Robbers S was consistently the lowest, and Dodger was always within the top two.

These site differences were more apparent in the latter half of the study period, especially for *S. melanops* which recruited slightly later. When differences between sites were tested for each sampling period, significant differences for *S. caurinus* were found in early July ($X^2=7.33$, $df=2$, $p=0.026$) and early August ($X^2=12.47$, $df=4$, $p=0.014$) (Table 3.5). Significant differences were also found for *S. melanops* in early August ($X^2=14.1$, $df=4$, $p=0.007$), while marginally significant differences were found in both late August ($X^2=9.66$, $df=4$, $p=0.047$) and late September ($X^2=6.06$, $df=2$, $p=0.048$) (Table 3.5).

3.4 DISCUSSION

3.4.1 Differences in habitat use and abundance between sites and species

Three species of rockfishes were observed on visual transects in this study. Their pattern of habitat use and abundance differed from one another, and also varied between study sites. These differences will affect factors such as location and timing of conservation efforts, as well as the targeted species if conservation efforts are to be made to protect eelgrass as a juvenile habitat for rockfishes.

S. melanops were much more abundant throughout the study than *S. caurinus*. The maximum density of *S. melanops* was 21/transect (60 m²) compared to 3.3/transect (60 m²) for *S. caurinus*, and only four *S. paucispinus* were found in total. These densities suggest that *S. melanops* would benefit most from the protection of eelgrass beds. However, densities could be quite different in the future because recruitment levels of rockfishes are highly variable in time (Love and Halderson 1991). In fact, a study on copper rockfish in the Gulf Islands in 1987 (Halderson and Richards 1987) reported densities up to 66/60 m² in eelgrass beds; this is much higher than any rockfish density in this study, suggesting that eelgrass beds could also be highly valuable for *S. caurinus*.

However, care must be taken when comparing rockfish abundance over such long time periods and geographical distances. For instance, it may be irrelevant to compare 1987 densities with recent studies because the spawning population of rockfishes has declined substantially in B.C. over the past 20 years (Yamanaka and Lacko 2001a). Also, rockfish may use eelgrass habitats more or less in different areas. The 1987 paper reported that *S. caurinus* in the Gulf Islands were only abundant in eelgrass once the fronds of annual kelp (*Nereocystis leutkeana*) had begun to decay. In Barkley Sound however, perennial kelp forests (*Macrocystis integrifolia*) exist, therefore juveniles would not need to shift from the ‘preferred’ kelp habitat, and might use eelgrass to a lesser extent than in places where the kelp is annual.

While rockfish densities from 1987 may be irrelevant today because the spawning population is so much smaller, the timing of their residency would not necessarily change. In this study, juveniles of the three observed rockfish species arrived in the eelgrass beds at different times. *S. paucispinus* were not observed on transects until the end of August, despite their availability in the plankton in May³. *S. caurinus* were already present in the beds before the start of the study (late May), while *S. melanops* were not observed at any site until the beginning of June. In the literature, *S. caurinus* are reported to settle as early as April, and *S. melanops* as early as May (Love *et al.* 2002). These reports support the findings in this study where *S. caurinus* were observed in the beds earlier than *S. melanops*. *S. paucispinus* however, reportedly settle over a longer period, beginning much earlier in the year; from February to August (Love *et al.* 2002). Despite this, no recently settled *S. paucispinus* (<4 cm) were observed at any time on transects in this study (none under 10 cm were recorded). It is possible that they did recruit to eelgrass as early as February and left before the start of this study. It is also possible that *S. paucispinus* were present in the beds during the study but were overlooked

³ Based on their presence in SMURFs at that site. See Chapter 4, and Figure 4.4 in particular, for more details

by divers due to low abundance, minimal pigmentation, and small size of the fish early in the season. Additionally, the elusive nature of the few *S. paucispinus* that were observed suggests that small fish may have been missed by divers because of an active avoidance on the part of the fish. However, the fact that no *S. paucispinus* were caught in smurfs after May suggests that they were not simply missed on transects, but rather, either left the beds early, or did not use it as an initial habitat at all. The idea that they initially inhabited eelgrass, but left early is supported by another study that noted migrations from eelgrass habitat only two weeks after settlement (Tolimieri and Levin 2005).

By the end of the September 2006, *S. melanops* and *S. caurinus* abundance had decreased in the beds (Figure 3.2). This decrease could be due to increased mortality of the juveniles, or movement out of the beds into other habitats. Other studies have reported movement of juvenile rockfish out of shallow habitats in the fall, and attributed it to factors such as decreasing temperature in shallow waters, increased water motion and storm events, and decreased eelgrass shoot density (Guido *et al.* 2002; Love and Haldorson 1991). All of these factors were observed qualitatively at the study sites in September, therefore it is probable that the decrease in abundance in September was due to emigration from the beds into deeper habitats rather than increased mortality. The abundance of *S. paucispinus* was so low that it is not possible to determine if the appearance of them in August and September was a trend towards increased abundance in the fall. Results from a student project in the area, however, revealed that *S. paucispinus*, *S. melanops*, and *S. caurinus* were caught in beach seines in nearby inlets in equal abundance between mid-October and November 30 (Palecnzy 2006) suggesting that patterns of migration from the bed were similar for all three species, despite differences in arrival times. Therefore, if eelgrass beds were to be protected as a juvenile habitat for these

species, disturbances to the fish community would only need to be avoided until the start of the fall season.

Interestingly, the variability in the size of *S. melanops* juveniles was very low throughout the study compared with *S. caurinus*, as seen in the size-frequency distribution in July through September (Figure 3.4). Rockfish are known to give birth to larvae over several months; for *S. caurinus* this happens from March-June, and for *S. melanops* from Jan-May (Love *et al.* 2002). Based on this long period of larval release we would expect constant settlement into the eelgrass beds over a period corresponding to the parturition period, and a wide range of sizes within the beds. In the case of *S. caurinus*, we saw small settlers at the start of the study, as well as in July and August (Figure 3.4), indicating that the juveniles settled over at least a 4 month period. Conversely, the size range of *S. melanops* was small throughout the study, and the mean size increased steadily, suggesting a single settlement pulse in June/early July. Differences in the number and timing of these pulses lead to interesting questions regarding the spawning population of rockfishes. It could be argued that a single settlement pulse is indicative of synchronized spawning by a single population of schooling fish, and that multiple settlement events indicate spawning from a larger population base. Determining where settling juveniles come from is beyond the scope of this study, but potentially useful to habitat conservation because it would be most beneficial to protect beds that supported juveniles from a more diverse population of adults. Further work using genetic analysis could be used to address these questions, and the knowledge gained from it would be a great addition to this study when attempting to prioritize beds for the conservation of rockfishes.

Rockfishes were observed in all beds except Fleming by early June. At Fleming, no rockfish were recorded until almost 2 months later, despite the fact that many were caught in smurfs at the site starting as early as May (Figure 4.6). Seine sets were conducted in early July

to eliminate the possibility that high shoot density was obscuring the fish and causing them to be missed by the divers. Two seine sets caught only 3 newly settled *S. caurinus* ranging in size from 1.7-2.8 cm. It is likely that the timing of the seine corresponded closely with the start of recruitment at the site because of the small size of the fish and the fact that rockfish were recorded on all subsequent visual transects.

The lack of rockfish at Fleming until late July may be explained by the presence of large numbers of juvenile lingcod (*Ophiodon elongatus*) which are voracious piscivorous predators who also use eelgrass as a juvenile habitat (Department of Fisheries and Oceans 2001). Figure 3.8 compares the abundance of *O. elongatus* with juvenile rockfishes at three sites (Dodger, Fleming and Roque). The abundance of the predator is much higher at Fleming than at the other two sites, and appears to be inversely related to the abundance of juvenile rockfish. It is likely that the presence of juvenile rockfishes was inhibited while *O. elongatus* was there; either by avoidance of the site on the part of settling rockfishes, or effective predation on the part of the lingcod. This example demonstrates that the timing of rockfish residency in a bed depends not only on parturition dates and timing of settlement, but also on other species using eelgrass as a juvenile habitat. Ultimately, this alteration of the timing probably decreased the number of juvenile rockfishes using Fleming as a juvenile habitat in 2006. Interestingly, *O. elongatus* are also of economic importance in B.C., and their populations of similar conservation concern (Department of Fisheries and Oceans 2001).

Modification of settlement patterns by processes such as predation has been documented by others. For example, Steele and Forrester (2002) found that heavy post-settlement predation on black-eyed gobies (*Coryphopterus nicholsii*) completely obscured spatial patterns of settlement and larval supply. Similarly, Levin *et al.* (1997) found that the recruitment levels of

pinfish (*Lagodon rhomboides*) were greater in areas where predators had been excluded and hypothesized that the pinfish actively chose habitats where predation pressures were lower.

It could be argued that due to the lower number of fish and delayed recruitment, Fleming had a reduced habitat value relative to the other sites. This is in agreement with data presented in Chapter 2 which suggested that Fleming was a less valuable site for juvenile rockfishes because of its small size and orientation with respect to incoming waves. Interestingly though, the fish that did reside at Fleming were significantly larger than at other sites by the end of the summer (Figure 3.5), suggesting that although few fish benefited from the site, it did have a high habitat value for the fish that were present. It was suggested earlier that the reduced number of fish at Fleming was due to the presence of many juvenile lingcod. This species also has highly variable recruitment between years (Pacific Fishery Management Council 2004), and thus it is possible that in years with low lingcod recruitment Fleming would have a high habitat value for rockfishes.

When rockfish abundance between regions of the eelgrass beds was compared, there were no significant differences at any of the sites for *S. caurinus*, but *S. melanops* were significantly more abundant in the deep area at Roque, and the shallow region at Dodger; a result which leads us to no general conclusion about which part of the bed rockfish prefer. These conflicting results likely came about for two reasons. Firstly, the maximum depth of Dodger was much deeper than Roque (Table 2.6), so the ‘deep’ region, as defined in this study, encompassed more habitat and may not be entirely comparable with Roque. Secondly, most *S. melanops* at Roque deep were counted on transects near the subtidal edge of the bed. It was noticed that the fish would actively forage and school over the sand, and then ‘duck’ into the adjacent eelgrass when approached by divers. Because eelgrass grew much deeper at Dodger, it did not have such a defined edge, but rather the shoot density declined slowly until it reached

zero, and thus did not supply the same type of refuge as the deep area at Roque. It can be concluded from this that the most valuable area of a subtidal bed won't be defined simply by its depth, but rather by a number of other factors that may include shoot density and the presence of a distinct edge. These factors and others, and their relationship with fish abundance, will be explored in more detail in Chapter 5.

Fewer fish were recorded in the intertidal region of this study compared to subtidal regions, although the comparison of abundance between tidal heights wasn't statistically significant. These results suggest that the intertidal area may be less important to rockfish than the subtidal region, but more sampling effort will be required before a definitive conclusion can be made. A study by Jenkins *et al.* (1997b) also found that fish assemblages in intertidal *Zostera* habitats in Australia were more similar to shallow unvegetated habitats than to subtidal seagrass habitats, indicating that seagrass fish species do not utilize the intertidal portion of the beds. If intertidal areas are truly less important for juvenile rockfish in Barkley Sound, it will have management implications for which beds, or which portion of the bed, conservation efforts should be concentrated on.

Further support for the idea that intertidal areas are less important for juvenile rockfish comes from comparisons of fish abundance at high and low tides. If fish were abundant in intertidal regions it is expected that they would be concentrated in subtidal regions during low tide events while the intertidal regions were inaccessible. However, no significant difference was found between high and low tide transects for either rockfish species in this study; in fact, rockfish abundance was actually higher during high tides.

A common feature of many seagrass studies is high variation in the abundance of nekton between samples (Jackson *et al.* 2001). This study was no exception; the variability of rockfish abundance between transects in a given bed was very high. Some of this variation may be

attributable to sampling effects such as low ‘catch efficiency’ of the visual transects, or the fact that transects were conducted at different locations within the bed, and at different times in the tidal cycle. It may have been possible to reduce this variation by sampling within the same zone of a bed, or consistently at the same tidal height (Rozas and Minello 1997) however, this was not feasible because one of the objectives was to compare rockfish abundance between bed locations and tidal heights. Another source of variation between transects may have been differences in catch (sighting) efficiency, particularly between observers (Edgar *et al.* 2001). This variation was minimized as much as possible in this study by conducting practice transects to standardize size estimates, and keeping one observer constant on the transects throughout the study. Regardless of these efforts, fish communities are inherently variable, and it is likely that our data would have been highly variable regardless of the methods used.

The drawback to this high variation, whether or not it was avoidable, is that it decreases the power to detect differences between sites (Rozas and Minello 1997). In this study significant differences in rockfish abundance were found between only 1/10 site comparisons for *S. caurinus*, and 4/10 site comparisons for *S. melanops* (Table 3.4). Had more differences been revealed between sites, better recommendations for prioritizing the sites could have been made. As it stands, *S. caurinus* abundance at Dodger and Fleming was significantly higher than at other sites, and *S. melanops* abundance at Roque, Dodger and Robbers N was significantly higher than at the other two sites. Since the abundance of both species was consistently and significantly higher at Dodger than at least two other sites, it may be the best choice of the five to focus conservation efforts. Robbers S had significantly fewer *S. melanops* than all other sites, and also had the lowest abundance of *S. caurinus* (although not significantly so for all site

comparisons), and would therefore be of lowest priority for protection as a juvenile rockfish habitat ⁴.

Another important consideration for evaluating the value of a particular eelgrass bed for rockfishes is the timing of the surveys. Significant differences in rockfish abundance between sites only occurred towards the end of the study period. Significant differences were found for *S. melanops* starting in August and for *S. caurinus* starting in July. Evaluating sites before August therefore, would decrease the chance of finding differences between them and would minimize the ability of researchers to prioritize beds.

3.4.2 Management implications of this research

Eelgrass beds are likely important for juvenile rockfishes because several species were found in the beds and in some cases were relatively abundant. If this is the case, then the conservation of eelgrass habitat has the potential to benefit rockfish populations. Any management recommendations made from this study, however, must be tempered with the knowledge that the data are from a small temporal and geographic scale and may not be applicable in other areas or years. In fact, it has been recommended that patterns of habitat utilization should not be compared except between long-term data sets (Blackmon *et al.* 2006). However, the timing of settlement and residency in eelgrass in this study is consistent with published values. Therefore, recommendations based on these data should be applicable in other areas and years. Similarly, patterns of abundance within a bed should be largely unaffected by differing settlement levels, and recommendations based on those data should also hold in other years and geographic areas. Finally, unless the relative supply of settlers to the

⁴ Interestingly, Robbers S was reported to have the highest abundance of fish, as well as the highest species richness, of any site evaluated by Parks Canada in 2006 (Robinson and Martel 2007). This emphasizes that we must be careful when recommending sites, to be clear about what they are recommended for, since different objectives could clearly lead to different recommendations.

sites differs from year to year, the relative habitat value of the beds evaluated in this study should be similar in future years. Based on this, the following can be recommended:

- Eelgrass beds with extensive intertidal area, and limited subtidal area, may be of limited value for rockfishes
- Protection of fish communities in eelgrass beds is most critical between June and September when they support the most juvenile rockfish
- Future evaluations of the value of beds for juvenile rockfishes should occur between August and September since it was during this period that the differences in abundance between sites were maximal.
- Of the sites evaluated in this study, Dodger had the highest overall value for rockfishes, and Robbers S the lowest⁵.

Overall species abundance however, may change over time and space and so recommendations regarding which species will benefit most, and how many juveniles can be expected to be protected in a given bed, may not be valid outside of the region of this study, or even in other years. Based on data from this study it seems that the conservation of eelgrass beds in general would benefit *S. melanops* most, and *S. caurinus* secondarily. Given the data in this study it does not seem that *S. paucispinus* relied on the eelgrass beds in this study as anything more than a transient habitat, therefore their conservation may be of little benefit to this species⁶. A general recommendation for evaluating eelgrass beds as a habitat for rockfishes is that variation in measurements of rockfish abundance within a bed should be minimized as much as possible, and replication should be sufficient, to maximize the ability to detect differences between sites.

⁵ However, it was shown that the presence of other species temporarily residing in eelgrass (in this case *O. elongatus*) may have lowered the abundance of rockfishes at Fleming in 2006, and decreased its value compared to the other sites. This is one instance of where the relative value of a bed might change between years because recruitment levels of *O. elongatus* are variable in time, and may have been higher in 2006 than average.

⁶ This may change with time or be different in other areas of BC, as *S. paucispinus* seem to be found increasingly in eelgrass beds, and other areas of B.C. have higher occurrences of them in eelgrass.

3.5 CONCLUSIONS

Many fish were observed in eelgrass on the visual surveys in this study, and many of these were juvenile rockfishes. This study successfully delineated the ‘how’, ‘when’, and ‘who’ of juvenile rockfish in eelgrass beds. Three species were observed in the beds during the summer month. *S. caurinus* and *S. melanops* were the most abundant. The timing of their settlement, size while in the eelgrass, and duration of residency was consistent with other studies on juvenile rockfish from a variety of habitats. The data suggest that rockfish may be more abundant in subtidal rather than intertidal areas- regardless of tidal height. Finally, significant differences in abundance between sites were found for both species during the late summer when rockfish density was highest.

3.6 FIGURES AND TABLES

Table 3.1. Family and species name of all fish observed in this study. Presence at each of the 5 study sites is indicated with an X.

Family	Species	Common name	Dodger	Fleming	Robbers N	Robbers S	Roque
Aulorhynchidae	<i>Aulorhynchus flavidus</i>	Tubesnout	X	X		X	X
Batrachoididae	<i>Porichthys notatus</i>	Plainfin midshipman	X	X			
Clinidae	<i>Gibbonsia</i> sp.	Kelpfish		X			X
Clupeidae	<i>Chupea harengus pallasi</i>	Pacific herring					X
Cottidae	<i>Synchirus gilli</i>	Manacled sculpin	X				X
	Cottid	Sculpin	X	X	X	X	X
Embiotocidae	<i>Brachyistius frenatus</i>	Kelp perch		X	X	X	X
	<i>Cymatogaster aggregata</i>	Shiner perch	X	X	X	X	X
	<i>Embiotoca lateralis</i>	Striper perch	X	X	X		X
	<i>Rhacochilus vacca</i>	Pile perch	X	X	X	X	X
Gasterosteidae	<i>Gasterosteus aculeatus</i>	Three-spined stickleback				X	X
Gobiidae	<i>Coryphopterus nicholsi</i>	Black-eyed goby	X	X	X	X	X
Hexagrammidae	<i>Hexagrammos decagrammus</i>	Kelp greenling	X	X	X	X	X
	<i>Hexagrammos stelleri</i>	White-spotted greenling	X	X			
	<i>Ophiodon elongatus</i>	Lingcod	X	X	X	X	X
Pholidae	<i>Pholis laeta</i>	Crescent gunnel	X	X	X	X	X
Pleuronectidae		Flatfish	X	X			X
Scorpaenidae	<i>Sebastes caurinus</i>	Copper rockfish	X	X	X	X	X
	<i>Sebastes melanops or flavidus</i>	Black or yellowtail rockfish	X	X	X		X
	<i>Sebastes paucispinus</i>	Bocaccio					X
Stichaeidae	<i>Apodichthys flavidus</i>	Penpoint gunnel	X	X	X	X	
	<i>Lumpenus sagitta</i>	Snake prickleback	X			X	
Syngnathidae	<i>Syngnathus leptorhynchus</i>	Bay pipefish	X	X		X	X

Table 3.2. Species composition of juvenile rockfishes at all sites in each month evaluated in 2006. The total number of fish recorded at each site is shown for every month. N/A indicates that the site was not sampled

Month	Species	Dodger	Fleming	Robbers N	Robbers S	Roque
May	<i>S. caurinus</i>	100%		n/a	n/a	100%
	<i>S. melanops</i>	0%		n/a	n/a	0%
	<i>S. paucispinus</i>	0%		n/a	n/a	0%
	N	3	0	n/a	n/a	1
June	<i>S. caurinus</i>	25%		40%		0%
	<i>S. melanops</i>	75%		60%		100%
	<i>S. paucispinus</i>	0%		0%		0%
	N	4	0	5	0	27
July	<i>S. caurinus</i>	10%				0%
	<i>S. melanops</i>	90%				100%
	<i>S. paucispinus</i>	0%				0%
	N	39	0	0	0	156
August	<i>S. caurinus</i>	11%	15%	0%	100%	1.5%
	<i>S. melanops</i>	89%	85%	100%	0%	97%
	<i>S. paucispinus</i>	0%	0%	0%	0%	1%
	N	160	20	18	2	133
September	<i>S. caurinus</i>	6%	24%	n/a	n/a	14%
	<i>S. melanops</i>	94%	70%	n/a	n/a	83%
	<i>S. paucispinus</i>	0%	6%	n/a	n/a	3%
	N	53	33	n/a	n/a	35

Table 3.3. Results of Kruskal-Wallis tests comparing the abundance of juvenile rockfish at different locations within the study sites. Significant results are shown in bold.

Site	<i>S. caurinus</i> - P-value	<i>S. melanops</i> - P-value	Locations compared	N
Dodger	0.499	0.015	Deep/Shallow	17/15
Fleming	N/A	N/A	Shallow	28
Robbers N	0.114	0.270	Intertidal/Shallow	4/10
Robbers S	0.840	0.720	Intertidal/Shallow/Deep	1/5/9
Roque	0.370	0.003	Deep/Shallow	15/18

Table 3.4. Pairwise comparisons of rockfish abundance between sites when sampling dates are pooled. Values indicate p-values corresponding to multiple Mann-Whitney U tests. Significant comparisons are indicated with bold type.

Comparison	<i>S. caurinus</i>	<i>S. melanops</i>
Dodger vs Fleming	0.136	0.093
Dodger vs Roque	0.008	0.417
Dodger vs Robbers N	0.022	0.280
Dodger vs Robbers S	0.033	0.001
Fleming vs Roque	0.235	0.004
Fleming vs Robbers N	0.192	0.704
Fleming vs Robbers S	0.323	0.011
Roque vs Robbers N	0.617	0.055
Roque vs Robber S	n/a	0.0001
Robbers N vs Robbers S	0.680	0.008

Table 3.5. Results from Kruskal-Wallis tests comparing the abundance of juvenile rockfishes between sites on every sampling date. Significant comparisons are shown in bold type.

	<i>S. caurinus</i>			<i>S. melanops</i>		
	X ²	df	P-value	X ²	df	P-value
Late May	1.45	2	0.480	n/a	n/a	n/a
Early June	3.17	4	0.530	7.56	4	0.109
Late June	n/a	n/a	n/a	2.01	2	0.368
Early July	7.33	2	0.026	4.91	2	0.086
Late July	2.67	3	0.446	2.60	3	0.457
Early August	12.47	4	0.014	14.1	4	0.007
Late August	3.74	4	0.440	9.66	4	0.047
Early Sept.	0.10	2	0.950	0.59	2	0.742
Late Sept.	4.27	2	0.118	6.06	2	0.048

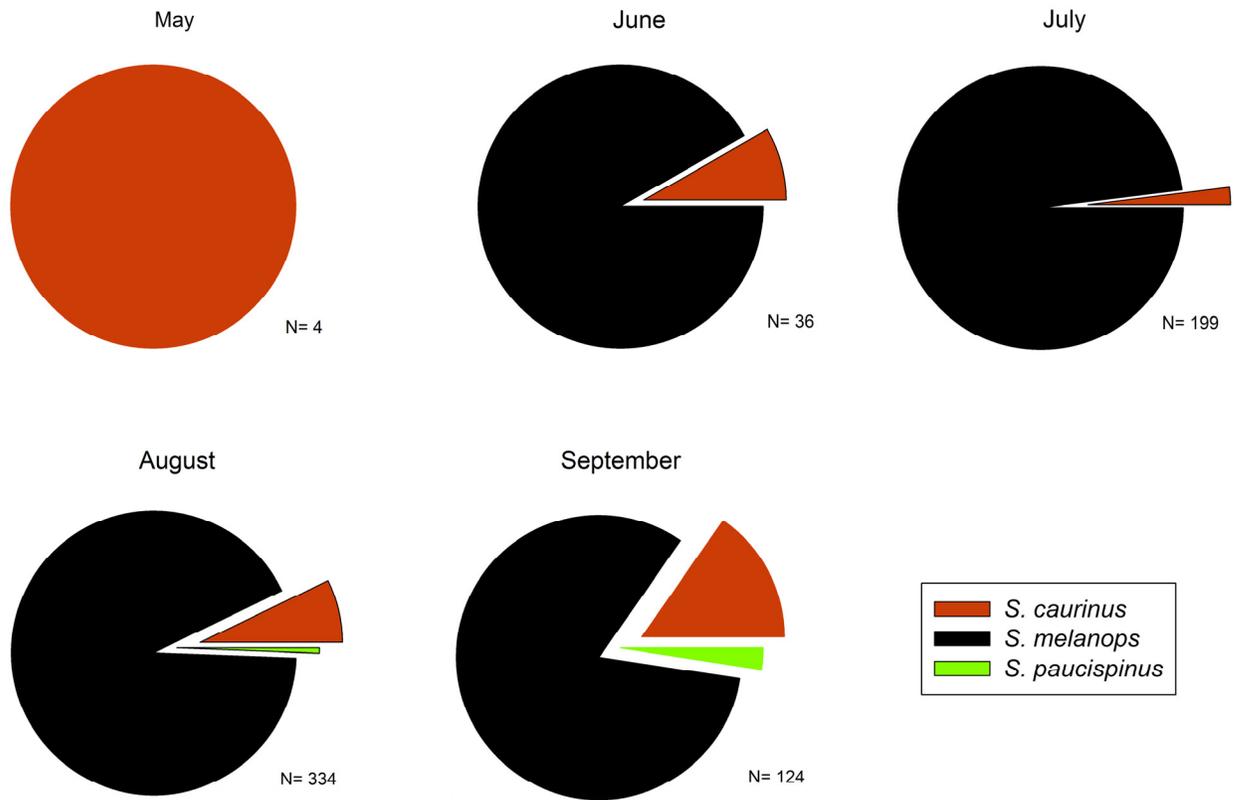


Figure 3.1. Relative abundance of each rockfish species observed on transects through eelgrass in each month of the study. Data are from all sites and years combined.

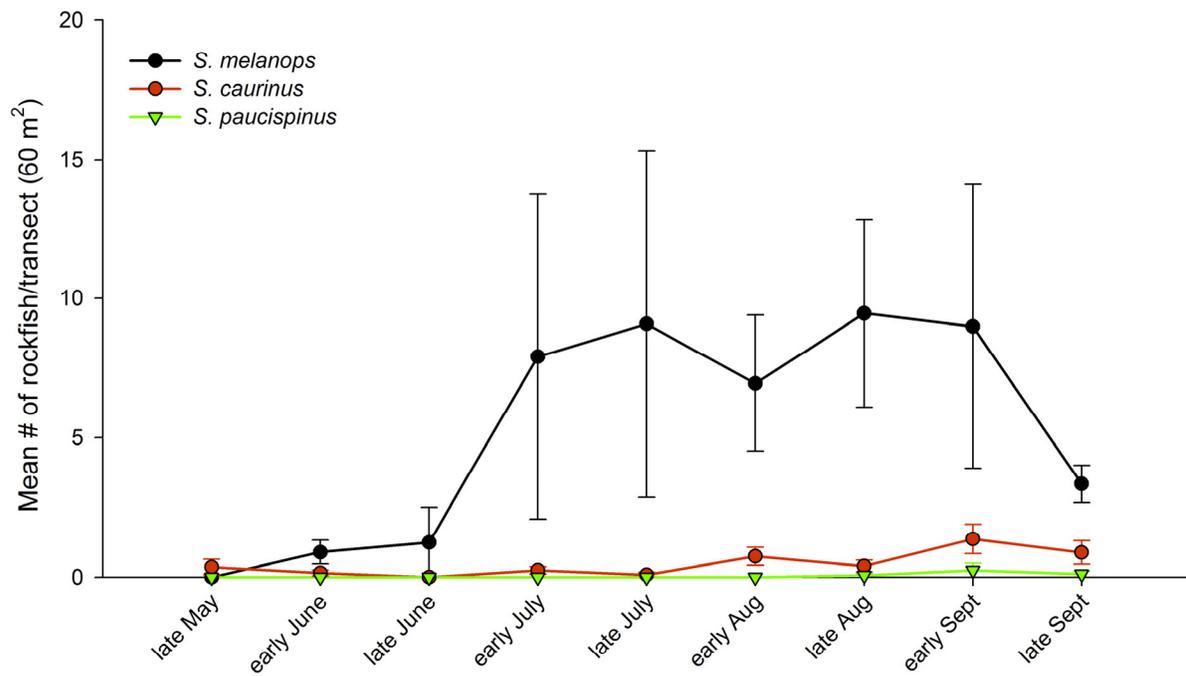


Figure 3.2. Average abundance of juveniles of three rockfish species from May – Sept. Data for this figure are from 2005 and 2006, and from all study sites combined. N=8-24 transects for each month. Bars represent standard errors.

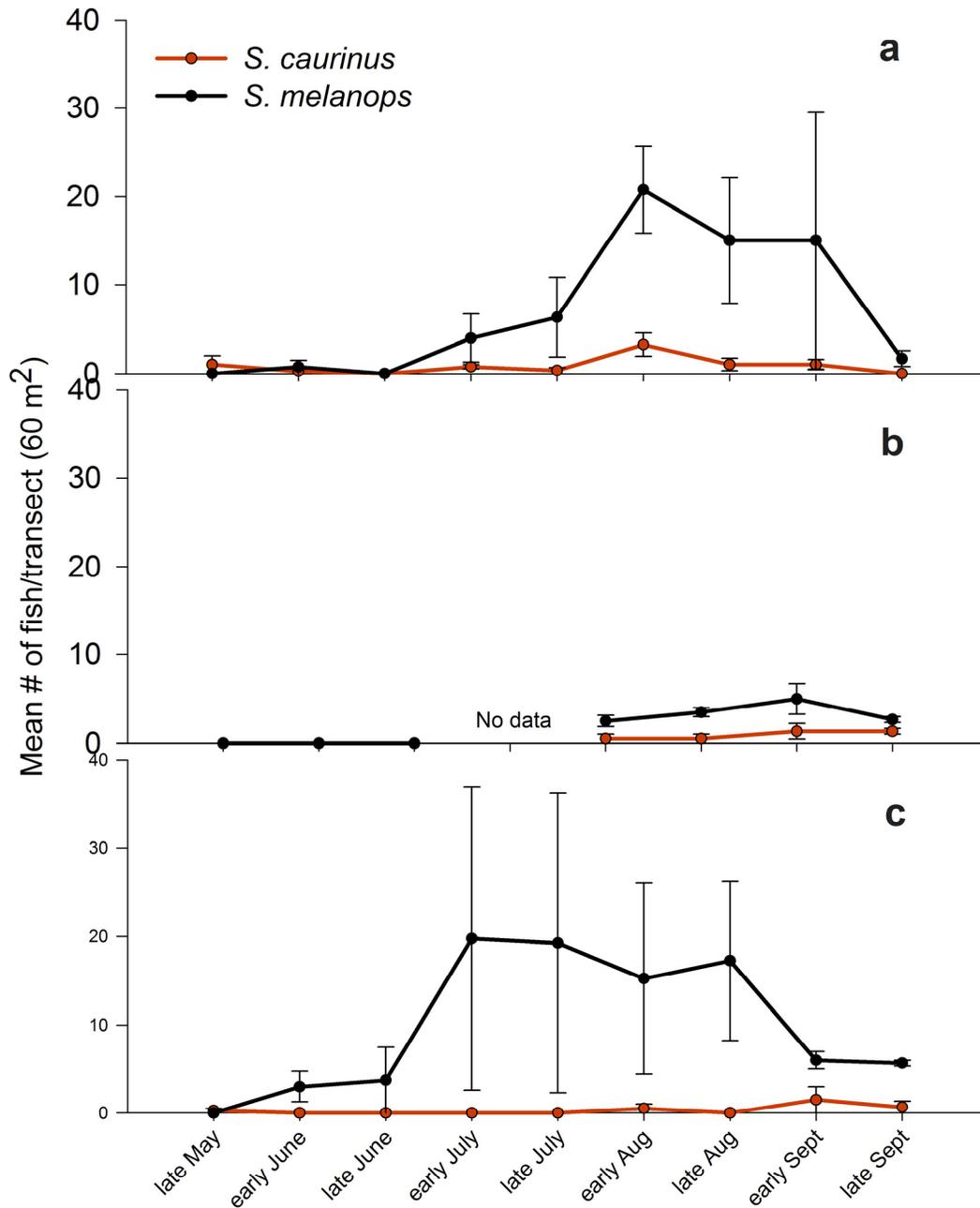


Figure 3.3. Mean number of juvenile copper (*S. caurinus*) and black (*S. melanops*) rockfish observed on transects from May-Sept 2006 at three sites (a= Dodger, b= Fleming, c= Roque). Bars represent standard errors.

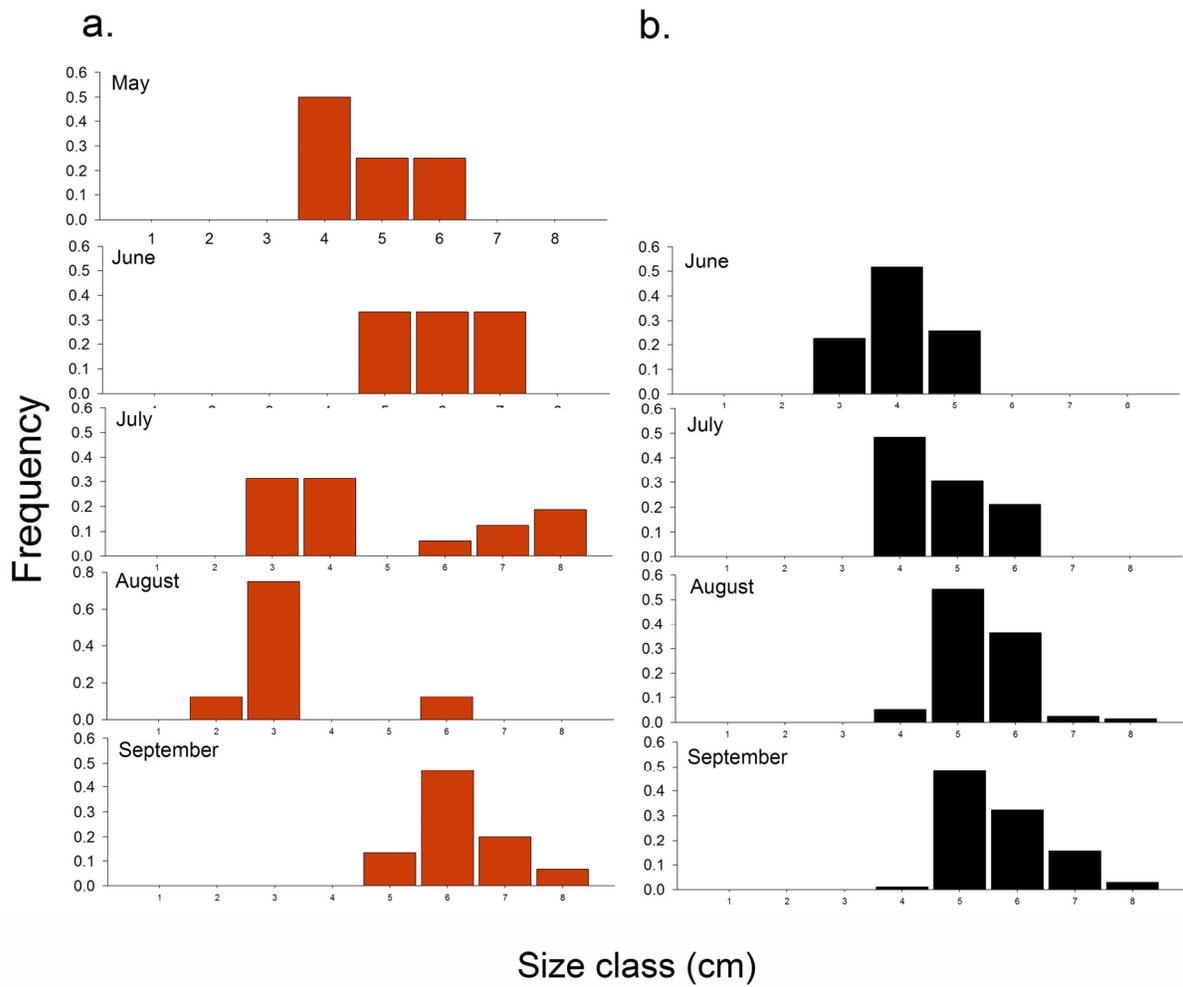


Figure 3.4. Size frequency distribution of (a) *S. caurinus* and (b) *S. melanops* observed on fish transects from May-September at all sites combined. No *S. melanops* were observed in eelgrass in May. Note changing Y axis for *S. caurinus* in August.

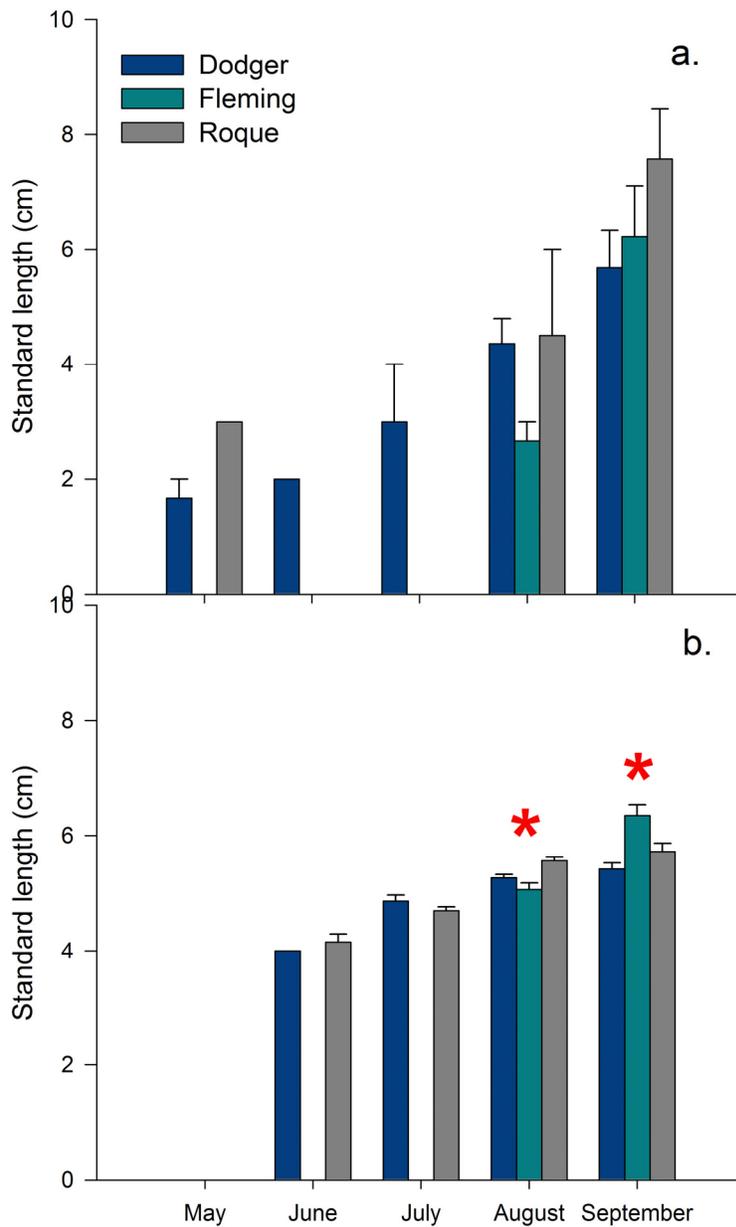


Figure 3.5. Mean size of juvenile *S. caurinus* (a) and *S. melanops* (b) rockfish observed on transects at Dodger, Fleming and Roque from May- Sept in 2006. Error bars indicate standard errors. * indicates months where significant differences were found between sites.

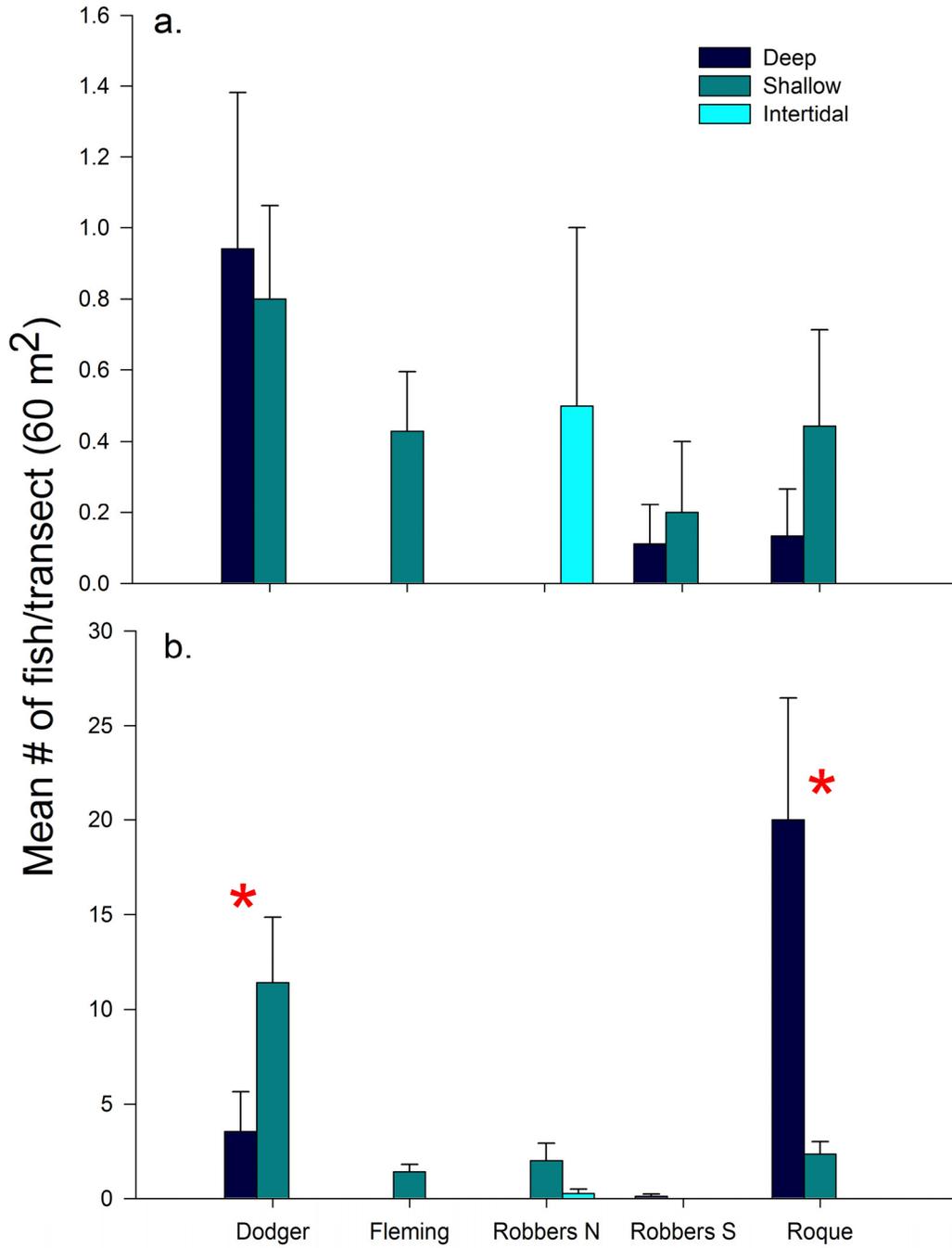


Figure 3.6 a-b. Mean abundance of a. *S. caurinus* and b. *S. melanops* in different bed locations (Intertidal, deep & shallow subtidal) at 5 study sites. Data are from 2005 & 2006. Error bars indicate standard errors. Sites with significant differences between bed locations are indicated with *.

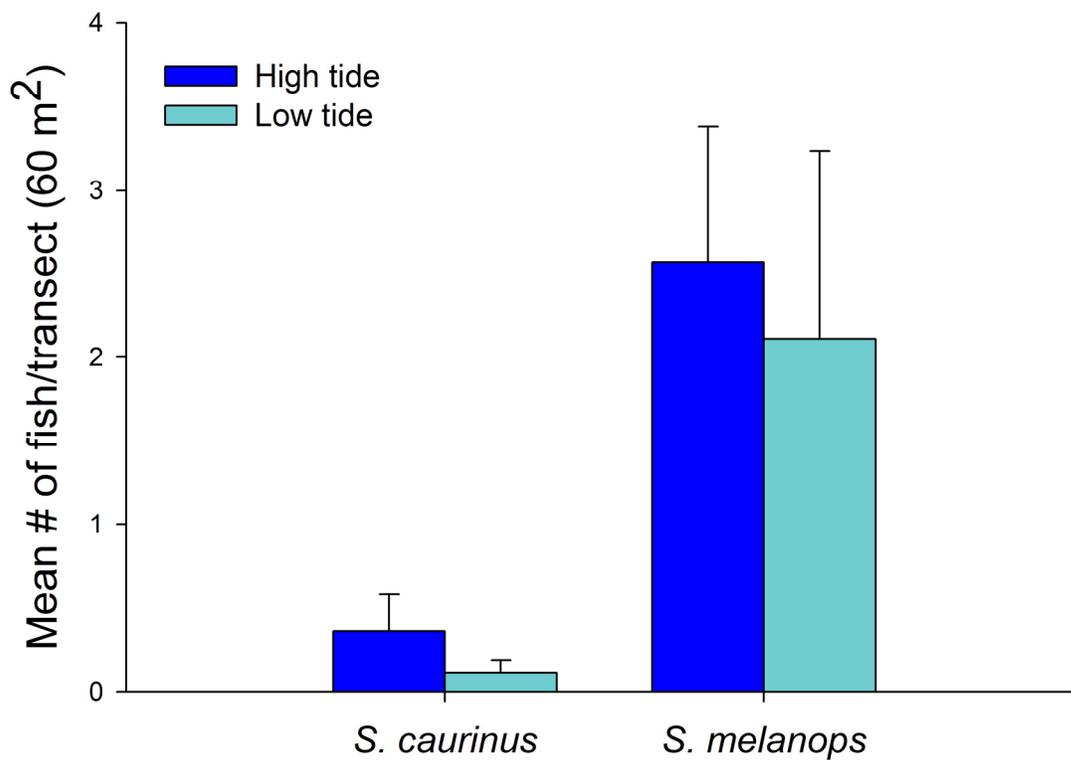


Figure 3.7. Abundance of juvenile *S. caurinus* and *S. melanops* within the subtidal region of eelgrass beds during high and low tide events. Bars indicate standard errors. No significant differences were detected between high and low tides for either species.

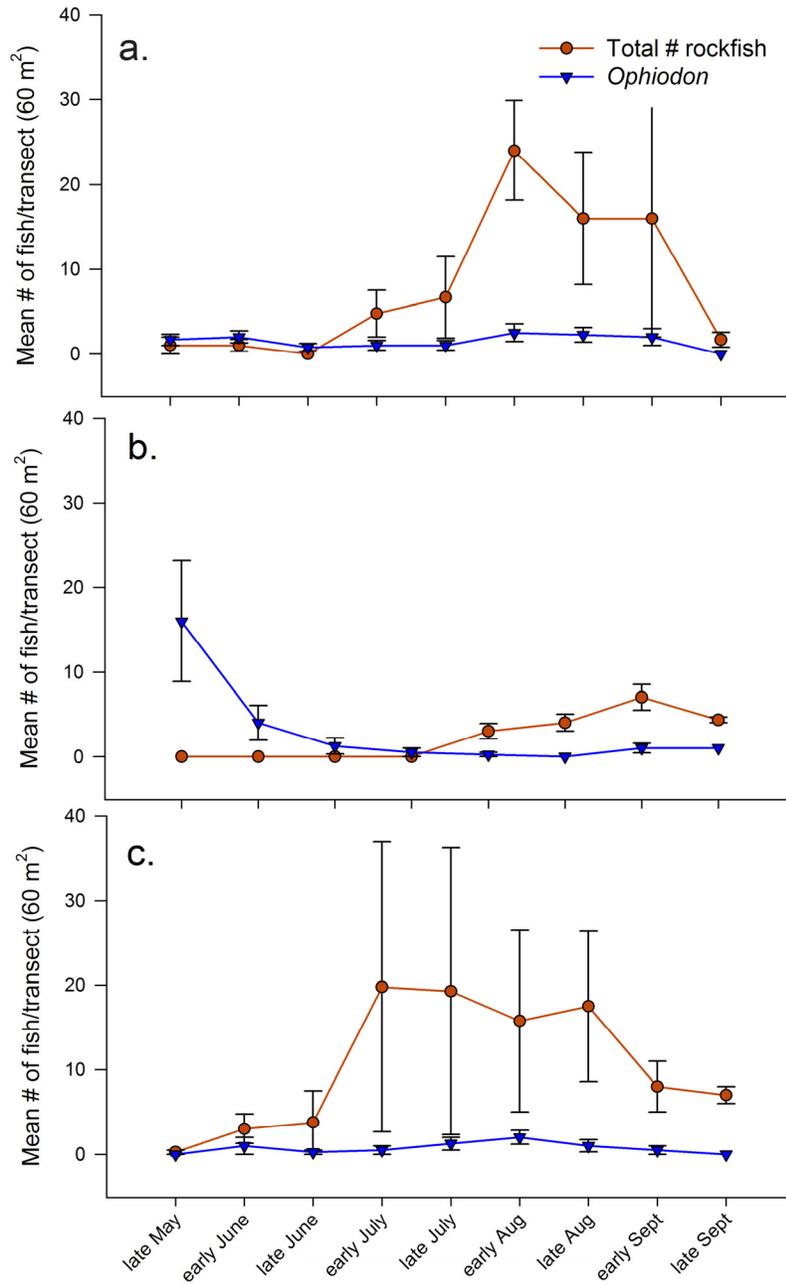


Figure 3.8. Mean number of juvenile rockfish (all species combined) and lingcod (*Ophiodon elongatus*) on transects at Dodger (a), Fleming (b) and Roque (c) in 2006. Bars indicate standard errors.

Chapter 4: Rockfish recruitment into eelgrass beds

4.1 INTRODUCTION

4.1.1 Background information

Fish settlement refers to the movement of juveniles from the plankton to a benthic environment. In rockfish (Genus: *Sebastes*) this happens after several months of development in the plankton for most species (Love and Haldorson 1991), and often corresponds to changes in body pigmentation. Copper rockfish, *Sebastes caurinus*, are known to settle from the plankton into vegetated habitats at a size of 1.8–2.0 cm (Love *et al.* 2002), while black and yellowtail rockfishes, *S. melanops* and *S. flavidus* respectively, settle at 3–4 cm (Laroche and Richardson 1980). The timing of the settlement period of rockfish varies, but for the species considered here, it usually begins in June for *S. melanops* and April or May for *S. caurinus* (Love *et al.* 2002).

Cues for inducing settlement are largely unknown for rockfishes. There is evidence that fish will settle once they reach a “competent size”, after a determined amount of time in the plankton, or when environmental conditions are favourable (Plaza *et al.* 2003). It is likely that all of these factors are important to different extents in different years, and that rockfish juveniles have multiple methods for ensuring maximal survival and settlement from the plankton.

The settlement phase can incur high mortality rates, capable of altering population structure (Watson *et al.* 2002), because it exposes fish to a new suite of predators, requires different survival techniques, and adaptation to a new food supply. Many rockfish species display thigmotactic behaviour at this stage and seek protection from

predation in highly complex habitats such as kelp and eelgrass beds (Ammann 2004). It is this behaviour which allows researchers to quantify the supply of settlers to an area with units of artificial substrate, because the fish are attracted from the plankton to the complexity of the monitoring units. SMURFs (an acronym for “Standard Monitoring Units for the Recruitment of Fishes”) were designed for this purpose (Ammann 2004). They attract pelagic juveniles that are ready to transition into a vegetated benthic habitat. SMURFs are replicable units of complex habitat that can be left continuously at study sites and sampled repeatedly without the loss of individuals. These units were evaluated by Ammann (2002), and the recommendations from his thesis were followed in this study; namely the construction design, depth of the SMURF unit, and sampling interval.

Ammann (2002) found that SMURFs were most effective at attracting species which would normally settle into vegetated habitats, such as kelp. For this reason it was thought that SMURFs would be an effective method of measuring the supply of pelagic juveniles to eelgrass beds in this study (see Ammann 2002 for a review of the advantages and disadvantages of this and other methods for evaluating larval supply, as well as a detailed description of a SMURF). The abundance of pelagic juveniles arriving at a site represents the absolute number of fish available to settle into it, prior to habitat selection or post-settlement mortality.

The evaluation of the supply of settlers to a site is an important tool for many areas of resource management. For instance, in systems where settlement levels determine the abundance of adult fish in later years (i.e. one that is recruitment limited- see definition in Armsworth 2002), documenting larval supply can lead to predictions of future stock size and structure. This information is especially important for managing exploited species

such as rockfishes. Also, knowledge of the larval supply to a site will allow sites with higher than normal levels of settlement to be identified. Such sites may be of high value for conservation because they have the potential to disproportionately increase adult populations (McNeill *et al.* 1992b). However, care must be taken in using this information because it should be coupled with data on how many of the incoming fish survive to become adults. This additional information will allow managers to avoid placing marine reserves in areas that are actually populations sinks and of little value to fisheries management (Bohnsack 1998). Finally, determining levels of settlement in conjunction with abundance of juveniles post-settlement will enable the distinction of sites that have poor habitat value versus ones that simply have low fish abundance due to low levels of settlement (Kenyon *et al.* 1999). Sites with low levels of settlement would be potential sites for stock enhancement, provided that post-settlement mortality was low.

4.1.2 Objectives

The objectives of this chapter are as follows:

- To determine the species composition of rockfish available to recruit into the five study sites
- To determine the temporal patterns of settlement at each of the five study sites
- To determine the size range of settlers for each species
- To compare the levels of settlement between study sites

4.2 METHODS

4.2.1 Field methods

The supply of juveniles to each study site was measured using SMURFs (Ammann 2002). A diagram of a SMURF with the buoy and mooring line, and pictures of the unit underwater are shown in Figure 4.1. SMURFs were deployed at the study sites no less than 15 m from the eelgrass beds and 1 m below the surface. They were positioned in such a way as to cover all incoming routes available to larval fish (to the extent possible given that their placement was restricted by minimum depth requirements, and a need to keep them away from boat traffic)(Figure 4.2).

In 2005 SMURFs were deployed at Robbers N and S at the end of July and removed on August 26. In 2006 they were deployed on May 11 at Dodger, Fleming and Roque, and were removed at the end of September. SMURFs were sampled every two weeks (see Ammann 2004 for a detailed description of methods). The SMURFs were accessed using snorkelling gear, carefully enclosed underwater in a BINKE net (Benthic Ichthyofauna Net for Coral/Kelp Environments (Anderson and Carr 1998), unclipped from the mooring line and brought to the surface where they were flushed with saltwater and shaken to remove the fish. Care was taken not to lose any fish during this process. Once emptied, the SMURF was cleaned of surface algae, and re-attached to the mooring line. All captured fish were euthanized with clove oil and held on ice in labelled plastic bags. In the lab, each fish was identified to species, weighed, and measured (standard lengths). Finally, all rockfish were photographed (except on final 2 collection dates due to time constraints), and preserved in ethanol.

4.2.2 Data analysis

All analyses in this chapter were performed using JMPIN version 4.0.4 unless otherwise noted. Abundance data were not distributed normally therefore, non-parametric statistical tests were used in all cases. An alpha level of 0.05 was taken to indicate a significant effect.

Along with newly settled rockfish, SMURFs also contained larger fish with benthic coloration, particularly later in the season. It was speculated that these were not newly settled fish, and were attracted to the units from nearby habitats such as kelp beds. To test whether this was likely, the distance from each SMURF to a kelp patch was measured using ArcMap, and regressed against both the number of fish >5 cm and the number <5 cm in each SMURF using SigmaPlot 10.0. A significant effect of distance was found for large fish ($r^2 = 0.33$, $p = 0.02$, $y = -0.02X + 2.5$), but not for small fish ($r^2 = 0.04$, $p = 0.79$, $y = 0X + 2.6$) when abundance data were log transformed, indicating that SMURFs closer to kelp beds captured more large fish. It was therefore likely that the large rockfish had migrated to the SMURFs after settling in the kelp, and did not represent newly settled fish. These large fish were thus excluded from the analyses (see Ammann 2002 for a description of similar methods). The cut off size for inclusion in the analysis was determined by examining photographs of a size progression of each species to see at what size the change from pelagic to benthic coloration occurred (Appendix I and II), and by examining the literature for published settlement sizes for each species (Laroche and Richardson 1980). The maximum standard length for classification as a new recruit was 4.5 cm for *S. melanops* and 3.0 cm for *S. caurinus*. These fish were termed 'recruit' sized fish.

Because rockfish settlement is highly variable between years (Ammann 2002; Laidig *et al.* 2007), SMURF data from 2005 and 2006 could not be combined. Therefore, each year was analyzed separately. Also, the abundance of *S. paucispinus* was too low to allow for any comparisons other than for species composition between sites.

Species composition

The species composition of rockfish in the SMURFs in 2006 was examined graphically to determine how many species were present in each month, and whether their relative abundance changed over time. The species composition for the month of August was also examined for sites measured in 2005.

Size range of settlers

The minimum and maximum size of settlers was determined for both *S. melanops* and *S. caurinus*. The size of settlers over time was examined graphically using a size frequency diagram, and differences in mean size between dates were tested for using a non-parametric Kruskal-Wallis test for each species.

Temporal patterns of settlement

The settlement of rockfish in 2005 was only evaluated in August, therefore temporal patterns could not be determined in that year. For 2006 data, the abundance of fish in the SMURFs was compared from May-September to determine patterns of abundance and recruitment pulses. This time period was chosen because it corresponded to the known duration and timing of settlement for most rockfish species. Data from all

sites were examined both separately and together. The duration of the settlement period was also compared for *S. melanops* and *S. caurinus* in 2006.

Spatial patterns of settlement

Mean number of settlers per SMURF were calculated for each species and compared between sites using non-parametric Kruskal-Wallis tests in both 2005 and 2006. Where significant differences were found, multiple Mann-Whitney U-tests were performed between all sites pairs to determine where the differences occurred. For pairwise comparisons the significance level was adjusted using the Bonferroni method (Townend 2002).

4.3 RESULTS

Species composition

Three species of ‘recruit-sized’ rockfish were caught in SMURFs throughout the study period in 2006, and only one in 2005. They included *S. melanops*, *S. caurinus*, and *S. paucispinus* (Figure 4.3). Their relative abundance in each month sampled in 2006 is shown in Figure 4.4. The presence of each rockfish species at the five sites is indicated in Table 4.1, along with 8 other species that were also caught in SMURFs. Most of these other species (with the exception of *Hexagrammos decagrammus*) were not considered to be pelagic juveniles, thus their size and abundance were not carefully documented after the first 2 sampling dates in 2006.

S. melanops made up more than 50% of the fish in SMURFs in every month except September. *S. paucispinus* were only caught in May and only made up 6% of the

total number of fish caught that month. The settlement period of this species (January-August in California)(Love *et al.* 2002) is known to start earlier than both *S. melanops* and *S. caurinus*, therefore it is possible that the peak in settlement occurred before the start of this study. *S. caurinus* comprised between 17 and 46% of the rockfish community in SMURFs between May and August, whereas in September all of the 'recruit' sized fish in SMURFs were *S. caurinus*.

Size range of settlers

Recruit sized *S. melanops* ranged in size from 3.0 to 4.5 cm and *S. caurinus* from 1.2 to 3.0 cm (Table 4.1). Despite the fact that largest individuals were omitted because they may not have been newly settled fish, the mean size of fish increased throughout the summer (Figure 4.5). This increase was significant for *S. melanops* ($X^2=73.1$, $df=3$, $p<0.0001$), but not for *S. caurinus* ($X^2=8.7$, $df=4$, $p=0.068$) when all months were compared.

Temporal patterns of settlement

In 2006, *S. melanops* recruits were abundant from late May to late June and in very low abundance thereafter (except for a small peak in early August), indicating the end of the settlement period. *S. caurinus*, on the other hand, exhibited low but steady settlement throughout the summer. There were pulses of new fish in late May, late June/early July, and a small pulse in late September (Figure 4.6). When settlement is examined at each site separately, however, it is obvious that these pulses only occurred at Fleming (Figure 4.7). The number of settling copper rockfish at the other sites was either zero, or very low during the study period. The duration of the settlement period was

calculated, however it cannot be determined absolutely because fish were already present in the plankton when the study began. Based on the data in 2006, the settlement period for *S. caurinus* lasted much longer than for *S. melanops* (*S. melanops* ended in late June, whereas *S. caurinus* lasted into September).

Spatial patterns of settlement

In August 2005, both *S. melanops* and *S. caurinus* were caught in SMURFs, however, only *S. caurinus* were small enough to be considered new recruits. There were significantly more of these recruits at Robbers N than at Robbers S ($X^2=14.3$, $df=1$, $p=0.0002$). The mean number of juvenile *S. caurinus* in the plankton was 3.1 fish/SMURF at Robbers N, and 0.25 fish/SMURF at Robbers S (Figure 4.8).

In 2006, there were also significant differences in *S. caurinus* abundance between the three sites sampled ($X^2=26.6$, $df=2$, $p=0<.0001$), but not for *S. melanops* ($X^2=1.6$, $df=2$, $p=0.4428$). *S. caurinus* were significantly more abundant at Fleming than at either Dodger or Roque ($X^2=12.9$, $df=1$, $p=0.0003$; $X^2=19.0$, $df=1$, $p=0<.0001$ respectively). Over all sampling dates, *S. melanops* abundance averaged 0.73, 0.71 and 0.35 fish/SMURF at Dodger, Fleming and Roque respectively, and *S. caurinus* abundance averaged 0.1, 0.57, 0.05 fish/SMURF at the same sites (Figure 4.9). Roque had the lowest supply of both *S. melanops* and *S. caurinus*, but the site with the highest supply differed depending on the species; Dodger had the greatest number of *S. melanops* and Fleming the greatest number of *S. caurinus*.

4.4 DISCUSSION

Over the course of this study, three species of rockfish were caught in SMURFs. These are the same three species that were observed on transects in the eelgrass beds. This is a good indication that SMURFs accurately reflect the species composition of rockfish available to recruit into eelgrass sites. However, this relationship breaks down to some degree when the species availability is analyzed on the basis of individual sites; some beds contained species that weren't collected in SMURFs (e.g. *S. melanops* at Robbers N and *S. paucispinus* at Roque), and vice versa. Also, juvenile canary (*Sebastes pinniger*) and vermilion (*Sebastes miniatus*) rockfish were observed just outside some eelgrass beds during training transects, but none were ever caught in the SMURFs. Ammann (2002) also noticed this discrepancy. For example, he collected many cabezon (*Scorpaenichthys marmoratus*) in SMURFs as pelagic juveniles, but never sighted them in nearby kelp beds; and blue rockfish (*Sebastes mystinus*) were never captured in a SMURF, but were very abundant in the kelp beds as juveniles.

The lack of *S. mystinus* in Ammann's SMURFs was attributed to the fact that they settle in rocky rather than vegetated habitats and may, therefore, not be attracted to SMURFs. The same reasoning may be applied to the lack of *S. pinniger* and *S. miniatus* in this study, especially since they were never seen in the eelgrass itself. The lack of *S. melanops* in SMURFs at Robbers N, despite their presence in the beds, was probably a result of the timing of the measurements. The supply of settlers to Robbers N was only evaluated in August, after the end of the settlement period recorded for other sites in 2006, and the fish observed in the beds had probably settled much earlier- before the SMURFs were in place. Juvenile *S. paucispinus* were sighted in some beds, but were not

caught in SMURFs at all the same sites. The number of *S. paucispinus* in the vicinity of these beds were likely underestimated due to the long time interval between sampling dates (2 weeks), as Ammann (2002) found that sampling intervals longer than 1-3 days significantly underestimated the abundance of this species.

The size of each species settling into SMURFs increased throughout the summer, however the relationship was only significant for *S. melanops*. A significant trend of increased size over the summer of 2006 was probably not found for *S. caurinus* because this species settled in multiple pulses over the summer thus lowering the mean size of the fish and increasing the variation. The fact that *S. melanops* did increase in size significantly over the summer may reflect a delay in settlement. If this were the case, it would mean that fish that settled later were not necessarily born later. Rockfish are able to remain in the plankton as ‘competent to settle’ fish for an extended period of time because of their pelagic juvenile stage. It has been suggested that such extended competent phases have evolved to increase the probability that the fish will encounter a suitable settlement site (Cowen 1991; Jenkins and May 1994). The idea that rockfish remained in the plankton to seek out a suitable habitat such as seagrass is supported in this study by the fact that the catch per unit effort (CPUE) of the SMURFs declined after the major settlement pulse, and the mean fish size increased; suggesting that there was a single pulse of reproduction followed by variable time in the plankton.

The minimum size of each species that was attracted to the SMURFs (*S. caurinus* 1.2 cm, *S. melanops* 3.0 cm) corresponds to the minimum size at settlement reported in the literature (Ammann 2002). This is a good indication that SMURFs were successful at attracting fish from the time when they are first ready to settle. It also suggests that

SMURFs did not underestimate settlement rates by failing to catch the entire size range of settlers.

Because rockfish were caught in the SMURFs on the first day of sampling, the start of the settlement period was likely earlier than late May, and its duration will be underestimated in this study. However, the fact that no *S. melanops* were found in the eelgrass beds until early June (Figure 3.3) suggests that settlement of this species had just begun and was likely not missed by much. *S. caurinus* on the other hand, were observed in the eelgrass at Dodger and Roque on the first sampling day, suggesting that *S. caurinus* began settling before the start of the study, and the settlement period of this species might be considerably underestimated. These results are consistent with what Ammann (2002) found in his SMURFs: the rockfish complex that included *S. caurinus* were caught in SMURFs as early as the end of April, and the complex including *S. melanops* and *S. flavidus* were not caught until mid May. Similarly, Love *et al.* (2002) reported that *S. caurinus* settle from the plankton as early as late April in California, and that *S. melanops* recruit at the beginning of June in Oregon (*S. flavidus*, on the other hand were reported to settle as early as April).

The timing of the peaks in settlement in this study also coincides with pulses reported in Ammann (2002); *S. melanops* and *S. flavidus* peaked from late May to late June with a smaller peak in late July. Similarly, in this study settlement of this species peaked from late May to late June, and again in early August (Figure 4.6). Ammann (2002) reported that *S. caurinus* settled in multiple large pulses, the first of which was in late June and the last in early August. In this study, *S. caurinus* also settled in multiple pulses, but these were not as pronounced as they were in Ammann (2002), and were not

consistent between sites (Figure 4.7). Love *et al.* (2002) reported that in British Columbia settlement of this species occurs primarily in July and August, whereas this study demonstrated that recruitment occurs in June and July.

These differences in the timing of peak settlement are not unexpected as there are generally large fluctuations in the timing and intensity of parturition and settlement of rockfishes between years (Moser and Boehlert 1991). Reasons for this variability include variation in oceanographic conditions (Ainley *et al.* 1993b; Ainley *et al.* 1993a), delayed fertilization by females until conditions are favourable for larvae (Wyllie-Echeverria 1987), and fluctuations in planktonic survival (Laidig *et al.* 2007). Also, most studies of rockfish settlement have been conducted south of British Columbia, and large differences may exist between warmer areas such as California and the colder waters along the coast of British Columbia.

For the month that settlement was evaluated in 2005 (August), there was a 12-fold difference in the abundance of recruits in SMURFs at Robbers N compared with Robbers S. In 2006, there were also significant differences in recruit abundance between the three study sites, but none as high as in 2005 (Figure 4.8 & Figure 4.9). Ammann (2002) also compared the abundance of rockfish recruits between sites and found significant differences in recruit abundance in SMURFs between sites for some, but not all, species. His is one of the few studies that has examined differences in abundance of settling rockfish between sites that are in similar proximity to one another as the sites in this study. Most studies that have assessed settlement patterns of rockfishes have been conducted on a much larger geographic scale (e.g. Ralston and Howard 1995; Laroche and Richardson 1980; Ainley *et al.* 1993a).

For species other than rockfishes, however, there are many examples of settlement studies in the literature on scales similar to this one. For example, McNeill *et al.* (1992b) found that one seagrass bed in Botany Bay, Australia had consistently higher levels of settlement than other beds in the area over a 3 season period, and Eckman (1987) found that settlement of scallops into eelgrass beds varied significantly between sites. In both of these studies, differences in the abundance of settling juveniles were attributed to hydrodynamics. It would be interesting therefore, to sample settlement rates at the sites in this study for multiple seasons to determine whether high settlement rates are consistent, while concurrently tracking current patterns to determine whether the rates are also determined by hydrodynamics.

Ammann (2002) found that SMURFs accurately reflected settlement rates to nearby reefs in his 2002 thesis. His SMURFs, however, were placed 100 m offshore from the kelp beds, and in 18 m of water. This distance above the seafloor, and away from nearby habitats, likely limited the number of large 'previously settled' fish that would take up residence in the SMURFs as an alternative to nearby habitats, or as a place to forage. In this study, SMURFs were located much closer than 100 m to the eelgrass beds, were often within 10 m of a kelp bed, and were positioned in water as shallow as 2 m on a low tide due to topographical and boat traffic constraints. Because of this, it was not difficult for larger fish to migrate into the SMURFs from nearby habitats or the seafloor. Indeed, large *S. caurinus* (>5 cm), juvenile lingcod (*Ophiodon elongatus*)(6-14 cm), and older young-of-the-year *S. melanops* (>5 cm) were found in SMURFs continually throughout the study period. These fish likely decreased the number of settlers in the SMURFs through either recruitment inhibition or predation, leading to an

underestimate of settlement rates. This idea is supported by the fact that Ammann (2002) found significantly more recruits in SMURFs without large predators. Similarly, others have found that juveniles of some species will avoid settling in habitats containing adults (Ohman *et al.* 1998).

The abundance of some species may also have been underestimated due to the longer sampling interval (2 weeks) in this study since Ammann found that *S. paucispinus* recruits were significantly underestimated with sampling intervals greater than 1-3 days, and *S. caurinus* were underestimated with sampling intervals less frequent than weekly. Due to these effects, it is likely that settlement rates measured with SMURFs in this study are an underestimate of true rates, and may not exactly reflect settlement rates to nearby eelgrass beds.

The problems that have been identified in this study stress the importance that SMURFs be anchored as deep and as far from other habitats as possible. The entry of larger fish may also be reduced if the ends of the outer tube were closed more securely during construction. Despite these problems, this study suggests that SMURFs were effective at identifying which species were available to the sites, as well as the duration of the settlement period, which in itself is valuable information. Also, these monitoring units were probably the best options available for measuring rockfish settlement in this study, and since there is no evidence that the supposed underestimates would not be consistent between sites, the data collected in this study should be valid for comparisons of settlement rates between sites.

In conclusion, the use of SMURFs in this study allowed the species composition of rockfish settling in eelgrass beds to be determined, the duration of their settlement

period to be resolved, and the abundance of settlers to be compared between sites. Three species of rockfish were found as settlers to the study sites; *S. melanops* was the most abundant, followed by *S. caurinus*, and finally, *S. paucispinus*. Settlement of *S. caurinus* was observed as multiple pulses beginning in May and lasting through September, while *S. melanops* were found to settle from June to August. It was speculated that the settlers of this species were from a single pulse of reproduction. The abundance of settling rockfish differed significantly between sites with the largest between site difference being between Robbers N and S in 2005 when a 12 fold difference was observed.

4.5 FIGURES & TABLES

Table 4.1. Species caught in SMURFS, their abundance, size and presence at each of the sites in 2005-2006. For rockfishes, only ‘recruit’ sized fish were included. N/A indicates that these species were not counted or measured consistently after the first sampling day. Note Robbers N & S were only sampled in 2005 from end July to end August, and Dodger, Fleming and Roque from May-Sept 2006.

Common name	Scientific name	Mean Rockfish Size (mm)					Presence at site				
		N	Size	+/-	SD	Range	Dodger	Fleming	Robbers S	Robbers N	Roque
Black rockfish	<i>Sebastes melanops</i>	109	39	+/-	4.2	30 - 45	x	x			x
Copper rockfish	<i>Sebastes caurinus</i>	83	20	+/-	2.8	12 - 30	x	x	x	x	x
Bocaccio	<i>Sebastes paucispinus</i>	2	28	+/-	2.7	26 - 30	x	x			
Lingcod	<i>Ophiodon elongatus</i>	5	92	+/-	27	69 - 139		x			x
Greenling	<i>Hexagrammos</i> sp.	3	44	+/-	8	35 - 50		x			
Sailfin sculpin	<i>Nautichthys oculoasciatus</i>	13	40	+/-	16.5	23 - 73	x	x			
Cabezon	<i>Scorpaenichthys marmoratus</i>	1	30	+/-	0		x				
Crescent gunnel	<i>Pholis laeta</i>	n/a	n/a		n/a	n/a	n/a	n/a	n/a	n/a	n/a
Penpoint gunnel	<i>Apodichthys flavidus</i>	n/a	n/a		n/a	n/a	n/a	n/a	n/a	n/a	n/a
Snailfish	<i>Liparis</i> sp.	n/a	n/a		n/a	n/a	n/a	n/a	n/a	n/a	n/a
Northern clingfish	<i>Gobiesox maendricus</i>	n/a	n/a		n/a	n/a	n/a	n/a	n/a	n/a	n/a

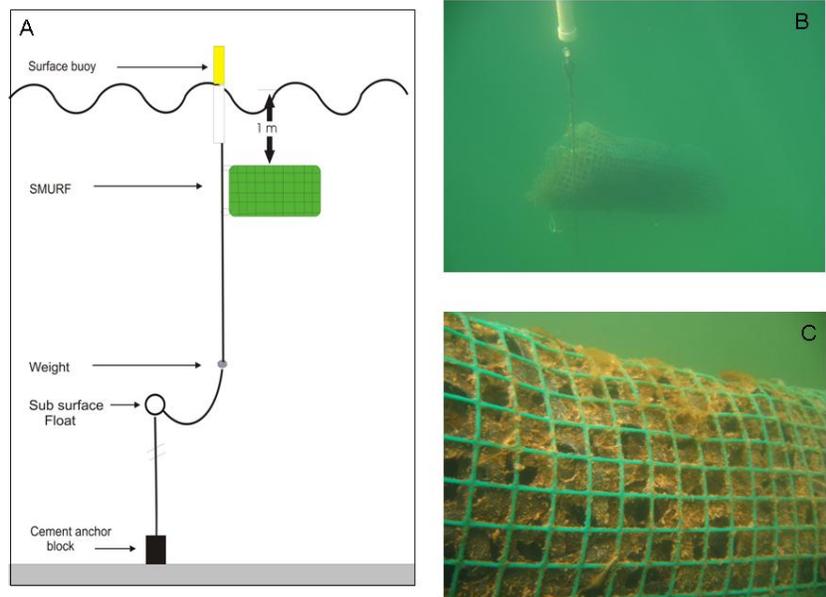


Figure 4.1. A. Schematic of a SMURF with buoy and mooring line (adapted from Ammann (2002)), B. Image of a SMURF deployed underwater, C. Close-up of a SMURF underwater.

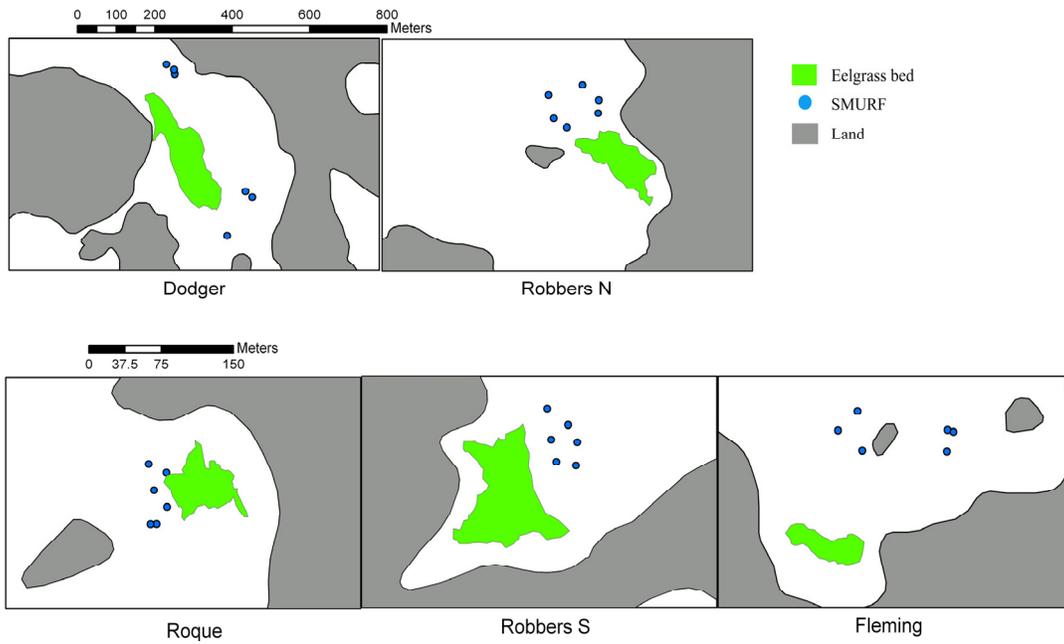


Figure 4.2. Map of the five study sites showing the proximity of SMURFs to each bed. Note, SMURFs at Dodger, Roque and Fleming were mapped using a GIS, whereas SMURFs at the other two sites were positioned approximately using hand drawn maps.

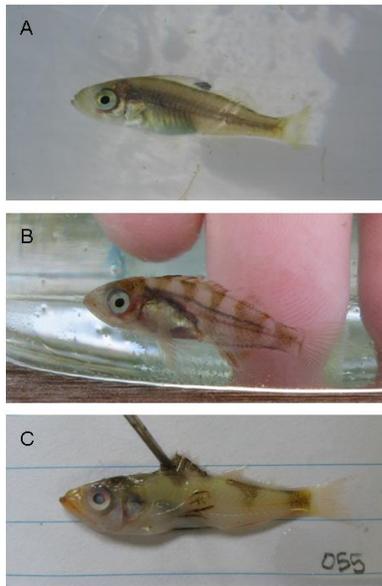


Figure 4.3. Images of juvenile rockfish caught in SMURFs. A. *Sebastes melanops*, B. *Sebastes caurinus*, C. *Sebastes paucispinus*. A-C were caught in 2006, and only B in 2005.

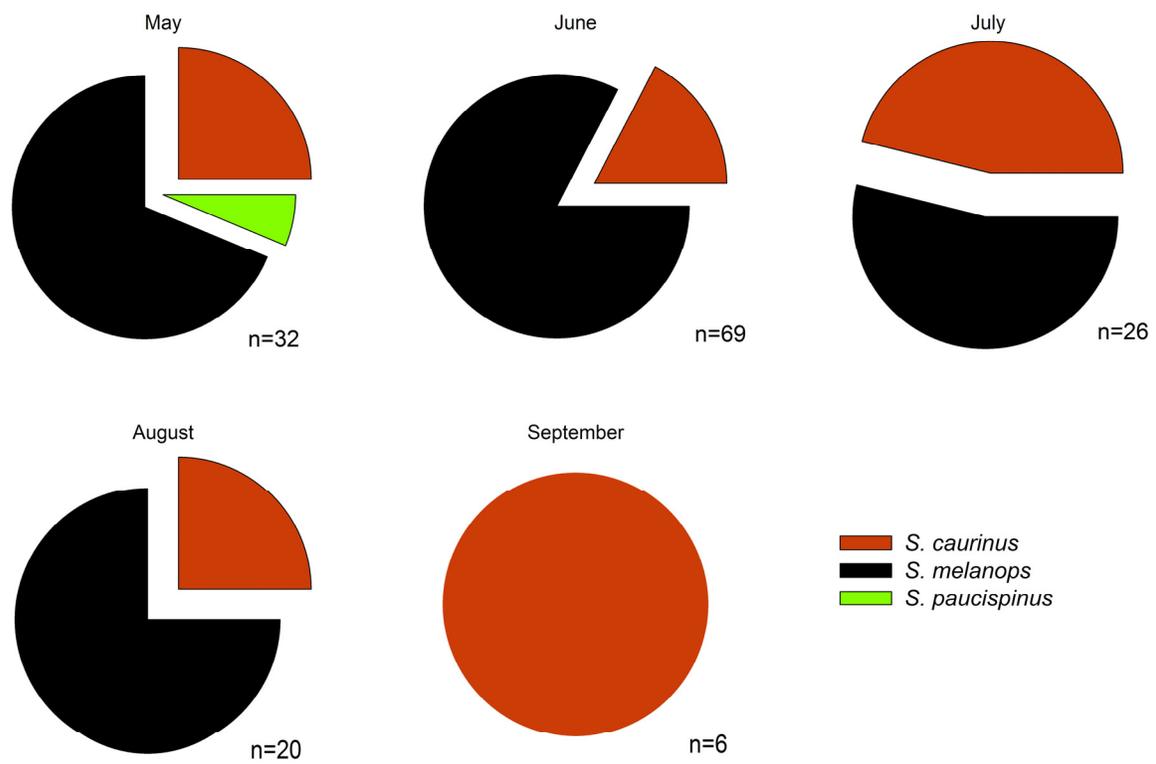


Figure 4.4. Relative abundance of newly settled rockfish of each species captured in SMURFs for each month during the 2006 field season. Data are from all sites combined.

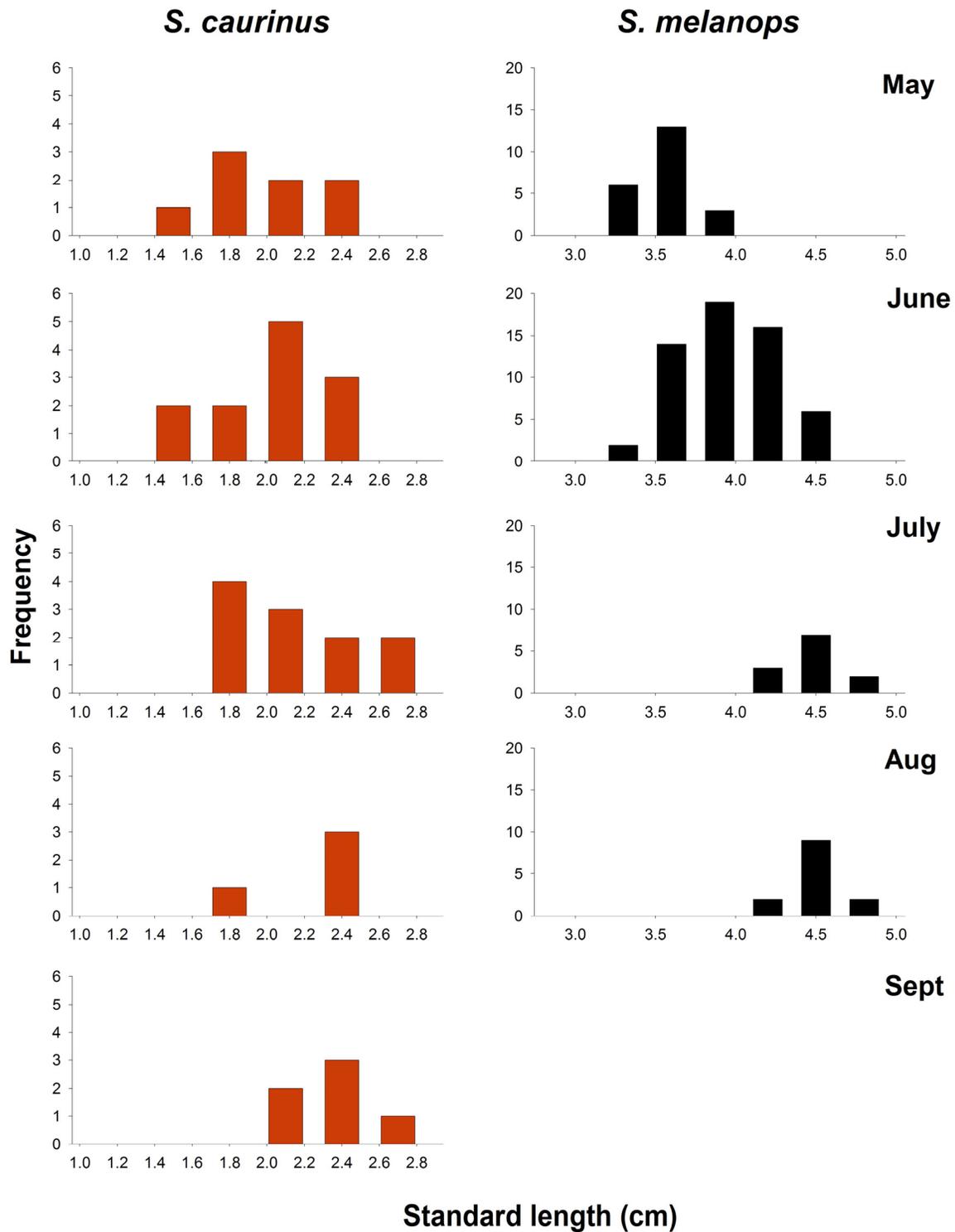


Figure 4.5. Size frequency of *S. melanops* and *S. caurinus* in SMURFs. Data are from 2006 only and include fish from Dodger, Fleming and Roque.

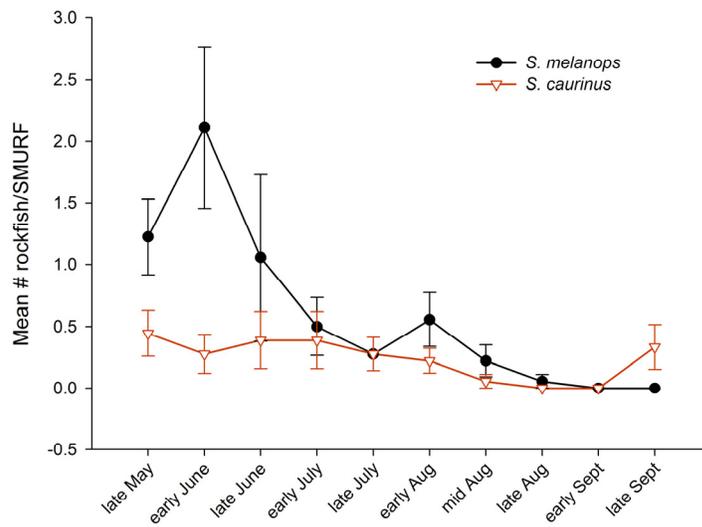


Figure 4.6. Mean number of ‘recruit’ sized rockfish captured in SMURFs at each sampling period. Data are from 2006 only and include fish from Dodger, Fleming and Roque. Bars indicate standard error of the means.

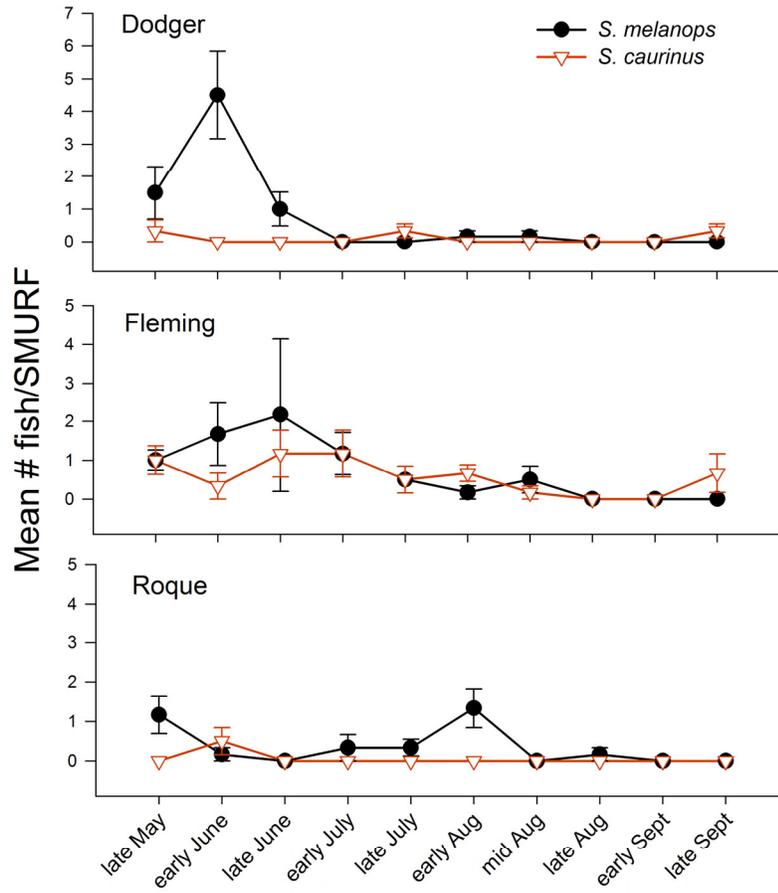


Figure 4.7. Mean number of 'recruit' sized rockfish captured in SMURFs at each sampling period for individual sites in 2006. Bars indicate standard error of the means.

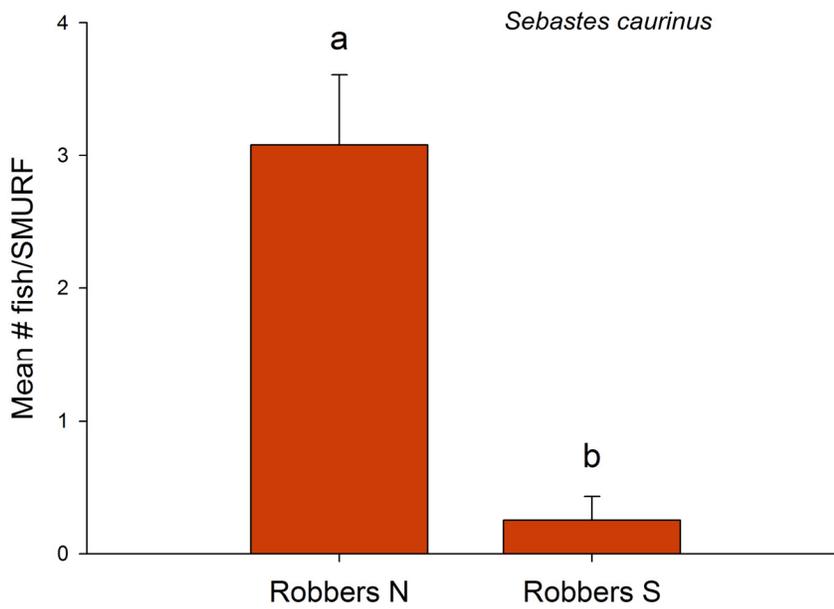


Figure 4.8. Mean number of newly settled rockfish in SMURFs at each site sampled in 2005. Data are from two sampling periods (mid and late Aug, n=12). Bars indicate standard error of the mean. Different letters indicate significant differences ($\alpha=0.05$). Data for *S. melanops* were not included because no newly settled juveniles were caught.

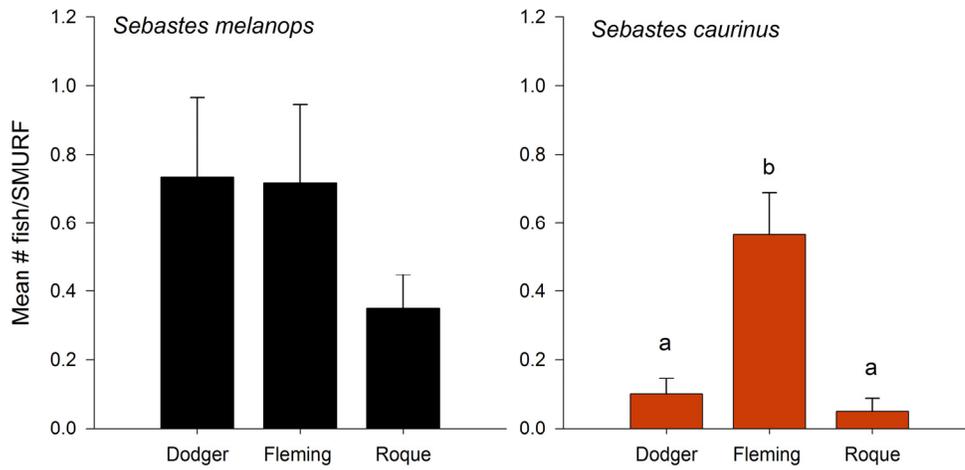


Figure 4.9. Mean number of newly settled rockfish in SMURFs at each site. Data is from all 10 sampling periods in 2006 combined. Bars indicate standard error of the mean. Different letters denote significant differences between sites ($\alpha=0.05$), and identical or no letters indicates no significant differences.

Chapter 5: Influence of larval supply and site characteristics on juvenile rockfish abundance in eelgrass beds

5.1 INTRODUCTION

5.1.1 Background information

The distribution of juvenile fish between seagrass beds is highly variable in space and time, and the factors that control it are not well understood (Jenkins *et al.* 1998). Studies that investigate these factors generally focus on two main questions: 1. what determines settlement rates to seagrass beds, and 2. are patterns of juvenile abundance controlled by pre or post-settlement processes (Armsworth 2002).

For species with a pelagic larval phase (such as rockfish) the initial abundance of juveniles in beds will be determined by larval settlement rates (Loneragen *et al.* 1998). These rates may be the result of random, indiscriminate settlement, or active habitat selection on the part of settling fish. Random settlement implies that larvae are passive particles at the whim of currents, whereas habitat selection implies an active choice on the part of the larvae (Jackson *et al.* 2001). In the first case, initial settlement patterns would depend on factors controlling larval supply such as hydrodynamics, whereas in the latter, site characteristics would be most influential. Traditionally, the first explanation has been presumed to be the case; in fact, most models consider fish larvae to be passive particles and there are certainly many studies that support this assumption (see for example: Bell *et al.* 1988; Eckman 1987; Levin 1996). A well cited paper by Bell and Westoby (1986a) has been influential in shaping this thinking in the past. These authors proposed that if settling fish discriminated between seagrass beds their survival rates

would be lower given the high mortality rates associated with the planktonic life stage. It was their belief (and that of many who have cited their papers) that fish, therefore, settle randomly into the first habitat encountered, and redistribute to more favourable microhabitats within it thereafter. Many recent studies, however, provide support for the idea that settlement from the pelagic environment is not always a random process (see Hindell *et al.* 2003). For instance, preferential selection of high value habitats by settling fish has been documented for Atlantic cod (*Gadus ogac* & *G. morhua*) (Laurel *et al.* 2003), leopard grouper (*Mycteroperca rosacea*) (Aburto-Oropeza *et al.* 2007) and a labrid, *Achoerodus viridis* (Worthington *et al.* 1992).

Overall, it is likely that both random settlement and habitat selection play a role in determining the initial distribution of juvenile fishes. The relative roles of these processes in determining settlement rates to a site may depend on the scale of investigation (Jackson *et al.* 2001). At large scales, patterns of distribution depend mainly on larval supply, which is influenced by physical processes such as hydrodynamic patterns, spawning biomass and environmental factors affecting planktonic survival (Laidig *et al.* 2007; Ralston and Howard 1995; Worthington *et al.* 1992). At smaller scales (i.e. individual beds) the abundance of newly settled fish could still be controlled by hydrodynamics (such as current direction and speed) if larvae are passive particles, or by bed characteristics (Bell and Westoby 1986b) if settlement involves habitat selection.

The initial abundance of juveniles can be further altered by factors such as competition, predation, and redistribution processes. The relative roles of pre- *versus* post-settlement processes in structuring patterns of juvenile abundance have also been a matter of debate in the literature. Some studies have shown that juvenile abundance is

directly related to larval supply. Indeed, Ammann (2004) found that the abundance of newly settled rockfish on Californian reefs was directly related to the supply of settlers to the site. Other studies, however, have demonstrated significant alteration of settlement patterns by post-settlement factors. For example, on another temperate reef Steele and Forrester (2002) found that high levels of predation on black-eye gobies (*Coryphopterus nicholsi*) following settlement completely obscured settlement patterns. However, it is likely that variability in juvenile abundance between sites stems from multiple causes that act both pre and post-settlement (Jenkins *et al.* 1997a).

Since significant differences in habitat characteristics were found between beds in this study (see Chapter 2), and the scale of investigation was small, it was hypothesized that site characteristics would influence the abundance of juveniles either through pre-settlement habitat selection, or post-settlement processes. If this were the case relationships would exist between bed characteristics and the abundance of newly settled juveniles within it. Given the large size and considerable swimming ability of settling rockfish, it was further hypothesized that habitat selection (either during settlement or afterwards) would play a role in causing these putative relationships.

To determine whether the supply of settlers was an important factor in determining the abundance of juvenile rockfish in eelgrass beds, I compared the abundance of planktonic juveniles in the vicinity of each eelgrass site (as measured by SMURFs⁷) with the number of juveniles seen in the beds. To determine whether habitat variables were related to the abundance of juveniles in an eelgrass bed I compared the

⁷ SMURF = “Standard monitoring units for the recruitment of fishes”. See Chapter 4 for a complete description of these units.

abundance of juveniles in eelgrass beds with each of the habitat variables described in Chapter 2.

Several studies have documented the occurrence of juvenile rockfishes in eelgrass beds (Dean *et al.* 2000; Halderson and Richards 1987; Murphy *et al.* 2000; Robinson and Martel 2007), and their abundance was shown to be variable between beds and geographic areas. However, these studies have not addressed the underlying reasons for the observed variability, or identified what role eelgrass bed characteristics play in determining rockfish distribution and abundance. The present study is an initial attempt to address these questions. The importance of understanding the factors that influence the distribution of juveniles cannot be understated. According to Jackson *et al.* (2001), understanding settlement processes is crucial for making effective management decisions, as different processes will lead to different priorities. For example, if fish abundance is dependent on settlement rates which are a passive process then bed location and size will be important factors in the prioritization process, whereas if settlement is an active process, smaller scale features will need to be measured in order to distinguish beds of the highest value. If the latter is the case, identifying habitat characteristics that are related to the abundance of juvenile rockfish will be important for effective habitat conservation.

5.1.2 Objectives

The objectives of this chapter were as follows:

- To determine whether the supply of pelagic juveniles is related to the abundance of juvenile rockfish in eelgrass beds.

- To determine whether measured habitat variables are related to the abundance of juvenile rockfish in eelgrass beds.

5.2 METHODS

In the following two sections assumptions of normality, equal variance and independence were tested prior to all correlation and regression analyses.

5.2.1 Comparisons of settlement levels to the abundance of fish in beds

In this chapter the abundance of settlers arriving at each site and their abundance in eelgrass beds was compared. In 2005, both SMURF sampling and eelgrass transects were conducted in August only, therefore, a major portion of the recruitment period was missed and it was not possible to evaluate recruitment patterns for this year. However, in 2006 sampling covered most of the recruitment period for *Sebastes melanops* and *S. caurinus*⁸ allowing patterns of abundance to be compared between the SMURFs and adjacent eelgrass beds for the three sites sampled that year (i.e. Dodger, Fleming and Roque). This comparison was accomplished by graphing the mean number of fish/SMURF over time with the mean number of juvenile rockfish on transects over time for each site. A Pearson correlation analysis was performed with the mean number of rockfish/SMURF (n=6) with the mean number of fish seen on transects (n=4) for the corresponding time period using JMP v 4.0.4.

⁸ Although the very beginning of the recruitment period for *S. caurinus* may have been missed

5.2.2 Comparisons between habitat variables and abundance of fish in beds

Principal components analysis was performed first, using the mean value of each habitat variable (n=11) for each site (n=5) in JMP v 4.0.4. The first principal component was then regressed against the mean number of juvenile rockfish in the eelgrass beds using SigmaPlot 10.0. Correlations between the initial variables and the first principal component revealed the variables most likely to have a strong relationship with the abundance of rockfish if the regressions proved significant.

Secondly, Pearson correlation analyses were performed on the mean number of juvenile rockfish observed in beds in the month of August versus the mean value of each habitat variable to determine if the abundance of fish was related to individual characteristics of the eelgrass beds. A list of the variables tested is shown in Table 5.1⁹. Where a graph of the relationship indicated that the relationship was not linear, a non-linear regression was fitted instead using an inverse first order function.

5.3 RESULTS

5.3.1 Correlations between juvenile supply and abundance of fish in eelgrass beds

The relationship between the abundance of fish in SMURFs (=supply of settlers) and the abundance of juvenile fish in adjacent eelgrass beds in 2006 is shown in Figure 5.1. The correlations between the two were not significant for either species or any site as shown in Table 5.2.

⁹ Methods for obtaining these data are described in Chapter 2

5.3.2 Correlations between habitat variables and abundance of fish in eelgrass beds

When a principal components analysis was performed on all eleven habitat variables the first axis accounted for 41.2% of the variation in the original data. Correlations between the variables and the first principal component axis demonstrate that PC1 is largely influenced by 5 variables; epiphytes ($r=0.76$), salinity ($r= -0.77$), temperature ($r= 0.96$), width ($r= -0.89$) and shoot density ($r= 0.74$)(Table 5.3). Although only two of these correlations were significant (temperature and epiphytes), all or some of them are likely to be influential on juvenile rockfish abundance. A regression of PC1 against juvenile rockfish abundance in the month of August was significant for *S. melanops* ($r= -0.97$, $p= 0.017$) (Figure 5.2), and marginally significant for *S. caurinus* ($r= -0.87$, $p= 0.055$) (Figure 5.3).

Graphs of the relationships between *S. caurinus* and *S. melanops* and individual habitat characteristics are shown in Figure 5.4 and Figure 5.5 respectively. The abundance of *S. caurinus* juveniles was significantly and positively correlated with leaf width ($r= 0.90$), while the abundance of *S. melanops* was significantly and negatively correlated with both epiphyte biomass ($r= -0.91$) and water temperature ($r= -0.98$) when linear relationships were tested. All significant non-linear relationships were negative, and were found between juvenile *S. caurinus* abundance and shoot density ($r= -0.89$) and water motion ($r= -0.99$), as well as juvenile *S. melanops* abundance and proximity to kelp ($r= -0.99$).

5.4 DISCUSSION

This study failed to find a relationship between the supply of settlers to a site (as measured by SMURFs), and the number of newly recruited fish in the eelgrass beds. This uncoupling of juvenile supply with the abundance in beds is not an uncommon finding for studies of the scale of individual beds. For instance, Jenkins *et al.* (1998) also failed to find a significant correlation between the abundance of Whiting (*Sillaginodes punctata*) larvae and recruitment in Australian seagrass beds. Similarly, Levin (1994) found that the abundance of Cunner (*Tautoglabrus adspersus*) settling from the plankton was unrelated to the abundance of newly recruited individuals in temperate reef algal beds.

The abundance of juveniles in an eelgrass bed can be uncoupled from larval supply for several reasons. First of all, post-settlement mortality and migration can mask initial settlement patterns (Levin 1996). Secondly, settling fish could exhibit habitat selection in favour of certain habitat characteristics and settle disproportionately into more favourable sites (Levin 1996). Finally, the lack of correlation in this study could be an artefact of the methods used if SMURFs didn't accurately reflect the supply of settlers, or if visual surveys under or over-estimated the abundance of fish in the eelgrass beds. It is not possible to definitively determine the mechanisms behind the uncoupling of settlement rates with the abundance of juveniles in this study, however, the most likely causes can be discussed.

High post-settlement mortality may have been significant at Fleming since the abundance of juvenile rockfish at this site appeared to be negatively related to the abundance of piscivorous juvenile Lingcod (*Ophiodon elongatus*). At Dodger and

Roque, post-settlement migration may have been an important factor, since these beds were in close proximity to another juvenile habitat utilized by rockfishes (i.e., kelp beds) (Table 2.7) and juveniles were seen in these surrounding habitats. It is also possible that the uncoupling at each site began prior to settlement. Given the extended planktonic period of rockfishes, and their considerable swimming ability, it is reasonable to think that pelagic juveniles would not necessarily settle passively into beds, but rather would actively select eelgrass beds with fewer predators, more favourable habitat characteristics, higher prey abundance, etc. Habitat selection at settlement has been documented for other faunal species that recruit to benthic habitats as competent swimmers (Moksnes and Heck 2006; Carr 1991; Levin *et al.* 1997), and is thought to be a major determinant of fish distributions within and between habitats (Ohman *et al.* 1998).

The methods used to estimate rockfish abundance in the plankton and eelgrass beds may also have contributed to the lack of correlation. For instance, the abundance of juveniles in eelgrass beds may have been underestimated by the visual transect method¹⁰ or, SMURFs may not have been perfectly placed to accurately measure incoming recruits at some sites due to constraints with minimum depths required for deployment. Finally, SMURFs were highly fouled at some sites which may have limited settlement in them. It is likely, however, that the contribution of these methodological factors was minor compared to the factors discussed above.

Many of the studies that have documented disconnect between settlement and recruitment have attributed it to post-settlement mortality (see Connell and Jones 1991; Stephens *et al.* 1986; Jenkins *et al.* 1998; Levin 1994; Steele and Forrester 2002; Watson *et al.* 2002). For example, Jenkins *et al.* (1998) hypothesized that many juveniles in

¹⁰ See section 3.1.4 for more details on the potential disadvantages of the visual transect method

seagrass beds were lost to wave disturbance which lowered the abundance of fish in the beds compared to larval supply. Levin (1994) also attributed the lack of correlation in his study to post-settlement mortality, but did not offer any insight into the cause of the mortality. Finally, in California, patterns of Black-eye goby (*Coryphopterus nicholsi*) settlement were completely obscured by high levels of predation at a scale of investigation similar to that of this study (Steele and Forrester 2002).

Aside from the supply of settlers, the distribution of juvenile fish within and between habitats has been known to be influenced by abiotic and biotic aspects of an eelgrass bed. Correlations between these variables and the abundance of juvenile fish would be present if juveniles preferentially settled into, or migrated to, eelgrass beds with more favourable characteristics, or if habitat characteristics influenced post-settlement survival. The relative roles of habitat selection and post-settlement predation in influencing the distribution of crabs between habitats in Florida was investigated by Moksnes and Heck (2006). They determined that habitat selection was the most important process in structuring abundance patterns, and suggested that crab larvae avoided settling in unstructured habitats that offered little refuge from predation. However, as discussed previously, many other studies have found that juvenile abundance is highly influenced by mortality after settlement. In this study correlations were found between juvenile abundance and some of the habitat variables measured, however, it was not possible to discern the cause of these correlations with the study design used.

These univariate correlations occurred between leaf width, water motion, shoot density and the abundance of juvenile *S. caurinus*, and between temperature, proximity to

kelp, epiphytes and the abundance of *S. melanops*. It is interesting that the two species did not have significant relationships with any of the same variables given their close taxonomic relationship and similar ecology. However, since *S. caurinus* and *S. melanops* settle at different sizes (Figure 4.5), have very different juvenile coloration (Figure 4.3), perhaps it shouldn't be surprising that they would associate with different characteristics of an eelgrass bed. It is also possible, however, that this study failed to find the same significant relationships for the two species simply because there were not enough sample sites (n=5) to detect all significant relationships. This idea is supported by the fact that the trends for all variables were similar for both species, and that the magnitude of the relationships were also similar (Figure 5.4 & Figure 5.5). It is also possible that some significant relationships were not detected because juvenile rockfishes associated with finer scale features of the bed than were measured in this study. The latter point is an important one as measures of habitat variables were averaged across individual beds; if fish made microscale habitat choices, bed-wide averages would fail to show these relationships.

There is also a possibility that an unmeasured variable significantly affected juvenile rockfish distribution. This is probably true for many short-term studies because of the impossibility of measuring everything, or predicting which variables will be important. The two variables that I think, in retrospect, may have been important, but did not measure, were abundance of drift macroalgae and prey. The abundance of juvenile and adult fish in seagrass beds has been related to the volume of drift macroalgae in other studies (Adams *et al.* 2004; Kulczycki *et al.* 1981), and may have been a contributing factor to the distribution of fish in this one, since it was found in abundance at some of

the sites, and often sheltered small *S. caurinus* (personal observation). The distribution and abundance of prey species has also been shown to be a significant determinant of juvenile fish abundance in seagrass beds (Jenkins and Hamer 2001). In this study juvenile pandalid shrimp recruited to Dodger in abundance in 2006 (personal observation) and may have initially been an important food source for *S. melanops* at that site.

The two bed characteristics that are most commonly reported to have a strong relationship with juvenile abundance, perhaps simply because they are easy to measure and are tested for most often, are shoot density and biomass. These variables are usually positively associated with the abundance of a species (Attrill *et al.* 2000; Gotceitas *et al.* 1997; Bell and Westoby 1986a). Most authors speculate that the underlying reason behind these relationships is increased habitat complexity leading to increased protection from predation (Connell and Jones 1991; Cocheret de la Moriniere *et al.* 2004; Orth *et al.* 1984). Contrary to these findings, shoot density was negatively correlated with the abundance of both rockfish species in this study, however this relationship was only significant for *S. caurinus*. A possible explanation for this might be that as visual predators, rockfish feed less effectively in dense vegetation, and therefore associate more with less densely vegetated beds. A similar explanation was offered by Holt *et al.* (1983) who failed to find a positive effect of *Halodule wrightii* shoot density on the abundance of juvenile Red drum (*Sciaenops ocellatus*). However, the negative relationship in this study could also be explained if rockfish were more abundant in deeper seagrass beds as shoot density was found to decline with water depth (Figure 2.8a). This is further

supported by the significant positive relationship that was also found with leaf width ($r=0.90$), since leaf width increased with depth (Figure 2.8c).

This study demonstrated a significant negative relationship between the abundance of *S. melanops* and temperature which would be expected during summer months when water temperatures are higher than average, and are a potential environmental stressor. Other studies have also shown reduced numbers of fish in eelgrass beds during increased water temperature events (Adams 1976). In Chapter 2 it was speculated that differing temperature values between beds would affect the habitat value of the bed because temperatures can drive ontogenetic migrations (Love and Haldorson 1991). A negative relationship was also found between *S. melanops* abundance in eelgrass and epiphyte biomass in this study, although the mechanism behind the relationship is not known. As discussed in Chapter 2, relationships with epiphytes are difficult to interpret unless floral and faunal epiphytes are evaluated separately, therefore comparing these results to other findings in the literature in order to speculate on the cause of the relationship did not prove useful.

A significant non-linear relationship was found between water motion and *S. melanops* abundance. Others have speculated that high wave action events remove juveniles from the seagrass beds creating a negative relationship between water motion and *juvenile* abundance (Jenkins *et al.* 1998). As was discussed in Chapter 2, however, measurements from Dodger seem too low based on experience at site. In addition, one station unit at Robbers N had very high amounts of zinc dissolution compared with all other measurements at the site. Together, the mean values for water motion at these two sites does not reflect the amount I think was actually present and leads me to question the

validity of the significant correlation found here, despite its concurrence with other findings in the literature. If I were to assign values to the sites based on my experience of how much water motion was present during my dives, the line of the graph would be approximately vertical suggesting a lack of relationship.

It was predicted earlier that adjacent kelp habitat would influence the abundance of juvenile rockfish in nearby eelgrass beds through post-settlement migration. The idea that kelp beds are a source of juvenile rockfish for eelgrass habitat is supported by the negative correlation between distance to kelp and the abundance of juvenile *S. melanops*. This relationship shows that the closer an eelgrass bed is to kelp habitat, the more *S. melanops* there are in the eelgrass. It is interesting that this relationship is non-linear because it suggests that there might be a maximum distance that juveniles will migrate between the two habitats.

In Chapter 2 leaf width was identified as the most important ‘plant-level’ variable for distinguishing the five sites by a discriminant analysis, and it was hypothesized that it would have a strong influence on the habitat value of the beds. In this chapter, leaf width was indeed found to be important as it had a significant positive correlation with *S. caurinus* and further supports the notion that increased leaf width improves the protection from predation afforded juvenile fish.

The question of whether the results of a correlative study can be extrapolated to other years, or areas, is commonly raised upon the presentation of data such as these. It is suggested that caution should be used when trying to extrapolate these data to other years since rockfish recruitment is highly variable between years (Love *et al.* 2002) and the extent to which they rely on eelgrass beds might vary similarly. It is possible that the

relationships between rockfish abundance and habitat characteristics seen here would be similar in years of high and low recruitment, but this should be tested over a longer period than was possible in this study.

In terms of extrapolating the results of this study to other areas, caution should also be exercised since rockfish are known to utilize different habitats in different geographical areas. For example, in Howe Sound, near Vancouver, juvenile *S. caurinus* are abundant in *Agarum* sp. beds, but are rarely seen in eelgrass beds (Marliave, Jeff B., Pers. comm.), whereas in the sites investigated in this study *S. caurinus* were commonly found in eelgrass beds as both juveniles and adults (see also conflicting results in Dean *et al.* 2000; Murphy *et al.* 2000). It is suggested that a cautious approach would allow the findings of this study to be applied in other high energy, high salinity, eelgrass beds in Barkley Sound (outside of inlets and river influences), and perhaps similar habitats on the west coast of Vancouver Island. Further work on the general applicability of these results to other areas and years needs to be done with longer term studies, larger sample sizes, and in other areas of the Pacific Northwest before they could safely be extrapolated any farther.

Since some variables measured in this study had very low correlations with rockfish abundance, future studies could confidently focus on the other, more influential, ones. The variables that could be omitted in future studies include abundance of settlers, eelgrass growth rate, canopy height, biomass and bed size. Since many of these variables were time-consuming to measure, and in the case of juvenile supply involved sacrificing fish for identification, the elimination of them in future studies will allow an equivalent

amount of effort to be invested in studying more sites, and eliminate the need to euthanize fish.

5.5 CONCLUSIONS

Other researchers have emphasized the importance of larval supply and settlement in determining patterns of juvenile distribution (reviewed in Jackson *et al.* 2001). However, at the scale investigated in this study there was no significant effect of settler abundance on the abundance of juveniles in eelgrass beds, but rather significant relationships were found with multiple habitat variables. These results suggest that either settlement was a passive event and post-settlement effects strongly influenced patterns of abundance, or that the patterns arose as a result of active habitat selection by settling or redistributing fish. This study has determined that: 1) SMURFS are not a useful means for determining the abundance of juvenile rockfish in an eelgrass bed; and 2) future studies on a similar scale should focus on interactions between habitat characteristics and fish abundance. The variables which should be the focus of future studies are those that were most correlated with PC1, because together they had a significant correlation with *S. melanops* and a marginally significant one with *S. caurinus*. These variables include: temperature, water motion, salinity, epiphyte biomass, proximity to kelp, shoot density and leaf width.

Because habitat use was variable between beds (as seen in Chapter 3), and seems to be related to habitat characteristics, effective protection of juvenile habitat will not simply be the case of protecting convenient eelgrass beds but rather, management plans

will need to account for these variations in habitat use and make use of correlations such as those presented in this chapter (Jenkins and Hamer 2001).

5.5.1 Questions for future research

The following is a list of questions that were born of this chapter and merit further research:

- What factors lead to the lack of relationship between settlement and recruitment?
- Are the relationships with habitat variables consistent across years and other areas?
- Are some of the other strong, yet not significant, relationships with habitat variables significant if more sites are sampled?
- Is the abundance of juvenile rockfish related to either prey abundance, or volume of drift algae?
- Are correlations the result of habitat selection at or after settlement, or due to differential survival?

More work should also be done to relate fish abundance to habitat variables on a smaller scale rather than with overall bed values given the small scale of the fish and the evidence in the literature that they may associate with smaller scale features than are typically measured (Connell and Jones 1991). Studies that address these questions will hopefully lead to an awareness of the importance of protecting juvenile habitats, and eventually the inclusion of them in management plans for rockfishes. If this occurs, the information gained from them will allow for the most valuable beds to be identified and conserved.

5.6 FIGURES AND TABLES

Table 5.1. Pearson correlations between habitat variables and abundance of juvenile rockfishes. Bold indicates values that are significant at $p=0.05$. * indicates that a non-linear equation was fitted.

	<i>S. caurinus</i>		<i>S. melanops</i>	
	Correlation (r)	P-value	Correlation (r)	P-value
Salinity	0.62	0.26	0.80	0.10
Growth	-0.53	0.36	0.12	0.85
Epiphyte biomass	-0.45	0.44	-0.91	0.03
Bed size	-0.64	0.24	0.31	0.61
Temperature	-0.75	0.14	-0.98	0.003
Biomass	-0.44	0.46	-0.06	0.92
Kelp proximity	-0.61*	0.27	-0.99*	0.01
Shoot density	-0.89*	0.045	-0.63	0.26
Canopy height	0.10	0.87	-0.12	0.85
Water motion	-0.99*	0.01	-0.62*	0.27
Leaf width	0.90	0.04	0.73	0.16

Table 5.2. Pearson correlations between larval supply and abundance of juveniles in eelgrass beds and their significance values for each site sampled with SMURFs in 2006. No relationships were significant at $\alpha = 0.05$.

	Dodger		Fleming		Roque	
	Correlation (r)	P- value	Correlation (r)	P- value	Correlation (r)	P- value
<i>S. caurinus</i>	-0.17	0.71	-0.60	0.21	-0.33	0.47
<i>S. melanops</i>	-0.37	0.41	-0.77	0.07	-0.32	0.49

Table 5.3. Pearson correlations between the first principal component axis and the original habitat variables. Significant correlations are indicated with bold type.

Habitat variable	Correlation with PC1 (r)	P-value
Salinity	0.78	0.12
Growth	0.11	0.86
Epiphyte biomass	0.76	0.14
Bed size	-0.49	0.40
Temperature	0.96	0.01
Biomass	0.32	0.60
Kelp proximity	0.58	0.30
Shoot density	0.74	0.15
Canopy height	0.14	0.83
Water motion	0.61	0.27
Leaf width	-0.89	0.04

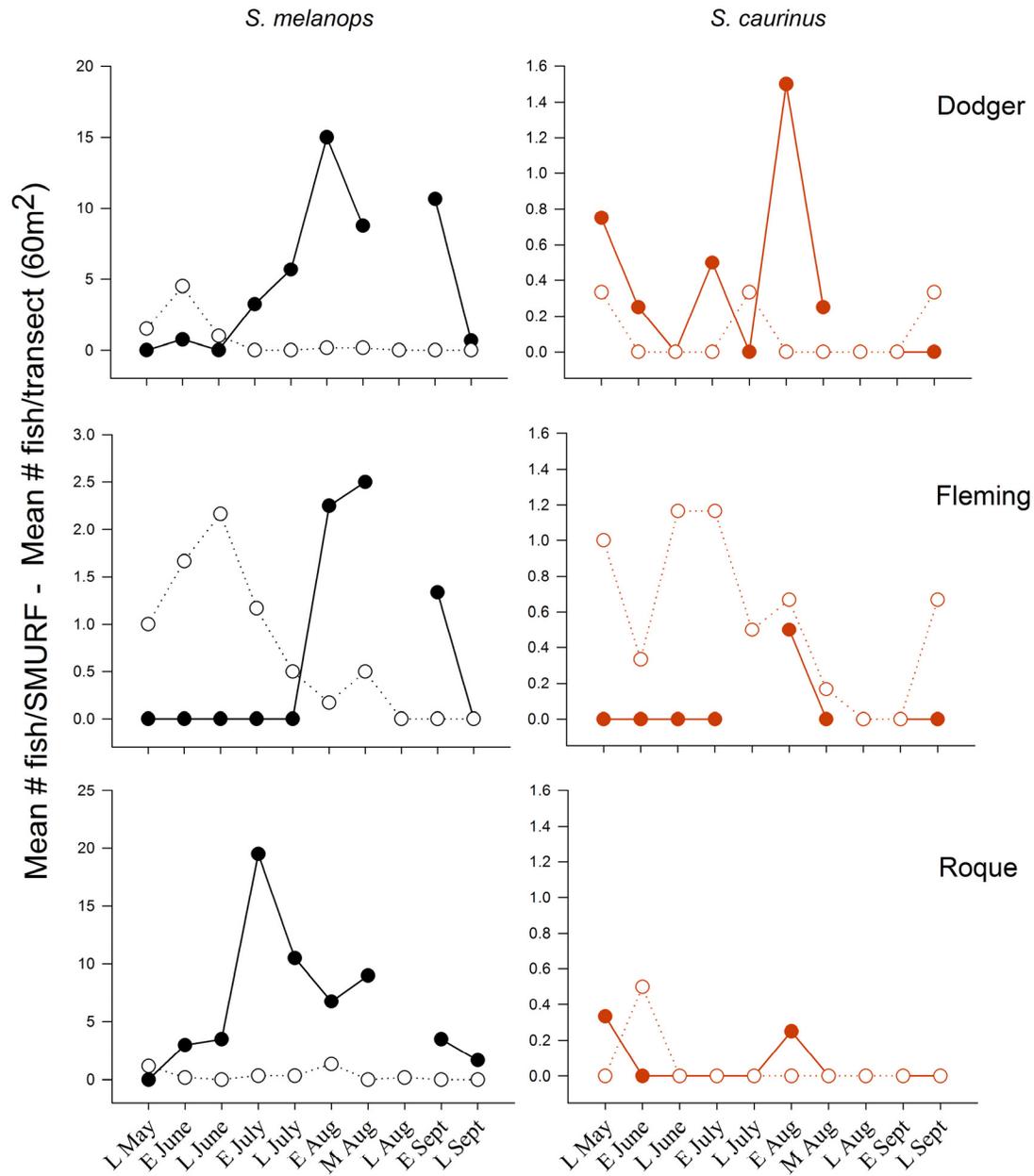


Figure 5.1. Mean number of ‘recruit’ sized *S. melanops* and *S. caurinus* observed on transects and caught in SMURFs over the study period. Open circles represent SMURF data and solid circles transect data. Early, mid and late are denoted with E, M, and L respectively. Note changing Y axis for *S. melanops*. Transect were not conducted in eelgrass in late August, therefore there is a gap on that line.

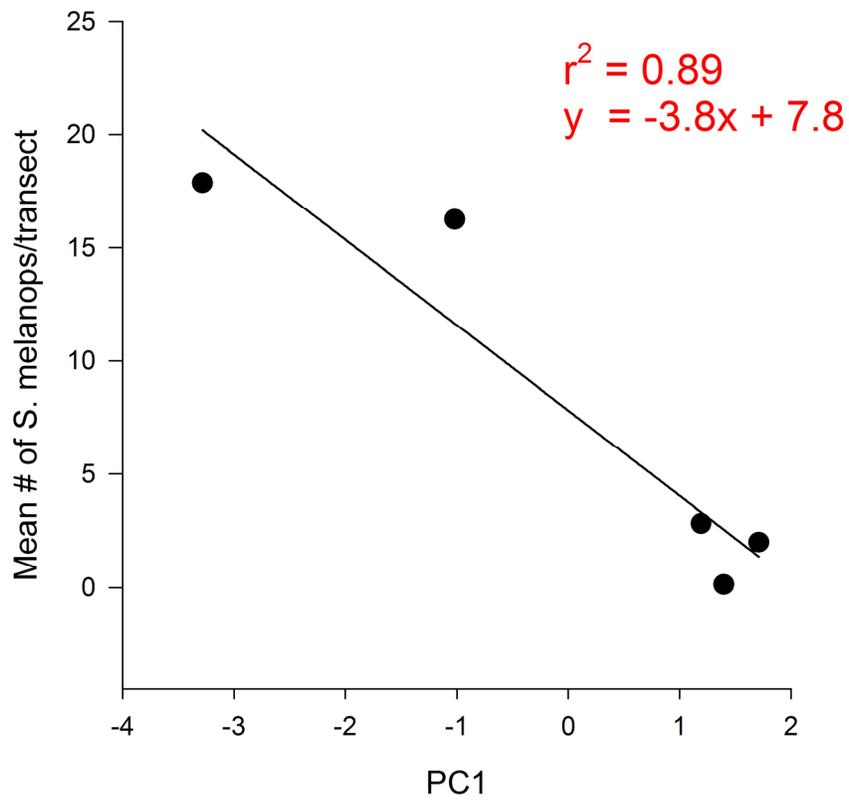


Figure 5.2. Regression between the abundance of juvenile *S. melanops* at the five study sites in August, and the first principle component of habitat variables. Regression is significant at $\alpha = 0.05$.

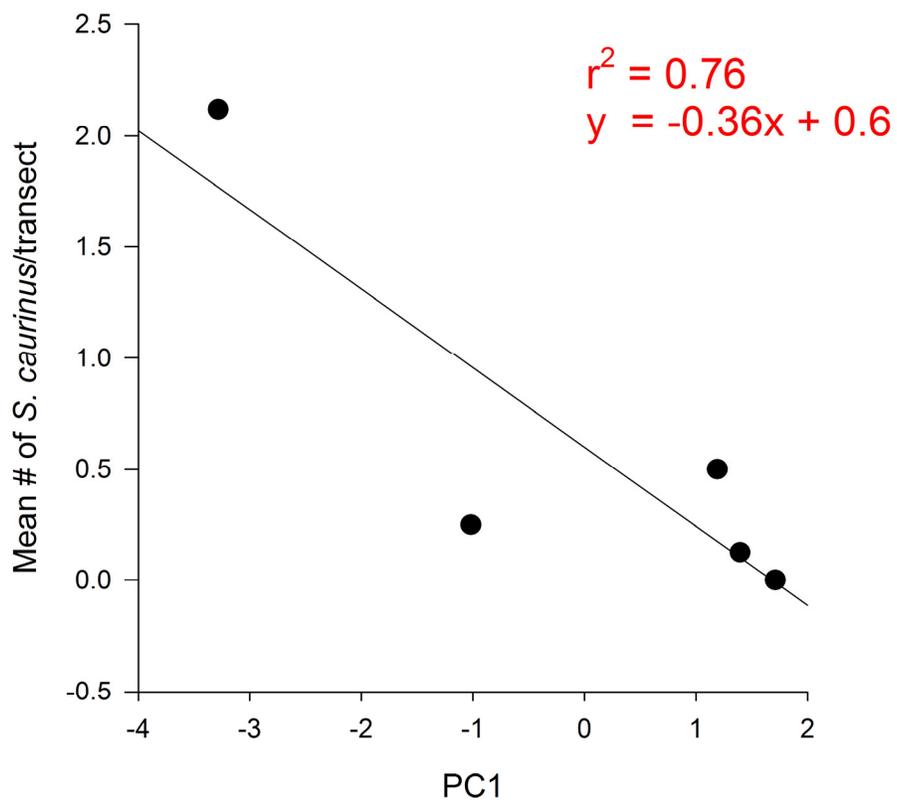


Figure 5.3. Regression between the abundance of juvenile *S. caurinus* at the five study sites in August, and the first principle component of habitat variables. Regression is not significant at $\alpha = 0.05$, but is significant at $\alpha = 0.10$.

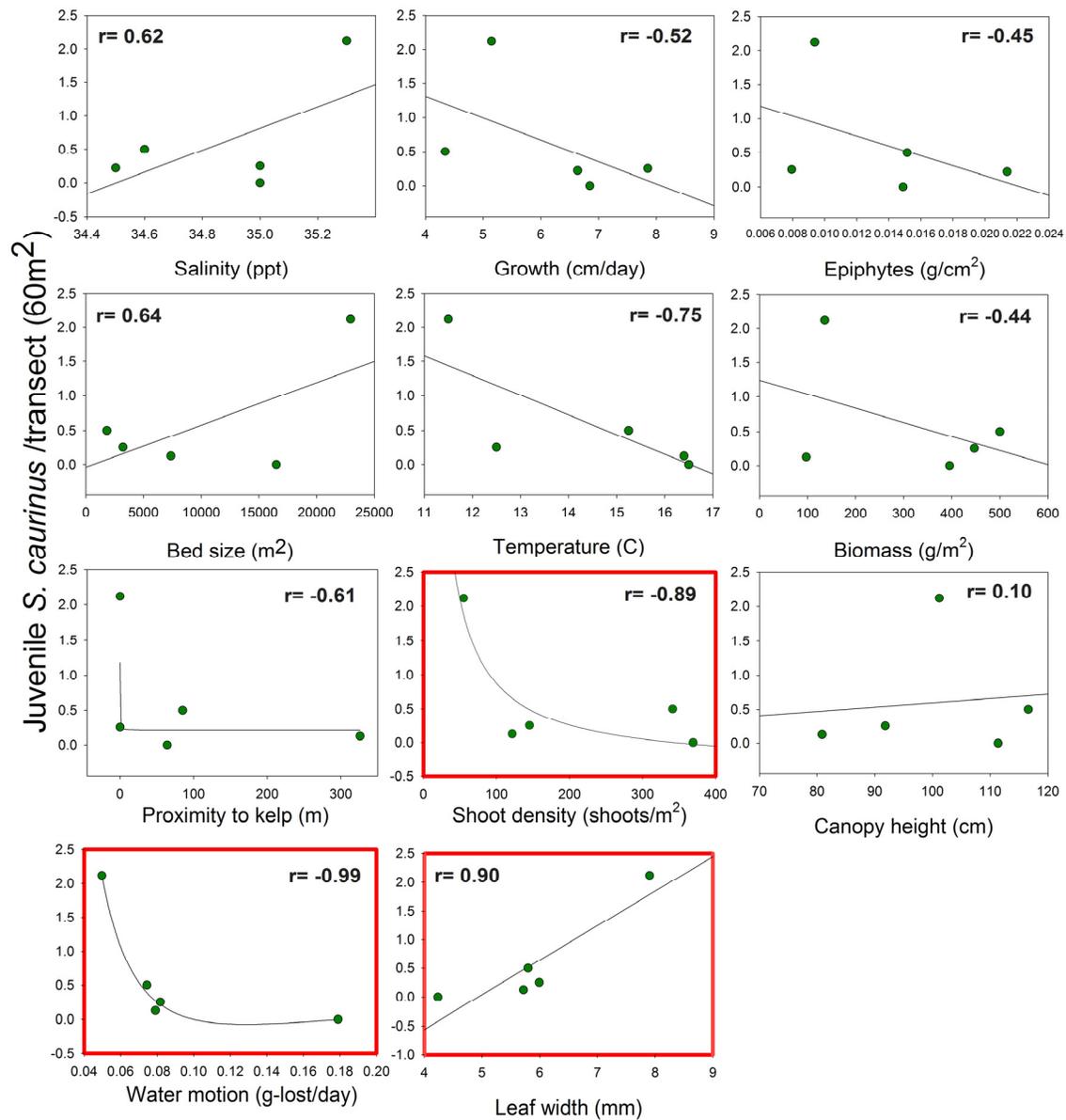


Figure 5.4. Relationships between juvenile copper rockfish abundance in eelgrass beds and all measured habitat variables. Red (bold) outline indicates correlations that are significant at $\alpha = 0.05$. Data are based on measurements taken in August. Note: proximity to kelp, shoot density, and water motion were fitted with non-linear models (inverse first-order functions), all others were assumed to be linear.

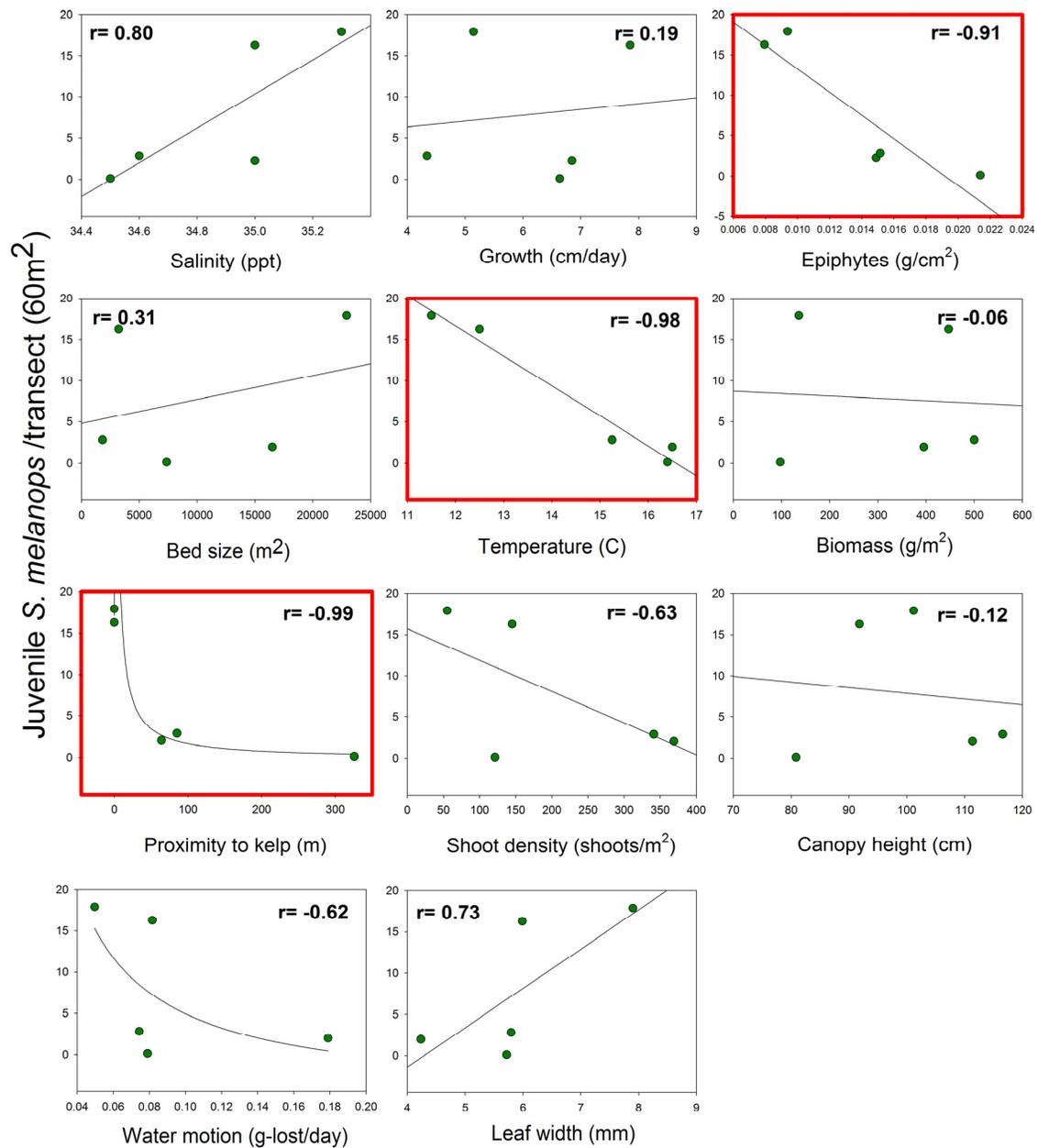


Figure 5.5. Correlations between juvenile black rockfish abundance in eelgrass beds and all measured habitat variables. Red (bold) outline indicates correlations that are significant at $\alpha = 0.05$. Data are based on measurements taken in August. Note: proximity to kelp, and water motion were fitted with non-linear models (inverse first-order functions), all others were assumed to be linear.

Chapter 6: Conclusions

6.1 GENERAL CONCLUSIONS

The overall objectives of this thesis were to identify differences in habitat use by juvenile rockfish between seagrass beds, and to relate these differences, in particular those of juvenile abundance, to the supply of settlers and characteristics of the eelgrass beds. This was accomplished through the use of ANOVAs, and a correlation analysis of all variables. Jackson *et al.* (2001) made recommendations to better conserve important fishery species that use seagrass beds. These included: 1) gaining a better understanding of how fish use differs between beds; and 2) identifying which seagrass characteristics were important in defining the habitat value for species that use the beds. The objectives laid out in this thesis addressed both of these points for at least two species of rockfishes, and will therefore significantly contribute to the information available for rockfish management decisions according to the above-mentioned authors.

The five eelgrass beds examined in this study differed significantly in their biotic and abiotic characteristics. It was hypothesized that these differences would reflect differences in the value of the habitats for juvenile rockfish, and would be correlated with their abundance. Variables most responsible for the inter-site differences were expected to be most influential on the habitat value for rockfishes. The information provided by this study confirms the hypothesis that, as in other areas, eelgrass beds in Barkley Sound are used by juvenile rockfish, and that this use is largely restricted to summer months. Two species of rockfish were consistently found in eelgrass beds during the study: *S. melanops* and *S. caurinus*, although there was some evidence that *S. paucispinus* uses eelgrass to a limited extent. The number of juveniles observed in the beds differed

significantly between sites, although these differences were restricted to months where abundance was highest.

The abundance of pelagic juveniles arriving at the sites was also found to differ significantly between beds. At the small scale of an eelgrass bed, the abundance of pelagic juveniles in the vicinity of a bed was not found to be a good indicator of recruitment levels into the beds, therefore, the use of SMURFs to measure the abundance of post-settlement rockfish in eelgrass beds is not recommended. A more direct measure of the number of fish in the bed (i.e. seining, visual counts etc.) will be necessary. However, the species composition of pelagic juveniles near the sites, as measured by SMURFs, was reflective of the species composition in the beds.

The abundance of juvenile rockfish was significantly correlated with some of the habitat variables measured; namely shoot density, temperature, proximity to kelp, epiphyte biomass, water motion, and leaf width. In Chapter 3, Dodger was identified as the most valuable eelgrass bed for rockfish because it contained both the highest number of juveniles, and for the longest period of time. Based on the conclusions from Chapter 5, it is likely that this high habitat value was related to wide leaves, the proximity of the bed to kelp habitats, low water temperature, and low epiphyte levels. The low abundance of predators, such as juvenile lingcod (*Ophiodon elongatus*) may also have increased the habitat value of this bed. This study also showed that intertidal areas were used the least by rockfishes, and that eelgrass characteristics consistent with deeper regions of the beds (wide leaves and low shoot density) were related to high juvenile abundance. Therefore, the area of extensive subtidal eelgrass at Dodger likely also contributed to its high habitat value for rockfishes.

My study has produced a very detailed description of five eelgrass beds, which will prove important as baseline data for future studies in the area. I have also shown which variables are most likely to influence the habitat value of the beds for juvenile rockfish and recommended that future studies focus on them. I have speculated that significant relationships with other variables, or more consistent relationships between species, may be obtained if future studies assessed more than 5 sites due to the tremendous amount of variation in rockfish abundance observed between transects in my study.

This study did not definitively test the nursery value of eelgrass beds for rockfishes based on the 2001 definition of a nursery by Beck *et al.* (2001) because all habitats used by the species were not tested, nor was movement to adult habitat assessed. However, it did provide information on inter-site variability of nursery value among eelgrass beds, assuming that the abundance of fish, and the length of their residency, was an adequate proxy for 'habitat value'. This information may be used as a starting point for future studies wishing to determine the nursery function of eelgrass habitat for rockfishes.

6.2 CONSERVATION IMPLICATIONS

6.2.1 Application of this information

“One of the basic aims of ecology is to develop explanatory and predictive models to account for observed patterns in the abundance and structure of populations” (Connell and Jones 1991). The objectives of this study were to do just that, and therein lies its most general contribution to ecology. More specific contributions made by this study

relate to habitat management and rockfish conservation. Eelgrass habitat has declined in area in recent years, despite the protection supposedly afforded it by the *Fisheries Act*¹¹. It is likely an important nursery habitat for many species, including rockfishes, and unarguably provides a myriad of other ‘services’ to coastal environments. Therefore, the evaluation of eelgrass beds as a juvenile habitat in this study will benefit their protection as it serves to raise the general awareness of the importance of eelgrass habitat, and increases the likelihood that it will be included in future habitat management plans more frequently.

It has been recognised for more than 25 years that information is lacking regarding the early life history requirements of rockfishes, and that this information is required to prevent their commercial over-exploitation (Laroche and Richardson 1980; Love *et al.* 2002). In spite of this recognition, information on habitat needs for juvenile rockfish is still lacking, and many rockfish populations have been reduced to critical levels. This alarming decline has prompted an interest in their conservation. Rockfish conservation areas (RCAs), where hook and line fishing is not permitted, have been established in adult habitats in an attempt to enhance these populations (DFO 2006). However, it has been well documented that the conservation of a species should also include preservation of juvenile stages, since they are often influential in determining future stock sizes, and that the best way to accomplish this is through spatial conservation measures (Love *et al.* 2002). The information provided by this study is therefore timely, given our ignorance of juvenile rockfish ecology (Love *et al.* 2002, Mathews 1990), and the lack of protection currently afforded them.

¹¹ The Fisheries Act can be viewed at http://laws.justice.gc.ca/en/showdoc/cs/F-14/bo-ga:s_34/en

6.2.2 Management recommendations

The concept that rockfish may use more than one juvenile habitat, whichever is available, is common to many species and habitats studied. For instance, Freese and Wing (2003) concluded that “*Aphrocallistes* sponges provide habitat for juvenile red rockfish. The association is fortuitous, in that many juvenile red rockfish have been noted in a wide variety of other habitat types”. It is also known that the rockfish species in this study do not rely exclusively on eelgrass beds as juvenile habitat. They have been recorded in kelp habitats, *Agarum* spp., cobble and boulder fields, etc. (Anderson 1983; Carr 1991; Halderson and Richards 1987; Love and Halderson 1991; Love *et al.* 2002; Mathews 1990). However, the loss or degradation of eelgrass, as occurred in the sponge study cited above, would be expected to negatively impact juvenile survival where other habitat types are not available. In Barkley Sound, eelgrass beds are a common subtidal habitat and may be more important to rockfishes than in areas where they are less common, or occur largely intertidally. Further to this, because rockfish recruitment is highly variable, and populations are sustained by infrequent, strong year classes, species rely on ‘bumper’ year classes to carry the population for many years (in some cases decades) (Love *et al.* 2002). In years with high recruitment, juvenile habitats may be limiting. It is in these years that eelgrass may become an ‘essential’ juvenile habitat, and that the preservation of those beds of highest value proves crucial for the maintenance of the species. Therefore, a precautionary measure of protecting eelgrass is recommended as a conservation strategy for rockfish. This action, if taken, can only be of benefit to marine ecosystems given the undeniable value of eelgrass in roles other than a nursery habitat.

It has long been recognized that not all juvenile habitats function equally, (Chittaro *et al.* 2005), and there is evidence of this in the present study based on the differences found in habitat characteristics and rockfish abundance between beds. Since we cannot preserve all beds, methods for prioritizing the selection of beds will be needed to use conservation resources most effectively. The chance that conserving eelgrass beds for rockfish will prove successful in increasing populations can be improved if beds with the highest habitat value are identified and conserved.

6.3 LITERATURE CITED

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Appendix I Photographic size progression of *S. caurinus* juveniles caught in SMURFs in 2006



1.5cm



2.6cm



1.9cm



2.7cm



2.0cm



3.4cm



2.2cm



4.2cm

Appendix II Photographic size progression of *S. melanops* juveniles caught in SMURFs in 2006



May 23
3.06cm



May 23
3.57cm



June 9
3.91cm



June 9
4.1cm



June 19
4.22 cm



June 9
4.3 cm



July 6
4.61 cm



July 6
4.85 cm



July 6
5.52 cm



Aug 2
6.44 cm