THE EFFECTS OF INCREASING HABITAT COMPLEXITY WITH ARTIFICIAL REEFS ON DEMERSAL FISH DENSITY IN COASTAL NEWFOUNDLAND WATERS

PHILIP S. SARGENT

National Library
of Canada

Acquisitions and
Bibliographic Services

395 Wellington Street
Ottawa ON K1A 0N4
Canada

Bibliothèque nationale
du Canada

Acquisisitons et
services bibliographiques

395, rue Wellington
Ottawa ON K1A 0N4
Canada

Your file Votre référence
ISBN: 0-612-84039-5
Our file Notre référence
ISBN: 0-612-84039-5

THE EFFECTS OF INCREASING HABITAT COMPLEXITY WITH ARTIFICIAL REEFS ON DEMERSAL FISH DENSITY IN COASTAL NEWFOUNDLAND WATERS

by

A thesis submitted to the
School of Graduate Studies
in partial fulfilment of the
requirements for the degree of
Master of Science

Biopsychology Programme
Memorial University of Newfoundland

August 2002

St. John's Newfoundland

Abstract

Habitat complexity and predator avoidance are important factors influencing the distribution of organisms. Structurally complex habitats offer refuge from predators and potential foraging areas. Artificial reefs increase habitat complexity in the aquatic environment. In this study, artificial reefs were used to test the hypothesis that increased habitat complexity would increase density of demersal fishes, such as cunners (*Tautogolabrus adspersus*), juvenile Atlantic cod (*Gadus morhua*), and juvenile rock cod (*G. ogac*) in the coastal subarctic waters of Newfoundland. In 1999 and 2000, five paired artificial reef and control transects and three additional control transects (each 80 m long) were deployed in Newman Sound, Newfoundland, Canada along the 15 m depth contour. Habitat complexity of substrate along the transects was expressed as fractal dimensions (D) measured at five resolutions (0.035 - 3.5 m). Densities of cunners and juvenile cod were measured on each transect during three autumn surveys in 1999 and four summer surveys in 2000 using scuba. Fractal dimensions were between $D = 1.00$ and 1.01 along unmanipulated control transects and $D = 1.16$ along reef transects, indicating a significant increase in habitat complexity due to the artificial reefs. During the day, most cunners (159 of 242) and juvenile cod (25 of 29) were observed on reef transects in close association with artificial reefs. When observed on control transects, cunners associated with unique features of increased complexity. Juvenile cod observed on control transects showed no habitat associations. Densities of cunners, juvenile Atlantic cod, and rock cod attenuated at rates of -1.08, -0.22, and -0.17 %/m respectively with distance from reefs to a "baseline" level at 15-20 m. In summer 2000, higher cunner densities were associated with increased habitat complexity during the day and few cunners (2) were observed at night. In contrast, no juvenile Atlantic cod were observed over any transects during the day, but at night, density increased, though no preference for complexity was observed. Juvenile rock cod were observed in similar densities during day and night, but individuals were aggregated near the reefs during the day and were dispersed at night. Cunners were associated with artificial reefs in both summer and autumn. Cunners were observed less frequently at low temperatures. Juvenile Atlantic cod associated with artificial reefs in autumn but not in summer. In contrast, juvenile rock cod associated with reefs in summer but not in autumn. Seasonal differences in habitat use by juvenile cod appeared to be linked to the breakdown in the thermocline in early autumn. Artificial reefs increased habitat complexity and in turn the diel and seasonal distribution and density of demersal fish species.

Acknowledgements

I wish to extend my gratitude to Dr. R. S. Gregory and Dr. D. C. Schneider for their supervision of the design and writing of this thesis. Dr. Gregory was of paramount importance in undertaking the field research for this study. I also thank him for his encouragement throughout this study. Dr. Schneider helped me grasp the analysis, statistics and process of shaping a thesis.

I wish to thank my grad committee: Dr. J. A. Brown, Dr. W. Montevecchi, and Dr. P. Snelgrove for their sage advice in writing my thesis.

I am grateful to B. Laurel, A. M. Gorman, N. Wells, J. Anderson, C. Anderson, K. Pike, A. Ogilvie, G. Sparkes, P. Frampton, S. Moulton, D. Ings, Dr. D. Methven, and Dr. J. T. Anderson for assistance in the field, advice and aid in data analysis. I am especially grateful to D. Cote for his patience, generosity and time in helping me collect data.

I thank Parks Canada (Terra Nova National Park) who provided accommodations, access to site facilities, personnel assistance, the park boat (Shanadithi II) and digital global positioning system (DGPS). Thanks to D. Scruton from the Department of Fisheries and Oceans who supplied us with a boat and motor. I wish to thank Dr. V. Gotceitas for assistance with my artificial reef design. I appreciate Hanlon's Tire Recycling in St. John's, NF for supplying the car tires free of charge for artificial reef construction.

This project was supported by a NSERC Strategic Projects Grant #337324 to Dr. D. C. Schneider, Dr. J. A. Brown, and Dr. J. T. Anderson.

Table of Contents

List of Tables

List of Figures

List of Abbreviations

Chapter 2

α - coefficient relating numbers of observations to distance

β - slope of the regression line

β_o - overall mean

$\beta_{\text{step size}}$ - fractal scaling exponent

β_{transect} - coefficient describing differences in means between transects

β_{group} - two means - one for the transect type

$\beta_{\text{step size x transect}}$ - interaction term measuring difference in fractal scaling exponents between transects

λ - step size

AC - adjacent control transect

ANCOVA - analysis of covariance

ANOVA - analysis of variance

AR - artificial reef

D - fractal dimension

D - *Desmarestia aculeata*

DC - distant control transect

GLM - general linear model

IFD - Ideal Free Distribution

K - (in equation 1) constant

K - (in Table 2.1) kelp (*Laminaria longicruris*)

L - distance

LC - large cobble

MPA - marine protected area

MC - medium cobble

n - number of surveys with data

PVC - polyvinylchloride

R - reef transect

RA - rate of attenuation

RI - rugosity index

S - sand

SA - scale of attenuation

SC - small cobble

scuba - self-contained underwater breathing apparatus

SL - standard length

US - United States

x - the distance from the reef

Chapter 3

FRP - fibreglass reinforced plastic

List of Formulae

Chapter 2

$$L(\lambda) = K\lambda^{1-D}$$

$$\beta = \frac{\ln(L_1 / L_2)}{\ln(\lambda_1 / \lambda_2)}$$

$$y = \alpha e^{\beta \bullet x}$$

$$\text{Proportion present} = \frac{\int_{\text{min}}^{L} \alpha e^{\beta x} dx}{\int_{\text{min}}^{\text{max}} \alpha e^{\beta x} dx} = 0.95$$

$$\ln L = \beta_o + \beta_{\text{ln step size}} \times \text{ln step size} + \beta_{\text{transect}} \times \text{transect} + \beta_{\text{ln step size} \times \text{transect}} \times \text{ln step size} \times \text{transect} + \varepsilon$$

$$\ln L = \beta_o + \beta_{\text{ln step size}} \times \text{ln step size} + \beta_{\text{group}} \times \text{group} + \beta_{\text{ln step size} \times \text{group}} \times \text{ln step size} \times \text{group} + \varepsilon$$

$$\text{RI} = \beta_o + \beta_{\text{step size}} \times \text{step size} + \beta_{\text{transect}} \times \text{transect} + \beta_{\text{step size} \times \text{transect}} \times \text{step size} \times \text{transect} + \varepsilon$$

Chapter 1 Overview

Artificial reef studies are numerous. Bohnsack and Sutherland (1985) reviewed 413 artificial reef references and I have examined 445 additional references from 1986 to 2001. Artificial reef studies have been conducted primarily in tropical, subtropical or temperate waters (e.g. Dewees & Gotshall 1974, Fast & Pagan 1974, Higo et al. 1980, Kock 1982, Buckley & Hueckel 1985, Chandler et al. 1985, Downing et al. 1985, Jessee et al. 1985, Lauffe & Pauley 1985, Sanders et al. 1985, Spanier et al. 1985, Guzmán et al. 1988, Hixon & Beets 1989, Chua & Chou 1994, Gregg 1995, Kim 2001). Few artificial reef studies (but see Gascon & Miller 1981) have been conducted in cold ocean regions such as Canada. Most of artificial reef studies have been of limited scientific value (Grossman et al. 1997). These studies have not quantified changes in habitat complexity after artificial reef deployment nor has the spatial scale of effect of artificial reefs been quantified. Further, few artificial reef studies have examined how abiotic factors such as time of day, season and temperature may affect species populations (Sanders et al. 1985).

The objectives of this thesis were to: (i) determine the effect of increasing habitat complexity on the densities of nearshore demersal fish species; (ii) determine the spatial scale of artificial reef effects on fish densities; (iii) examine the differences in density patterns between day and night near artificial reefs among species; and (iv) examine seasonal and temperature effects on species densities near artificial reefs.

Chapter 1 is an overview of the objectives and organization of this thesis. Chapter 2 presents the results of the experimental study including an introduction, methods and materials, results, discussion of the major findings, and conclusions and recommendations in the format of a publishable scientific paper, though with more detail than is required for publication. The introduction to Chapter 2 summarizes pertinent scientific literature to show the direction I have taken with regards to research on habitat complexity, artificial reefs and demersal nearshore species of Newfoundland. Chapter 3 is a general discussion of topics in the fields of artificial reef research, habitat complexity, and Atlantic cod beyond the scope of the results in Chapter 2. To prevent redundancy, all references used throughout this thesis are presented after Chapter 3.

Co-authorship Statement for Chapter 2

Philip S. Sargent designed and identified the research proposal, collected field data, performed statistical analyses and prepared a manuscript (Ch 2) in collaboration with David C. Schneider and Robert S. Gregory. David C. Schneider and Robert S. Gregory contributed in the development of the methods. Philip S. Sargent, David C. Schneider, and Robert S. Gregory contributed to the development of the manuscript. All co-authors contributed in editing the manuscript.

Chapter 2 Using artificial reefs to measure the effect of habitat complexity on the density of nearshore demersal fish species.

2.1 Introduction

Habitat is a subject of central importance in terrestrial and aquatic ecology. There is considerable variability in the types of habitat available in ecosystems. Habitat in the marine environment includes natural substrates such as mud, sand, gravel, cobble, boulder, bedrock, coral reefs, seagrass, macroalgae (including beds of kelp, kelp forests, etc.) and combinations of these substrates. Marine habitat may also include artificial structures constructed by humans (wharves, floating piers, oil platforms, breakwaters, shipwrecks, artificial reefs, etc.). There is also considerable variability in the distribution of habitats ranging from uniform to patchy.

Organisms often associate with specific habitat types. The Ideal Free Distribution (IFD) model predicts that organisms should match the distribution of resources (Kennedy & Gray 1993). However, reanalysis of the IFD model has shown that the current form of the model does not accurately depict the distribution of foraging animals. The distribution of organisms does not match the distribution of resources, which may be explained by violation of the model's assumptions (Kennedy & Gray 1993). The presence of predators and the effects of waterflow, and temperature may also affect the distribution of organisms (Kennedy & Gray 1993). Individual animals select habitats that promote growth and reproduction and reduce the risk of mortality (Gibson et al. 1996).

Predator avoidance influences habitat use (Gotceitas & Brown 1993, Fraser et al. 1996) and limits the local distribution of some species (Krebs 1994). Animals vulnerable to predation must make important survival decisions, such as using habitats that provide protection from predators (Stein 1979). The density of prey organisms in a complex habitat often correlates with the density of predators (Kock 1982, Buckley & Hueckel 1985). Increased predator density may lead to increased prey mortality, relative to noncomplex habitats, due to increased predator - prey encounters. However, as habitat complexity increases, both the predator capture rate and the risk of predation decrease (Cooper & Crowder 1979, Stein 1979, Coull & Wells 1983, Johns & Mann 1987, Russo 1987, Gotceitas & Colgan 1989, Nelson & Bonsdorff 1990, Mattila 1992, Gotceitas & Brown 1993, Gotceitas et al. 1995, 1997, Tupper & Boutilier 1995, 1997) because complex habitats increase available refugia (Mattila 1992). Examining when and why organisms select habitat complexity increases our understanding of their ecology. Manipulating habitat complexity and determining its effects on fish densities will increase our understanding of the spatial distributions of animals and how and why different habitats are used.

Artificial reefs alter habitat complexity in the aquatic environment. They have many different applications including the dissipation of waves (Mottet 1981), prevention of beach erosion (Anon. 1993, Jenkins & Skelly 1994), anti-trawling devices (Guillen et al. 1994, Munoz-Perez et al. 2000), use as biological filters to reduce eutrophication in coastal areas (Laihonen et al. 1997), reduction of excess nutrient loading at cage aquaculture sites (Bugrov 1994), and most commonly to enhance areas for sport diving by enhancing production of macroalgae, corals, other invertebrates and fishes (Brock 1994, Rhodes et al. 1994, Hower 1998). Even offshore oil platforms have been promoted as "artificial reefs" (McGurrin & Fedler 1989, Quigel & Thornton 1989, Reggio 1989). Artificial reefs also constitute a tool to investigate the effects of habitat complexity on fishes (Bohnsack 1991) in field experiments. Artificial reefs influence the distribution of fishes (Bohnsack 1991) and have been shown to increase fish abundance by causing fish to aggregate and increase productivity of barren sea floors (Fast & Pagan 1974, Guzmán et al. 1988, Chua & Chou 1994).

One marine finfish species on which the effects of habitat complexity can be examined in coastal Newfoundland waters is the cunner, *Tautogolabrus adspersus* (Walbaum). Population density of this species increases as substrate rugosity (complexity) increases (Tupper & Boutilier 1997). Therefore the effects of artificially increasing habitat complexity with artificial reefs should be apparent for this species. Another common species in coastal Newfoundland waters on which the effects of increasing habitat complexity may be examined is Atlantic cod, *Gadus morhua* (Linnaeus). This species was the most important commercial finfish species in Newfoundland history prior to stock collapse east of Newfoundland and subsequent implementation of a moratorium on the groundfish fishery in 1992. Despite the economic significance of Atlantic cod, the biology of juveniles (≤ 3 years old) received little research attention before 1990. A comprehensive understanding of juvenile Atlantic cod behavioural ecology and density in relation to habitat could lead to identification of enhancement opportunities and management tools critical for the future Newfoundland cod fishery.

Reducing predation on juvenile Atlantic cod can potentially enhance their survival and increase the population size of adults. The variability of survival in early life stages of fish is an important factor limiting the size of adult populations (Bradford 1992, Tupper & Boutilier 1995). Postsettlement mortality due to piscivory determines adult patterns of abundance (Hixon 1991). Juvenile Atlantic cod are prey to a wide variety of organisms (Scott & Scott 1988). Atlantic cod are cannibalistic (Daan 1973, Gjøsæter 1987, Scott & Scott 1988, Fraser et al. 1996, Grant & Brown 1998, Linehan et al. 2001) and eat large numbers of their young, especially those 0-2 years old (Bogstad et al. 1994). Juvenile Atlantic cod are prey to squid and pollock (Scott & Scott 1988), Greenland cod, *Gadus ogac* (Richardson) (= "rock cod" in Newfoundland) and cunners (Linehan et al. 2001). Marine mammals such as seals (grey seals, *Halichoerus grypus* (Fabricius) - Bowen et al. 1993; hooded seals, *Cystophora cristata* (Erxleben), harbour seals, *Phoca vitulina* (Linnaeus), and harp seals, *Phoca groenlandica* (Erxleben) - Hammill & Stenson 2001) are predators of juvenile Atlantic cod. Juvenile Atlantic cod are also prey to birds (Great cormorants, *Phalacrocorax carbo* (Linnaeus) - Barrett et al. 1990 and when abundance of capelin, *Mallotus villosus* (Müller), is low, Atlantic puffins, *Fratercula arctica* (Linnaeus) - Brown & Nettleship 1984). To increase the chances of survival, juvenile Atlantic cod must find suitable habitat that will provide refuge from predation.

Juvenile Atlantic cod are able to identify situations of increased predation risk and respond in ways that improve their chances of survival (Gotceitas & Brown 1993, Gotceitas et al. 1995, 1997, Fraser et al. 1996). Juvenile Atlantic cod appear able to assess the refuge potential of available habitats in avoiding predation and make selections based on the safety these habitats offer (Gotceitas et al. 1995). Laboratory studies have shown that in the presence of a larger conspecific predator, juvenile Atlantic cod associate with higher substrate complexity (Gjøsæter 1987, Gotceitas & Brown 1993, Gotceitas et al.

1995, Fraser et al. 1996). Despite the success of laboratory studies in examining mechanisms underlying habitat selection, predator avoidance, and feeding behaviour, field studies offer insight into the behavioural ecology a species may adopt in its natural environment. Several field studies have found that various life stages of juvenile Atlantic cod associate with habitat complexity (Gregory et al. 1996, 1997, Gregory & Anderson 1997), which has been attributed to predator avoidance (Keats et al. 1987, Fraser et al. 1996). Juvenile Atlantic cod abundance (Tupper & Boutilier 1995) and survival (Linehan et al. 2001) increase in areas of high complexity. Of all the habitat types available in nearshore Newfoundland, few appear suitable for juvenile Atlantic cod (Gregory et al. 1996, 1997). Manipulating the natural environment to test habitat associations of juvenile Atlantic cod will increase our understanding of habitat selection for this species.

Density-dependent mortality may be important in Atlantic cod (Linehan et al. 2001). Density-dependent mortality, an important factor regulating populations within cohorts of juvenile fish, may result from limited juvenile habitat. It is possible that only juveniles that settle in suitable habitat survive (Myers & Cadigan 1993). If this idea is correct, a relatively fixed number of juveniles may survive no matter how many were initially produced, due to food limitation or increased predation risk outside nursery areas (Myers & Cadigan 1993). Density-dependent factors could be especially important for demersal stages of juvenile Atlantic cod when habitat availability is low (Gregory & Anderson 1997). If habitats that allow feeding with low predation risk are limited, less competitive juvenile Atlantic cod may be forced to utilize habitats with limited food available and increased predation risk (Salvanes et al. 1992, Myers & Cadigan 1993). This limitation presents habitat managers with an opportunity for enhancement. Addition of suitable juvenile habitat should increase juvenile Atlantic cod abundance.

Artificial reefs may be useful as shelter for juvenile fishes. Tire pyramid artificial reefs have been observed to restrict the sizes of fish that can shelter within them so that more protection is provided for juveniles and small adults than larger piscivorous fish (Chua & Chou 1994). Tire reefs are potential nursery areas that can increase survival of juveniles of some commercial fish species (Guzmán et al. 1988). Artificial reefs are often more productive than nearby natural reefs (Fast & Pagan 1974, Buckley & Hueckel 1985, Laufle & Pauley 1985, Guzmán et al. 1988, Hixon & Beets 1989, Bohnsack 1991).

Artificial reef studies have been of limited scientific value to date (Grossman et al. 1997). Bohnsack and Sutherland (1985) reviewed 413 references to artificial reefs and found general agreement that artificial reefs were effective in attracting fish and were a potentially important fishery management tool. However, most of the published accounts they examined were either descriptive studies or discussed reef construction. Conclusions were often based on little or no scientific evidence and few artificial reef studies employed experimental methods including controls and replication (Bohnsack & Sutherland 1985, Grossman et al. 1997, Lindberg 1997). In my study, I have addressed several of these problems with artificial reef research. I examined 445 references to artificial reefs published between 1986 and 2001 since Bohnsack and Sutherland's (1985) review. Of these references, around 23% related to fish and few (15) of these references examined diel or seasonal patterns. No references quantitatively determined the spatial scale that artificial reefs affect the density of fish. Examination of diel and seasonal patterns of behaviour in certain fish species and the spatial scale of effect of artificial reefs may aid managers when considering implementation of artificial reef programs.

Most published marine artificial reef studies have been conducted in warm water regions (Costa Rica - Guzmán et al. 1988; Guam - Kock 1982; Israel - Spanier et al. 1985; Japan - Higo et al. 1980; Korea - Kim 2001; Kuwait - Downing et al. 1985; Puerto Rico - Fast & Pagan 1974; Singapore - Chua & Chou 1994; Virgin Islands - Hixon & Beets 1989). Many other artificial studies have been conducted in the United States (California - Dewees & Gotshall 1974, Jessee et al. 1985; Florida - Chandler et al. 1985, Sanders et al. 1985; North Carolina - Gregg 1995; and Washington - Buckley & Hueckel 1985, Lauffe & Pauley 1985). With the exception of Gascon & Miller (1981), no studies on artificial reefs have been conducted with respect to marine finfish species in coldwater regions such as Canada and there have been no studies on Canada's east coast.

In many artificial reef studies, "complexity" has often been assumed to increase, but what constitutes a significant increase has not been quantitatively defined (Bohnsack 1991). Artificial reefs and natural reefs are made up of different structural elements in different combinations. Therefore, they have been difficult to compare (Bohnsack 1991). The complexity of natural substrates has been measured by calculation of rugosity indices (Luckhurst & Luckhurst 1978, Chandler et al. 1985, Connell and Jones 1991, Tupper & Boutilier 1995, 1997). However, the precise scale of measurement has not always been presented (e.g. Tupper & Boutilier 1995, 1997). The scale used to calculate complexity will affect the resulting rugosity index. Fractal dimensions offer a means of overcoming the difficulty in quantifying complexity. Fractal dimensions are mathematical expressions of the ruggedness of a substrate's shape, independent of the nature of a substrate (Gee &

Warwick 1994). They provide direct measure of within-habitat complexity and allow potential comparisons of complexity between completely different habitats (Gee & Warwick 1994). Fractal dimensions also incorporate the concept of habitat scale (Gee & Warwick 1994). Fractal dimensions reflect the rate of change of distance measured in a linear regression when using different scales of measurement. The greater the rate of change between scales, the more complex the substrate. No artificial reef studies to date have quantified the increase of habitat complexity relative to the surrounding substrate after reef deployment. This thesis represents the first use of fractal dimensions or rugosity indices to quantify the complexity added by artificial reefs in relation to the surrounding substrate.

No published artificial reef studies have reported the spatial scale at which an artificial reef is effective. The scale of effect of artificial reefs is important for determining optimal spacing between artificial reefs. To ensure an effective artificial reef program, it is important to know the effects of the surrounding substrate on the density of the species of interest. By determining the distance over which the species density attenuates, the optimal spacing for artificial reefs can be calculated to maximize production potential, and minimize costs of reef construction and deployment.

Artificial reef studies have all but ignored abiotic factors (e.g. time of day, season and temperature) that are known to affect fish populations (Sanders et al. 1985). Habitat requirements change spatially and temporally as fish grow (Gibson et al. 1996). The habitat itself may also change over time. Organisms vulnerable to predation may undertake horizontal, diel movements to occupy habitats with greater complexity during periods when predators are more active (Stein 1979). Atlantic cod are primarily visual predators (Brawn 1969) and hence adults, which are potential predators of juvenile conspecifics, may be more efficient during daylight hours. Therefore, juvenile Atlantic cod should associate with habitat complexity to reduce predation risk during the day. At night, visual ability becomes restricted, reducing the efficiency of older conspecifics and other predators. Juvenile Atlantic cod may therefore rely less on refugia at night (e.g. Linehan et al. 2001). The abundance and distribution of juvenile Atlantic cod may change in relation to habitat complexity due to seasonal behaviour patterns. Wigley and Serchuk (1992) studied seasonal and temporal distributions of juvenile Atlantic cod <37 cm in the Georges Bank-Southern New England region and found that juvenile Atlantic cod changed depth seasonally to maintain preferred temperatures. Seasonal distribution patterns may also affect predator-prey relationships. Examination of the diel and seasonal use of artificial reefs could lead to better understanding of species predator- prey relationships with respect to predator avoidance.

Ecological knowledge of juvenile Atlantic cod, juvenile rock cod, and cunners can be gained by determining how these species use common habitat. Rock cod and cunners are predators of juvenile Atlantic cod (Linehan et al. 2001) and have been observed together in several habitats (personal observation), but how these species share a common habitat has not been examined. There have been few ecological studies on rock cod (but see Mikhail & Welch 1989, Morin et al. 1991) and no ecological studies of this species in Newfoundland. Age 0 rock cod and age 0 Atlantic cod co-occur in eelgrass, *Zostera marina* (Linnaeus), beds during much of the summer and autumn periods in nearshore Newfoundland waters (Ings et al. manuscript, personal observation). Diet overlap occurs within the nearshore fish community (Salvanes et al. 1992). Competition for food and cannibalism can both be important in regulating fish production (Salvanes et al. 1992). Juvenile rock cod may compete with juvenile Atlantic cod for food and shelter.

I hypothesized that (1) artificially increasing substrate complexity will increase densities of demersal species; (2) fish density will attenuate with distance from reefs; (3) the association of juvenile Atlantic cod will be stronger during the day than during the night; and (4) seasonal and temperature differences affect densities of species near artificial reefs.

2.2 Materials and Methods

2.2.1 Study area

I conducted this study in Newman Sound, (48°35' N, 53°55' W) Terra Nova National Park, Bonavista Bay, Newfoundland (Fig. 2.1) as part of a larger project on habitat associations of juvenile Atlantic cod. Newman Sound was a suitable location to conduct research on habitat complexity because (1) there were large areas of low complexity substrate; (2) these areas were accessible by scuba divers; (3) previous literature exists for juvenile cod in this area (Keats 1990, Keats & Steele 1992, Linehan et al. 2001, Cote et al. 2001a, 2001b); and (4) financial resources and personnel were both available.

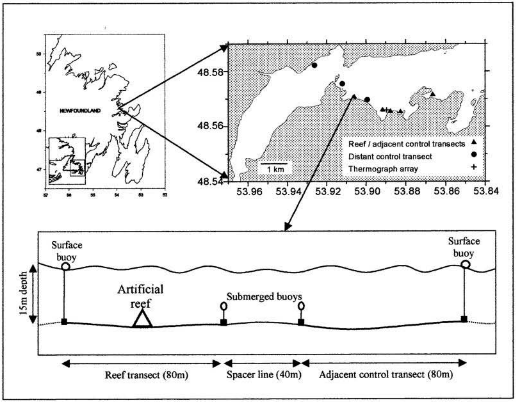

Fig. 2.1. Terra Nova National Park, Newman Sound, Bonavista Bay, Newfoundland, Canada showing locations of artificial reef / adjacent control sites, distant control sites, and the array of temperature probes. The bottom panel shows a lateral view of the layout of a reef transect paired with an adjacent control transect.

2.2.2 Reef design and construction

Five tetrahedral artificial reefs were constructed from welded rebar frames 2.0 m high and 2.0 m wide at the base in 1999. The vertical relief, measured as a straight line from the top of the reef to the centre of the reef base, was 1.9 m. The size of each reef was comparable to artificial reefs available commercially (e.g. Artificial Reefs Inc. 2001, Reef Ball Development Group Ltd. 2001); though my reefs were lighter in weight. Sides of old car tires were cut off and attached to this frame with rebar wire (Fig. 2.2). Only the sides of the tires were used, to enable divers to survey within the reefs. Each side of a reef had a different pattern of tires (Fig. 2.2a-c) to determine by observation whether fish preferred one particular pattern. However, fish observed near artificial reefs were not observed predominantly near any one pattern of tires. Fish observed near reefs swam around and through the reefs.

2.2.3 Site selection

In August 1999, scuba divers scouted outer Newman Sound for large areas of relatively noncomplex substrate (sand, gravel or small cobble), with little or no vegetation at the 15 m depth contour. This substrate had to extend for at least 80 m for distant control transects and 200 m for reef transects paired with adjacent control transects. Suitable locations were marked with a surface buoy attached to a cement construction, or "cinder" block.

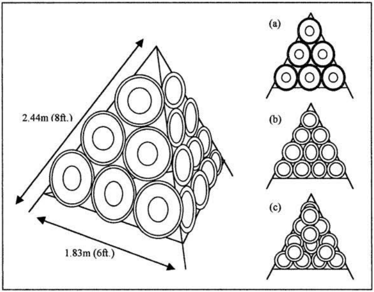

Fig. 2.2. Artificial reef design. (a) side facing forward left; (b) side facing forward right; and (c) side facing away from the observer.

2.2.4 Reef and transect deployment

Reefs were transported by boat and released over the marked sites in August 1999. Releasing the reefs from the surface helped randomize reef placement, because they landed within 15 m of the previously deployed cinder blocks. Reefs were positioned at 15 m depth and anchored by cinder blocks attached to each bottom corner.

Three types of transects were used in this study: reef transects, adjacent control transects and distant control transects. All transects were 80 m of yellow polypropylene rope marked every 5 m with orange flagging tape. These transects were placed on the substrate with L-shaped rebar spikes. Artificial reefs were placed at the midpoint of reef transects. The substrate of adjacent and distant control transects were left unmanipulated. Each adjacent control transect was placed 40 m from its paired reef transect. Distant control transects were used to control for the proximity of adjacent control transects to the artificial reef transects. Fish present over adjacent control transects may have been attracted by the artificial reefs but a distant control transect, >100 m from reef/adjacent control sites, was expected to be less influenced by the reef placement. Small submerged floats were attached by a 2 m rope at the ends of the reef and adjacent control transects closest to one another so that divers could easily find their way from one transect to its paired transect (Fig. 2.1). Forty metre 'spacer' lines were laid out between the reef and adjacent control transects at each site in 2000 (Fig. 2.1) to allow direct movement between transects during night surveys. A small red surface buoy marked each end of reef/adjacent control and distant control sites.

In 1999 there were four reef/adjacent control transects, and one distant control transect. One other reef transect was placed without an adjacent control transect in 1999. This reef transect was inadvertently placed in a bed of kelp, *Laminaria longicruris* (de la Pylaie). I left this reef in place due to logistical constraints and used it to investigate whether the reef altered species densities on kelp substrate. Surface buoys were submerged at the end of October 1999 so that ice would not destroy or move them. The reefs and transects were deep enough that ice would not affect them.

There were a total of five reef/adjacent control transects and three distant control transects in 2000. In the beginning of the 2000 season (late June - early July) divers relocated transects and reefs and assessed them for overwinter damage. The rebar wire holding the tires in place on the reefs had rusted through on most reefs and was replaced with plastic "cable" ties. Two additional distant control transects were deployed in late July, and an adjacent control transect was placed for the reef in the kelp bed.

Ten thermographs were deployed on July 18, 2000, near the middle of the study area (48° 34' 57" N, 53° 53' 24" W, Fig. 2.1) to obtain a continuous hourly temperature record for outer Newman Sound that year. Thermographs were attached to a rope anchored by cinder blocks with a large surface buoy. Starting at the surface, each probe was spaced ~1.5 m along the rope to a depth of ~13.7 m. Thermographs were also attached to the middle of one side of each reef during the first 2000 surveys (July 18-26). Three thermographs were subsequently lost from the array, and the temperature probe at 9.2 m stopped recording on August 25 so data below 8 m depth were missing after this date.

2.2.5 Scuba surveys

All transects were surveyed by divers three times in 1999 (Sept. 27-30, Oct. 11-22, Oct. 28-29) and four times in 2000 (July 17-27, Aug. 6-15, Aug. 28 - Sept. 5, Sept. 21-22). Two divers swam along each transect and recorded the number and location of Atlantic cod, rock cod and cunners on plastic slates. Divers swam parallel to one another on opposite sides of the transect, which divided their fields of view. Each diver scanned 2 m horizontally from the transect (estimated by armspan) for each 5 m of transect (10 m^2 each). Data observed within the 0-5 m marks on the transects were recorded at the 5 m mark, data from 5-10 m were recorded at the 10 m mark and so on. During the first survey period in 1999 only one diver recorded data. The diver swam directly over the transect and recorded species present within 1 m of each side of the transect, approximately 2 m horizontally (estimated by armspan), for every 5 m section of the transect (10 m^2). During the latter two survey periods in 1999 and all the 2000 day surveys, two divers recorded observations.

Cod were identified as Atlantic cod or rock cod but were recorded as *Gadus* spp. when it was not possible to differentiate between the two species. Atlantic cod were often identified by their prominent lateral line, which becomes apparent as series of elongated white marks towards the caudal peduncle at age 1 and is unmistakable at ages ≥2 years (Methven & McGowan 1998). Juvenile Atlantic cod were identified to age classes of 0, 1, 2, 3, and 4+ years. Age 1 Atlantic cod were defined as 120-190 mm, a composite of that described by Methven and Schneider (1998) where those 120-140 mm SL likely represented age 1 cod that had overwintered and those 148-190 mm SL likely corresponded to the upper limit of age 1 cod. Age 1 Atlantic cod also have a relatively mottled colour (Gregory & Anderson. 1997). Cod smaller than 12 cm were categorized as age 0. These were omitted from the final analysis because they could not be identified to species during surveys. Age 2-3 juveniles are more uniform in colour (Gregory & Anderson 1997). Age was estimated by sight based on the length groups 190-270 mm SL for age 2 and 270-370 mm SL for age 3 juveniles (Methven & Schneider 1998). Juvenile cod >370 mm were determined as age 4+ but were omitted from the final analysis because they were rarely observed. Bogstad et al. (1994) found that Atlantic cod ranging from 30 cm upward are conspecific predators on juvenile cod, but Atlantic cod larger than 40 cm are seldom prey to cod larger than themselves. Therefore, age 4+ juveniles may behave more like predators to younger conspecifics than as prey themselves to older conspecifics.

Age of observed rock cod was estimated based on the age - length categories used for Atlantic cod (Methven & Schneider 1998). The actual age - length distributions for rock cod are not likely identical to Atlantic cod. Therefore, no age estimation finer than the grouping of ages 0, 1-3, and 4+ was attempted. Juvenile rock cod estimated as age 0 or 4+ were omitted from the final analysis. Cunners were counted but no age determination was attempted. In two instances, counts of cunners were omitted from the final analysis when it appeared they had been attracted by the presence of the divers. This behaviour was not observed for the two cod species.

The first three survey periods conducted during the day in 2000 were paired with night surveys initiated after twilight. Typically two sites were surveyed during a day and the same sites were surveyed that night, or within a few nights. During night surveys, only one diver recorded observations. The recording diver (P. Sargent) used a small dive light attached to the side of his dive mask to scan across the transect for fish. Fish observed within 1 m of each side of the transect were recorded. The second diver swam behind and above the recording diver holding an aluminum pole with two large dive lights attached to each end. This configuration illuminated the bottom in front of the recording diver.

To address the concern that divers may have "herded" fish ahead of them, the direction in which the surveys were recorded was reversed from one survey to the next. This reversal of direction was accounted for during data analysis so that patterns in fish distributions associated with particular substrate features of each site could be examined. During this study, fish showed no behavioural reactions of attraction or repulsion to the presence of divers and lights at night unless divers attempted to move within approximately 0.5 m of the fish. When divers moved closer than 0.5 m juvenile cod retreated to a greater distance from the diver.

During the first surveys in 1999 one diver videotaped the substrate along the transects to identify substrate characteristics with which certain species may have associated. Substrate was categorized as sand, small cobble (rocks <10 cm diameter), medium cobble (rocks 10-25 cm diameter), large cobble (rocks >25 cm diameter), kelp (*L. longicruris*), *Desmarestia aculeata* (Linnaeus) or as a combination of these substrates. Before the transects were removed, the reef/adjacent control site at Hefferns Cove East was videotaped a second time because it was observed that at this site the kelp, *L. longicruris*, had grown considerably from 1999 to 2000.

Two artificial reefs were removed on September 21-22, 2000. These two sites and the two sites where reefs were still present were surveyed once again on September 26 to allow qualitative comparisons of before and after effects of reef removal.

2.2.6 Substrate complexity measurements

The complexity of reef and control transects was measured using five scales (step sizes) ranging two orders of magnitude (0.035, 0.175, 0.35, 1.75 and 3.5 m). Rugosity indices (Luckhurst & Luckhurst 1978) were calculated for the 1.75 and 3.5 m step sizes at both 10 and 80 m scales. The number of steps taken by each step size over the 10 m or 80 m distance along the transect multiplied by the step size determined the surface distance covered. This distance was then divided by the linear distance over which it was measured (10 or 80 m) to produce a substrate rugosity index. The higher the index, the higher the complexity. A rugosity index is only an estimate of the surface area of a substrate (Luckhurst & Luckhurst 1978). However, these estimates have been used in the past to make comparisons. Rugosity indices <1.00 are due to measurement error or misspecification of the true linear distance covered by the transect. The transects sometimes sank into the substrate or twisted around the rebar spikes keeping them in place so that the linear distance over which complexity was measured was not precisely 10 m or 80 m respectively.

Fractal dimensions were calculated as a measure of substrate complexity for each transect by measuring distance with all five step sizes over a 10 m linear distance of the transects. This method was equivalent to the "dividers method" described by Mark (1984). Ln (# steps) × step size was regressed against ln (step size),

$$L(\lambda) = K\lambda^{1-D}, \tag{1}$$

where L is the length measured using the step size λ, K is a constant, $1 - D$ is the slope of the regression, and the fractal dimension is D (Mandelbrot 1982). The boundary dimensions are $1 < D < 2$ (Gee & Warwick 1994). The larger the value of D the more complex the object measured.

Several different devices were used for each step size to measure surface distances. The smallest step size (0.035 m) was measured with a small aluminum chain (6.3 m long). Each chain link had an inside diameter of 3.5 cm. The next two step sizes were constructed from sections of PVC piping cut to appropriate lengths and strung on short sections of polypropylene rope. This created short chains of PVC piping with link sizes of 17.5 cm and 35 cm respectively. The 1.75 m step size was made from one continuous length of PVC piping. The largest step size (3.5 m) was made of a 3.5 m aluminum pole. The 17.5 cm and 35 cm step sizes were marked at 2 cm intervals while the larger 1.75 m and 3.5 m step sizes were marked at 25 cm intervals. Divers were able to estimate the distance that a measuring device extended beyond the end of the 10 m section being measured within approximately 0.5 cm on the 17.5 and 35 cm PVC piping and 5 cm on the larger poles.

Entire 80 m transects were measured with the two larger step sizes (1.75 and 3.5 m), producing two data points when at least three points are needed for regression in fractal dimension calculations. Therefore, fractal dimensions were estimated at the scale of 80 m from the slope of the line plotted between the two distances measured from these step sizes,

$$\beta = \frac{\ln(L_1 / L_2)}{\ln(\lambda_1 / \lambda_2)} \tag{2}$$

where β is the slope of the line, L_1 is the length measured using step size λ_1, (1.75 m), and L_2 is the length measured using step size λ_2 (3.5 m). One minus the slope (1 - D), in this case 1 - β equaled the fractal dimension (D).

To make direct comparisons of complexity between the 10 m transect sections and the full 80 m transects, equation 2 was also used to calculate fractal dimensions for the two large step sizes at the scale of 10 m. This allowed me to determine whether complexity estimated at the scale of 10 m differed from that estimated at the scale of 80 m.

Complexity was only measured using all five step sizes for a 10 m section in the middle of each transect. This approach allowed comparison of the complexity where the artificial reef was present on the reef transects to equivalent sections on adjacent and distant control transects. Beginning at one end of the 10 m section of transect, each measuring device was laid along the substrate under the transect to conform as closely as possible to all contours and depressions along the 10 m distance (Fig. 2.3). If the transect curved horizontally the measuring device was pivoted along that curve until the measuring device came back in line with the transect. The number of times each step size was flipped within the 10 m section was recorded on underwater slates.

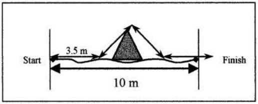

Fig. 2.3. Measurement of substrate complexity on a reef transect using the 3.5 m step size.

Due to the time constraints of using scuba at 15 m depth, complexity along the reef transects was measured in two steps; first underwater and then on land. Divers could only measure the complexity using the two largest step sizes for the full 10 m underwater. The complexity of the substrate was measured with the smaller step sizes over the 10 m section excluding the distance occupied by the reef itself. The complexity of the substrate was measured up to the point where the transect touched the base of the reef and this was repeated on the opposite side of the reef to the end of the 10 m section of transect. After the study the reefs were removed and taken to shore, where the complexity of one reef was measured with the three smaller step sizes. Because of the reef design, two levels of complexity within the reef were measured at once. A complexity measure of the tire openings themselves would produce a measure equivalent to a series of tires lying flat on the substrate. However, the open space within the reef had to be taken into account. As shown in Fig. 2.4, the smaller step sizes, in the form of chains, were inserted into the openings of the tires and allowed to drop to the substrate. The chains then followed the substrate until the chain could extend vertically to touch the next part of the reef. This process was continued across the entire reef. Because there were three different tire patterns on the reefs, complexity was measured across each pair of sides and the results were averaged for each step size (Fig. 2.5). The averages were then added to each 10 m reef transect complexity measure for each step size respectively. The first 10 m of one of the adjacent control transects (Hefferns Cove East) was also measured at all step sizes as it appeared more complex than other control transects and seemed to affect the density of cunners. The full 80 m transects were measured with the two larger step sizes to make overall comparisons of complexity among reef, adjacent control, and distant control transects.

2.2.7 Data analysis

The mean density of each fish species within every 5 m section of transect was calculated for each transect type in 1999 and 2000. The kelp bed site was averaged separately and compared to other sites. Surveys from sites (reef/adjacent control transects and distant control transect) with no observations of a particular species were not used in computing averages. Standard errors were calculated for the averages. When there was only one observation for the entire year, no standard error could be calculated.

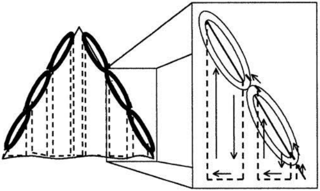

Fig. 2.4. Longitudinal section of an artificial reef demonstrating how complexity was measured by weaving a chain (small dashed lines) of the step size across and within the reef.

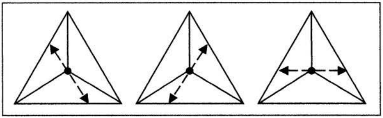

Fig. 2.5. Top view of artificial reefs showing sections across which complexity was measured (arrowhead dashed lines) at scales 0.035, 0.175, and 0.350 m.

The calculated data sets for average number of fish observed per m^2 during the day on reef transects in 1999 and 2000 were combined for each species to examine the attenuation of density with distance from reefs. I then computed the scale of attenuation (SA) of the artificial reefs by plotting the average species densities with distance from the reefs. SA is important in determining optimal reef spacing. A preliminary estimate of SA was made by regressing average number of fish observed with distance from reefs on linear and log - linear plots. From this the most linear plot was accepted. All density data were log (x + 1) transformed. Because the linear and log - linear plots were similar, the log - linear plots were used for analysis because they allowed calculation of a rate of percent decrease in fish density with distance from the reefs (rate of attenuation). The preliminary SA estimate was determined visually as the distance from the reef at which fish density converged on or near zero. Then beginning at 40 m from the reef, data from each 5 m section of transect were removed progressively from the overall data plot towards the estimated SA distance. Three regressions were then plotted: (1) the data within the distance of the preliminary SA, (2) data from 5 m greater than the preliminary SA from the reef, and (3) data from 5 m less than the preliminary SA. Predicted values from each of these regressions were plotted against their respective residuals. The model was rejected if the residuals formed a pattern. The estimate of attenuation parameter was accepted when residuals were homogeneous as judged by a uniform band from left to right in residuals plots against fitted values. The distance at which the regression's residuals showed no pattern was determined as the best estimate for the SA for that particular species. The relation was expressed by

$$y = \alpha e^{\beta \bullet x} \tag{3}$$

where α is the coefficient relating fish count per m^2 to distance, β is the slope and expresses the percent decrease in density of each species with distance from the reef within the SA, and x is the distance (m) from the reef. The rate at which a species attenuates within the SA is the rate of attenuation (RA) and is estimated by β.

The distance at which 95% of the expected density of a species occurs within the SA was estimated using equation (4), which gives the area under the regression line,

$$\text{Proportion present} = \frac{\int_{\min}^{L} \alpha e^{\beta x} dx}{\int_{\min}^{\max} \alpha e^{\beta x} dx} = 0.95 \qquad (4)$$

where β is the slope of the regression, L is the distance at which 95% of the area under the regression line falls, *min* is the minimum distance at which the regression line begins, and *max* is the maximum distance at which the regression line ends (i.e. SA). The minimum distance at which the regression line begins is 5 m because the first observations were recorded 0-5 m away from the reefs.

Comparisons were made between 1999 and 2000 data for each species. I used late September to late October (autumn), 1999, and late July to late September (summer), 2000, to make seasonal comparisons. To compare day and night artificial reef usage patterns by fish, I used the first three day surveys and corresponding night surveys in 2000. To assess the effects of reef removal, I compared data from the last full set of surveys (Sept. 21-22), when all the reefs were still in place, with data collected after two reefs were removed (Sept. 26). The two reef removal sites (manipulations) were compared before and after reefs were removed. A similar comparison was made with the two sites where the reefs remained in place (controls).

I determined whether the amount of complexity added by artificial reefs was significant in relation to the unmanipulated substrate, using the estimated fractal dimensions and rugosity indices with Minitab (1992) and SAS (1999) using $\alpha = 0.05$. First, complexity (fractal dimensions) within each group of transects at the scale of 10 m were compared using an ANCOVA with the General Linear Model (GLM) command using equation 5 to determine whether transects within each grouping were different,

$$\begin{aligned}\ln L = \beta_o + \beta_{\ln \text{ step size}} \times \ln \text{ step size} + \beta_{\text{transect}} \times \text{transect} + \beta_{\ln \text{ step size} \times \text{transect}} \\ \times \ln \text{ step size} \times \text{transect} + \varepsilon\,,\end{aligned} \tag{5}$$

where L is the distance measured, β_o is the overall mean, $\beta_{\ln \text{ step size}}$ is the fractal scaling exponent, β_{transect} is the coefficient describing differences in means between transects, $\beta_{\ln \text{ step size x transect}}$ is the interaction term measuring difference in fractal scaling exponents between transects, step size is the resolution at which the distance was measured, transect was each transect within a group of transect types measured (e.g. reef transect #1, #2, #3, and #4) and ε is the model error term. This model was used to compare regression slopes within each group of transects, excluding the kelp bed site, (4 reef transects, 4 adjacent control transects, and 3 distant control transects). The interaction term ($\beta_{\ln \text{ step size} \times \text{transect}} \times \ln \text{ step size} \times \text{transect}$) determined whether there was a significant difference in complexity among transects.

To determine whether the groups of transects differed from one another, I compared the slopes of regressions for each group using an ANCOVA executed by the GLM:

$$\ln L = \beta_o + \beta_{\text{ln step size}} \times \text{ln step size} + \beta_{\text{group}} \times \text{group} + \beta_{\text{ln step size} \times \text{group}} \times \text{ln step size} \times \text{group} + \varepsilon \,, \tag{6}$$

where the symbols are the same as in equation 3 except β_{group} represents two means, one for the transect type, and group represents the type of transect (reef, adjacent control, or distant control).

I next determined whether the results of complexity comparisons among groups of transects were the same when compared at the 10 m and 80 m scales. Groups of transects were compared by fractal dimensions with an ANCOVA executed with the GLM command using equation 6. However, this time only the data from the two largest step sizes were used at the 10 m and 80 m scales. The results were then compared between these scales. The same comparisons were made using the rugosity indices calculated for the 10 m sections and 80 m transects with a two-way ANOVA using the GLM command and equation 7:

$$\text{RI} = \beta_o + \beta_{\text{step size}} \times \text{step size} + \beta_{\text{transect}} \times \text{transect} + \beta_{\text{step size} \times \text{transect}} \times \text{step size} \times \text{transect} + \varepsilon \,, \tag{7}$$

where RI is the rugosity index and all the other symbols are the same as in equations 5 and 6.

I made comparisons of complexity of the reef and adjacent control transects in the kelp bed to non-kelp transects. I used ANCOVAs with the GLM command and the model in equation 6 to compare the slope of the regression for the 10 m section of the kelp bed reef transect with that of the other reef transects and for the kelp bed adjacent control transect with the other adjacent control transects. I also compared the fractal dimension estimated for the first 10 m of the adjacent control transect at the Hefferns Cove East site with other adjacent control transect estimates by comparing slopes using an ANCOVA with the GLM command from equation 6.

2.3 Results

2.3.1 Substrate complexity

The dominant substrate types along most transects were sand and small or medium cobble (Table 2.1). The substrate along the distant control transect at New Bridge Cove was not uniform, ranging from sand to light cobble. This was not recorded in enough detail to include in Table 2.1. The substrate under the -40 to -20 m marks of the adjacent control transect of the site at Hefferns Cove East was predominantly large cobble which was rare along other transects (Table 2.1). There was a noticeable difference between years in the amount of macroalgae, *D. aculeata* and *L. longicruris*, on the reef transect at this site (Table 2.1). The amount of macroalgae increased in 2000, especially between the 15 and 40 m marks of the transect. During reef removal, initial growth of *D. aculeata* and *L. longicruris* was observed on transect ropes and small flora and fauna on the tires.

Table 2.1. Observed substrate types at each study site for each 5 m section of each transect.

		Distance from centre of transect (m)															
Site		-40	-35	-30	-25	-20	-15	-10	-5	5	10	15	20	25	30	35	40
Mount Stamford	R	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S
	AC	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S
Little South Broad	R	MC	M/LC	SC	SC	MC	SC	MC/S	MC/S	SC/S	S	S	S	MC/S	MC/S	SC/S	SC/S
	AC	MC	MC	M/LC	MC	MC	MC/S	LC/S	LC/S	LC/S	LC/S	LC/S	LC/S	SC/S	SC/S	SC/S	S
Hefferns West	R	S	S	S	S	S	S	S	S	S	S	S	S	S	MC	MC	MC
	AC	S	S	S	SC/S	SC/S	S	S	S	S	S	S	S	S	S	S	S
Hefferns East (1999)	R	S	S	S	S	S	S	S	S	S/K/D	S/K/D	S	S	S	S/K	S	S/K
	AC	M/LC	M/LC	M/LC	M/LC	M/LC	MC	MC	SC/S	SC/S	SC/S	SC/S	SC/S	SC	SC	SC/S	SC/S
Hefferns East (2000)	R	S	S	S/K/D	S	S	S/K	S/D	S	S/K/D	S/K/D	S/D	S	S/K	S/K/D	S/K	S/K
	AC	M/LC	M/LC	M/LC	M/LC	M/LC	MC	MC	SC/S	SC/S	SC/S	SC/S	SC/S	SC	SC	SC/S	SC/S
Little Minchins	R	MC	MC	MC	MC/S	SC/S	SC/S	SC/S	S/K	S/K	S/K	S/K	S/K	S/K	S/K	S/K	S/K
	AC*	S/K	S/K	S/K	S/K	S/K	S/K	S/K	S/K	S/K	S/K	S/K	S/K	S/K	S/K	S/K	S/K
Mount Stamford	DC	S	S	S	SC/S	SC	SC	SC	SC	SC	S	SC/S	SC/S	SC/S	SC/S	SC/S	SC/S
Outside Narrows	DC*	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S	S

Key: * - Transect not videotaped but recalled from diver observations.

R = reef transect; AC = adjacent control transect; DC = distant control transect; S = sand; SC = small cobble; MC = medium cobble; LC = large cobble; K = kelp (*L. longicruris*); D = *D. aculeata*

The plotted regressions of the four reef transects, four adjacent control transects, and three distant control transects gave fractal dimension estimates of $D = 1.16$, $D = 1.00$, and $D = 1.01$ respectively (Fig. 2.6). There was no significant difference in complexity within non-kelp reef transects, adjacent control transects, or distant control transects when compared by fractal dimensions estimated at the 10 m scale using 5 step sizes (Table 2.2). Substrate complexity of reef transects was statistically higher than adjacent and distant control transects. There was no statistical difference in complexity between the two types of control transects (Table 2.2). Fractal dimensions of control transects were essentially Euclidean (i.e. $D \approx 1.00$), verifying that substrate complexity among controls was low. The difference in complexity between non-kelp reef transects was close to significant ($p = 0.060$). However, the slopes of their regressions ranged from -0.1769 to -0.1476, (difference of 0.0293) so that substrate complexity of reef transects were not substantially different.

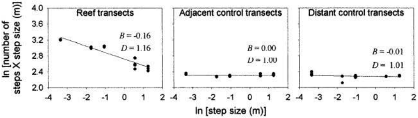

Fig. 2.6. Linear regressions of the natural log of the number of steps multiplied by step size (distance measured) plotted against the natural log of the step sizes for the reef transects, adjacent control transects and distant control transects.

Table 2.2. Results of statistical comparisons of complexity estimated by fractal dimensions and rugosity indices at different scales ($\alpha = 0.05$).

Comparison	Scale (m)	Step sizes used	Model (equation #)	F - value (interaction term)	p - value (interaction term)	Result
D among 4 non-kelp R	10	All	5	$F_{[7,15]} = 3.70$	0.060	Not significant
D among 4 non-kelp AC	10	All	5	$F_{[7,19]} = 0.86$	0.487	Not significant
D among 3 DC	10	All	5	$F_{[5,14]} = 0.42$	0.669	Not significant
D between R and AC	10	All	6	$F_{[3,39]} = 146.29$	< 0.0001	Significant
D between R and DC	10	All	6	$F_{[3,34]} = 87.06$	< 0.0001	Significant
D between AC and DC	10	All	6	$F_{[3,34]} = 0.29$	0.595	Not significant
D between kelp bed R and non-kelp R	10	All	6	$F_{[3,24]} = 0.21$	0.650	Not significant
D between kelp bed AC and non-kelp AC	10	All	6	$F_{[3,24]} = 1.36$	0.257	Not significant
D between Hefferns East AC (first 10 m) and non-kelp AC	10	All	6	$F_{[3,24]} = 6.12$	0.022	Significant
D between R and AC	10	1.75 & 3.5 m	6	$F_{[3,15]} = 4.03$	0.068	Not significant
D between R and DC	10	1.75 & 3.5 m	6	$F_{[3,13]} = 2.39$	0.153	Not significant
D between AC and DC	10	1.75 & 3.5 m	6	$F_{[3,13]} = 0.47$	0.510	Not significant
D between R and AC	80	1.75 & 3.5 m	6	$F_{[3,15]} = 0.45$	0.514	Not significant
D between R and DC	80	1.75 & 3.5 m	6	$F_{[3,13]} = 0.63$	0.446	Not significant
D between AC and DC	80	1.75 & 3.5 m	6	$F_{[3,13]} = 1.79$	0.211	Not significant
RI between R and AC	10	1.75 & 3.5 m	7	$F_{[3,15]} = 3.71$	0.078	Not significant
RI between R and DC	10	1.75 & 3.5 m	7	$F_{[3,13]} = 2.36$	0.156	Not significant
RI between AC and DC	10	1.75 & 3.5 m	7	$F_{[3,13]} = 0.32$	0.587	Not significant
RI between R and AC	80	1.75 & 3.5 m	7	$F_{[3,15]} = 0.33$	0.578	Not significant
RI between R and DC	80	1.75 & 3.5 m	7	$F_{[3,13]} = 0.69$	0.424	Not significant
RI between AC and DC	80	1.75 & 3.5 m	7	$F_{[3,13]} = 1.75$	0.215	Not significant

Key: *D* = fractal dimensions; RI = rugosity index; R = reef transects; AC = adjacent control transects; DC = distant control transects

Fractal dimensions for the kelp bed reef and adjacent control transects were D = 1.18 and D = 1.02 respectively. These estimates were higher than those of the non-kelp reef transects and adjacent control transects respectively but not statistically.

There was a statistically significant difference in substrate complexity between the non-kelp adjacent control transects and the first 10 m section of the adjacent control transect at Hefferns Cove East (where substrate was large cobble) (Table 2.2). The estimated fractal dimension for the first 10 m section of the adjacent control transect at Hefferns Cove East was D = 1.03. When complexity was compared among types of transects by fractal dimensions estimated using the two largest step sizes there were no differences at either the scale of 10 m or 80 m (Table 2.2). There were no differences in complexity between types of transects when compared by rugosity indices estimated using the two largest step sizes compared at the scale of 10 and 80 m (Table 2.2). The fractal dimensions estimated for each transect at both the scales of 10 and 80 m used in these comparisons are presented in Table 2.3.

2.3.2 Temperature

Biweekly plots of temperature data from the array and reef thermographs in 2000 showed there was general thermal stratification of the water column from 0 to about 16 m at the study sites from July 20 to mid-September (Fig. 2.7). On August 1 the water column temporarily became isothermal. From September 12 to the end of the study period, there was no thermal stratification. Thermal stratification broke down at the end of August as the differences in water temperature throughout the water column were reduced relative to the previous plot in mid-August (Fig. 2.7). The change from a

Table 2.3. Step sizes, number of steps taken, linear distance measured and calculated rugosity indices and fractal dimensions for non-kelp reef, adjacent control and distant control transects.

Site	Step size (m)	# of steps	Actual distance covered (m)	Linear distance covered (m)	Rugosity index	*D* (10 m)	*D* (80 m)
Outside Narrows	3.5	2.80	9.80	10	0.98	0.99	0.90
	1.75	5.57	9.75	10	0.98		
	3.5	22.64	79.25	80	0.99		
	1.75	42.26	73.95	80	0.92		
New Bridge Cove	3.5	2.83	9.90	10	0.99	0.99	1.01
	1.75	5.63	9.85	10	0.99		
	3.5	22.61	79.15	80	0.99		
	1.75	45.49	79.60	80	1.00		
Mount Stamford	3.5	2.87	10.05	10	1.01	0.97	1.00
	1.75	5.63	9.85	10	0.99		
	3.5	22.64	79.25	80	0.99		
	1.75	45.29	79.25	80	0.99		
Little South Broad Cove reef transect	3.5	3.43	12.00	10	1.20	0.86	1.02
	1.75	7.57	13.25	10	1.33		
	3.5	23.39	81.85	80	1.02		
	1.75	47.31	82.80	80	1.04		
Little South Broad Cove control transect	3.5	2.87	10.05	10	1.01	0.98	1.12
	1.75	5.66	9.90	10	0.99		
	3.5	21.04	73.65	80	0.92		
	1.75	45.83	80.20	80	1.00		
Hefferns Cove East reef transect	3.5	3.43	12.00	10	1.20	1.38	1.08
	1.75	8.91	15.60	10	1.56		
	3.5	22.83	79.90	80	1.00		
	1.75	48.11	84.20	80	1.05		
Hefferns Cove East control transect	3.5	2.89	10.10	10	1.01	0.96	1.06
	1.75	5.63	9.85	10	0.99		
	3.5	21.99	76.95	80	0.96		
	1.75	45.91	80.35	80	1.00		
Mount Stamford Cove reef transect	3.5	3.21	11.25	10	1.13	1.21	1.02
	1.75	7.43	13.00	10	1.30		
	3.5	23.41	81.95	80	1.02		
	1.75	47.54	83.20	80	1.04		
Mount Stamford Cove control transect	3.5	2.99	10.46	10	1.05	1.00	0.98
	1.75	5.98	10.46	10	1.05		
	3.5	23.10	80.85	80	1.01		
	1.75	45.57	79.75	80	1.00		
Hefferns Cove West reef transect	3.5	3.61	12.65	10	1.27	0.90	0.92
	1.75	6.74	11.80	10	1.18		
	3.5	24.91	87.20	80	1.09		
	1.75	47.03	82.30	80	1.03		
Hefferns Cove West control transect	3.5	2.99	10.45	10	1.05	0.93	0.97
	1.75	5.69	9.95	10	1.00		
	3.5	23.07	80.75	80	1.01		
	1.75	45.14	79.00	80	0.99		

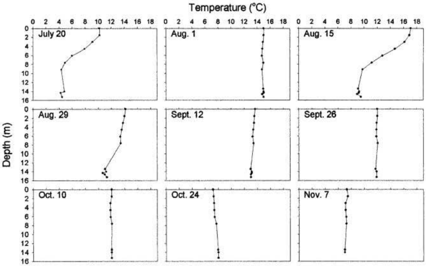

Fig. 2.7. Representative temperature - depth profiles taken approximately every two weeks from daily mean water temperature data of temperature probes throughout outer Newman Sound, Bonavista Bay, Newfoundland July - November 2000.

stratified to an isothermal water column corresponded with an increase in bottom temperature (Figure 2.8). Temperature was consistent among artificial reefs changing equally with little variance among sites (Fig. 2.8). Daily temperature means went through periods of fluctuation interspersed with periods of stability in 2000. Water temperature at reefs showed significant variability until the end of August after which water temperature increased and stabilized. Water temperature at the reefs gradually decreased from September until the end of the study with interspersed periods of high variability.

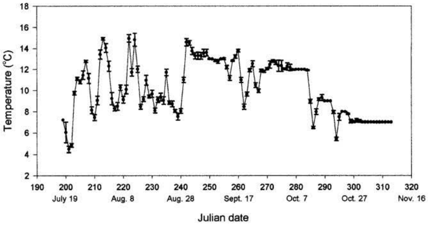

Fig. 2.8. Means of mean daily water temperature data from all artificial reef sites plotted for the 2000 field season. Error bars indicate standard deviations.

2.3.3 Fish species density and distribution

I observed 1001 fish associated with transects during my study; 99 in 1999 and 902 in 2000. Of all these fish, 414 were observed over reef transects. I observed 617 cunners, 257 juvenile Atlantic cod, and 26 juvenile rock cod. Cunners associated with artificial reefs in 1999 and 2000 (Fig. 2.9). Fifty cunners were observed on non-kelp sites in 1999 and 192 were observed during the day in 2000 (242 total). Twenty-four cunners observed in 1999 and 135 observed in 2000 were over reef transects. The highest peaks in numbers observed were near the reef but there were also relatively high numbers observed at the ends of the reef transect in 1999 and increased numbers between 20 and 40 m on the reef transect in 2000 (Fig. 2.9). Twenty-six of the cunners observed in 1999 and 53 in 2000 were over adjacent control transects. There were no cunners observed on distant control transects in 1999 and only 4 observed in 2000, which were dispersed. There was a substantial number of cunners associated with the first 30 m of the adjacent control transects. These cunners were associated with complexity from the large cobble at the Hefferns Cove East site.

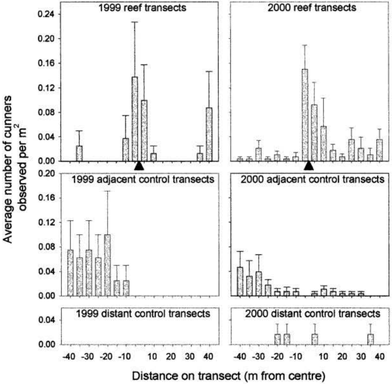

Fig. 2.9. Average numbers of cunners observed along transects in 1999 and 2000. Error bars indicate one standard error from each mean. Abundance recorded at 0 - 5 m was plotted at the 5 m mark and so on. Black triangle indicates the location of the artificial reefs.

Cunners were observed more frequently near the artificial reef than elsewhere in Little Minchins Cove but there were also relatively high numbers over the rest of the transects (Fig. 2.10). A total of 372 cunners were observed at Little Minchins Cove between 1999 and 2000 during the day. Thirty-four cunners were observed at Little Minchins Cove in 1999, which was almost as high as the number observed at all non-kelp sites combined (50). In 2000, 338 cunners were observed from two transects in Little Minchins Cove in a bed of *L. longicruris* compared to 192 observations from 11 transects that had little *L. longicruris*. In 1999, the average number of cunners observed was highest near the reef but there were almost equally high numbers at 35 and 40 m on the reef transect. No cunners were observed from -40 to -15 m on the reef transect of Little Minchins Cove (Fig. 2.10) corresponding with a lack of *L. longicruris* for the first 20 m on this transect (Table 2.1). Instead of *L. longicruris*, there was a medium cobble or sand substrate in this area of the transect. The density of cunners observed was higher within the *L. longicruris* bed. In 2000, cunners were observed more frequently near the artificial reefs in Little Minchins Cove similar to non-kelp reef transects. However, the average number of cunners did not decrease as rapidly with distance from the reef at the Little Minchins site compared to non-kelp reef transects. Further, there was a peak in cunners numbers, almost as great as that observed at the reef, 20 m from the reef in Little Minchins Cove. The peaks in average numbers of cunners observed on the adjacent control transect of Little Minchins Cove at 20 and 25 m were not as high as those observed near the reef on the reef transect.

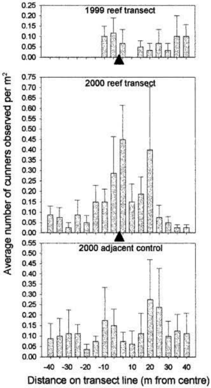

Fig. 2.10. Average numbers of cunners observed along the reef transect in 1999 and along the reef and adjacent control transects in 2000 at Little Minchins Cove. Error bars indicate one standard error from each mean. Abundance recorded at 0 - 5 m was plotted at the 5 m mark and so on. Black triangles indicate location of the artificial reefs.

The number of cunners observed was associated with water temperature. The 1999 study period was in the autumn and water temperature decreased throughout the surveys, from 10-12°C in late September to close to 5°C in late October. Fewer cunners were observed as temperature declined. No cunners were observed in late October. During two surveys on July 17, 2000 water temperature was 4-5°C and two cunners were observed to be inactive, resting on the bottom next to small rocks.

Atlantic cod and rock cod were observed more frequently on reef transects than control transects and were often near the reefs (Figs. 2.11). Twelve juvenile cod (*G. morhua* and *G. ogac*) were observed in 1999 and 17 were observed in 2000. Nine of those cod observed in 1999 and 16 observed in 2000 were over reef transects. No juvenile cod were observed on distant controls in 1999 or adjacent controls in 2000. In both years those cod observed on reef transects were concentrated near the reefs. Those observed on control transects showed no patterns of distribution. Seven Atlantic cod were identified in 1999, and only one was observed in 2000. Four of these Atlantic cod in 1999 and the one observed in 2000 were on reef transects and were concentrated near the reefs. The only juvenile Atlantic cod observed on control transects were dispersed over the transect (1999). Age 1 juvenile Atlantic cod were observed only on reef transects near the artificial reef. Age 2 and 3 Atlantic cod were observed on both reef and control transects (Fig. 2.12). No rock cod were identified in 1999; 11 were observed in 2000. All rock cod were observed on reef transects. The highest average numbers of rock cod observed were near the reefs. The average numbers of all *Gadus* spp. were low compared to average numbers of Atlantic cod and rock cod on the reef transects in 2000 (Fig. 2.11). This was because Atlantic cod, rock cod and unidentified *Gadus* species together were observed in more surveys than either Atlantic cod or rock cod separately. When the number of *Gadus* spp. observations was divided by the higher number of surveys, lower overall averages were the result. Due to this method of calculation, sample size influenced densities in the histograms. It was therefore not possible to make direct comments on fish densities. Instead it was only possible to discuss the relative frequency of fish observations.

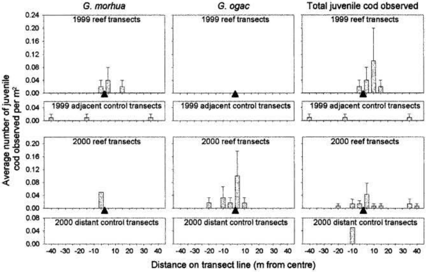

Fig. 2.11. Average number of juvenile cod observed along transects in 1999 (end Sept. - end Oct.) and 2000 (mid-July - late Sept.). Error bars indicate one standard error from each mean. Total juvenile cod include identified *G. morhua*, *G. ogac*, and *Gadus* sp. that were not identified to species. Abundance recorded at 0 - 5 m was plotted at the 5 m mark and so on. Black triangles indicate location of the artificial reefs.

There appeared to be seasonal differences in habitat usage among *Gadus* species between 1999 and 2000. The 1999 surveys were taken between the end of September and the end of October (autumn), while the 2000 survey data were collected from late July to the late September (summer). In the autumn of 1999 seven Atlantic cod were observed but only one was identified in the 2000 summer period. This Atlantic cod was observed during the last surveys in September, close to the autumn period. In contrast, no rock cod were identified during the autumn of 1999, while in the summer period of 2000 eleven rock cod were observed.

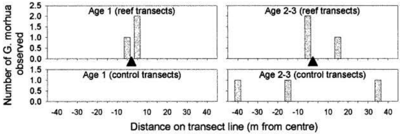

Fig. 2.12. Total number of age 1 and age 2-3 Atlantic cod observed along reef and all control transects in 1999 and 2000. Abundance recorded at 0 - 5 m was plotted at the 5 m mark and so on. Black triangles indicate location of artificial reefs.

Despite the presence of *L. longicruris* over the substrate in Little Minchins Cove, both Atlantic cod and rock cod observed had higher average numbers near the artificial reef similar to that observed at the other sites (Fig. 2.13). *Gadus* spp. observed on the reef transect in 1999 peaked in frequency near the reef. In 2000 the average numbers had a relatively lower peak near the reef and there was almost an equally high average number observed between -35 m and -25 m (Fig. 2.13). There were no Atlantic cod or rock cod on the adjacent control transect in 2000 and the only unidentified *Gadus* spp. observed were on the Little Minchins Cove adjacent control transect. There was a single peak for those observed on the adjacent control in 2000 at -40 m and the rest of those observed were dispersed (Fig. 2.13). An adjacent control transect was not deployed in

1999 so comparisons could not be made with non-kelp adjacent transects in that year. No Atlantic cod were identified at this site in 1999 and like the non-kelp reef transects, in 2000 only one Atlantic cod was observed over the reef transect near the reef in Little Minchins Cove. Several rock cod were observed at the kelp site in 1999 but none were observed at non-kelp sites. No rock cod were observed on control transects in Little Minchins Cove or the non-kelp sites in 2000 and rock cod observed over reef transects in Little Minchins Cove and non-kelp reef transects peaked in average numbers near the reef.

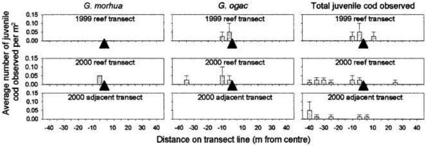

Fig. 2.13. Average numbers of juvenile cod observed along the reef transect in 1999 and reef and adjacent control transects in 2000 at Little Minchins Cove. Error bars indicate one standard error from each mean. Total juvenile cod include identified *G. morhua*, *G. ogac*, and *Gadus* sp. that were not identified to species. Abundance recorded at 0 - 5 m was plotted at the 5 m mark and so on. Black triangles indicate location of the artificial reefs.

Numbers of cunners were highest during the day and decreased dramatically at night (Fig. 2.14). Most cunners were concentrated near the artificial reefs during the day and low numbers of cunners were dispersed on the adjacent and distant control transects. Only two cunners were observed at night (Fig. 2.14). Cunner density decreased greatly at night relative to day at the kelp site, Little Minchins Cove (Fig. 2.15). During the day cunner observations were high and concentrated near the artificial reef. In contrast, they were dispersed on the adjacent control transect.

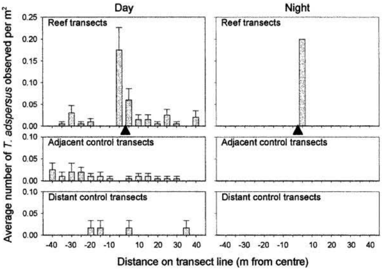

Fig. 2.14. Average numbers of cunners observed along non-kelp site transects during day and night surveys in 2000. Error bars indicate one standard error from each mean. Abundance recorded at 0 - 5 m was plotted at the 5 m mark and so on. Black triangle indicates the location of the artificial reef.

In contrast to cunners, there was a dramatic increase in the number of juvenile Atlantic cod observed during the night surveys of 2000 relative to the day surveys (Fig. 2.16). No juvenile Atlantic cod were observed during the day from late July to the end of August but 237 juvenile Atlantic cod ages 1-3 were identified during the three night survey periods over non-kelp transects. There was no obvious pattern to the distribution of Atlantic cod during the night and they showed no affinity for the artificial reefs. The number of rock cod observed did not noticeably increase at night relative to the day (Fig.

2.16). During the day, rock cod were observed only near artificial reefs. At night, rock cod density was similar to day but their distribution was dispersed, with no apparent affinity for reefs.

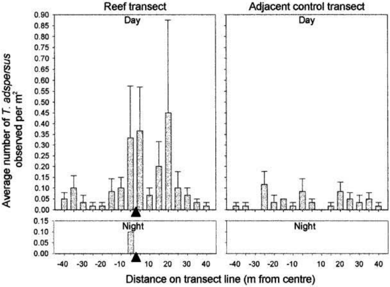

Fig. 2.15. Average numbers of cunners observed along transect lines during day and night surveys in 2000 at Little Minchins Cove. Error bars indicate one standard error from each mean. Abundance recorded at 0 - 5 m was plotted at the 5 m mark and so on. Black triangle indicates the location of the artificial reef.

In the kelp site, no Atlantic cod were observed during the day but density increased and dispersed at night over the reef and adjacent control transects (Fig. 2.17). There was a peak in the average number of rock cod near the artificial reef in the day, while none were observed on the adjacent control transect. At night the average number of rock cod observed did not increase greatly and rock cod were found on both the reef and control transects (Fig. 2.17).

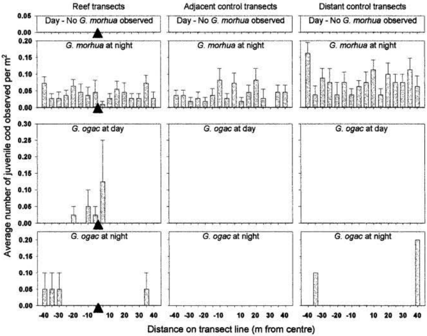

Fig. 2.16. Average numbers of Atlantic cod and rock cod observed along non-kelp site transects during day and night surveys in 2000. Error bars indicate one standard error from each mean. Abundance recorded at 0 - 5 m was plotted at the 5 m mark and so on. Black triangles indicate the location of the artificial reef.

The effects of reef removal on the average numbers of the three demersal species of interest were most obvious for cunners as observations decreased near the centre of the reef transect when the reefs were removed (Fig. 2.18). Due to the low numbers of Atlantic cod (4) and rock cod (6) observed it was difficult to determine the effect of the reef removals for these species. For the sites where the reef was present both before and after (controls), there was a peak in the average number of cunners at the reefs in the before period. The average number decreased in the after period with only one fish observed on reef transects. Only one cunner was observed before reef removal and none were observed after reef removal on the adjacent control transects. The overall decrease in cunners may correspond with the decrease in temperature at this time of year. The overall average number of cunners was higher on sites where reefs were removed (manipulations) relative to the sites where reefs were not removed (controls) (Fig. 2.18). Relatively high cunner numbers corresponded with *L. longicruris* present on, one of the removal sites (Heffems Cove East) (Table 2.1). The peak in cunner numbers near the centre of the reef transects disappeared after the reefs were removed.

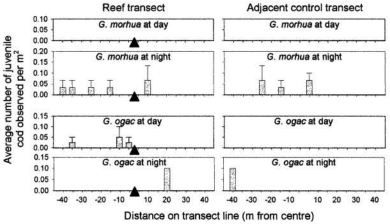

Fig. 2.17. Average numbers of juvenile cod observed along transects during day and night surveys in 2000 at Little Minchins Cove. Error bars indicate one standard error from each mean. Abundance recorded at 0 - 5 m was plotted at the 5 m mark and so on. Black triangle indicates the location of the artificial reef.

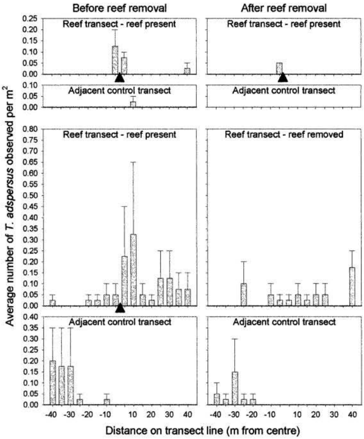

Fig. 2.18. Cunner abundances before and after removal of artificial reefs. Average numbers of cunners observed along transects. Error bars indicate one standard error from each mean. Abundance recorded at 0 - 5 m was plotted at the 5 m mark and so on. Black triangles indicate location of artificial reefs.

2.3.4 Scale of attenuation (SA) and rate of attenuation (RA)

Density of Atlantic cod, rock cod, and cunners attenuated to a "baseline" level 15 - 20 m from the artificial reefs. The SA for cunners was 15 m, as estimated by the exponential relation of fish density to distance within 15 m of the reef (Fig. 2.19). The exponential RA for this model was -1.08%/m to the 15 m SA. Ninety-five percent of the cunners occurred within 14.5 m of the reefs.

The SA for juvenile Atlantic cod was 15 m from reefs (Fig. 2.19). Atlantic cod density attenuated at a rate of -0.22%/m from the reefs to the 15 m SA. Ninety-five percent of these fish were within 14.5 m of the reefs. The SA estimate for rock cod was 20 m from the artificial reefs (Fig. 2.19). Juvenile rock cod density attenuated to baseline 20 m from reefs at a rate of -0.17%/m. Ninety-five percent of the juvenile rock cod occurred within 19.2 m of the reefs.

The RA for each demersal species from the reef at the kelp site was lower than non-kelp reef sites. In contrast to the non-kelp reef transects, the SA for cunners was 25 m at Little Minchins Cove, compared to 15 m at non-kelp reefs. The RA for the density of cunners was -0.56%/m to the 25 m SA, which was nearly half the rate found for non-kelp sites (-1.08%/m). Only one juvenile Atlantic cod was observed at Little Minchins Cove for both 1999 and 2000; insufficient data to make estimates. The estimate of the SA for rock cod was 15 m in Little Minchins Cove, compared to 20 m estimated for non-kelp reefs. Juvenile rock cod density attenuated at a rate of -0.18%/m to the 15 m SA. This rate was similar to that of non-kelp sites (-0.17%/m) to the SA of 20 m.

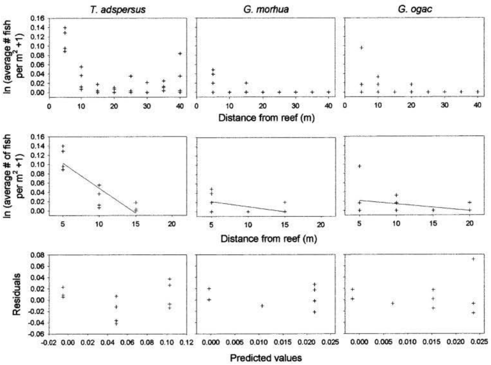

Fig. 2.19. Plots of log transformed average species abundances plus one with distance from the artificial reefs (top row), regressed log transformed average species abundances plus one against distance from the artificial reefs within scales of effect for each species (middle row) and respective plots of residuals against predicted values from linear regressions (bottom row).

2.4 Discussion

2.4.1 Increased production from artificial reefs

There are two hypotheses regarding the effect of artificial reefs on fishes, the production hypothesis and the attraction hypothesis (Bohnsack 1989). The production hypothesis states that artificial reefs provide additional critical habitat that can increase the carrying capacity and eventually the overall abundance and biomass of a species. In contrast, the attraction hypothesis states that reefs simply attract fish due to behavioural preferences and do not increase the overall biomass. Despite numerous studies on artificial reefs, the attraction-production controversy continues for several reasons (Bohnsack et al. 1997). One reason is that research needs to be conducted at larger temporal and spatial scales to distinguish the impacts of reefs from natural variability (Bohnsack et al. 1997). Another reason is that even if habitat is limiting, the amount of habitat added by artificial reefs in most areas is negligible compared to that already present in natural reefs (Bohnsack et al. 1997). Lindberg (1997) suggested that no single experiment will directly measure production at the level of breeding populations or fishery stocks. Therefore, production can only be tested by indirect methods. Studies have demonstrated that increasing habitat complexity with artificial reefs can increase local fish abundances (Bohnsack 1991, Grossman et al. 1997, this study) due to habitat or recruitment limitation in the natural environment (Grossman et al. 1997). However, it remains inconclusive whether increased biomass production on a large regional basis can be attained using artificial reefs (Bohnsack 1991, Grossman et al. 1997). Understanding the relative importance of attraction and production is critical for fisheries management

(Bohnsack 1989). This question must be answered to determine whether artificial reefs can enhance Atlantic cod production.

The idea that artificial reefs provide shelter from predation has been a suggested mechanism for increased production. However, the significance of increased survival on total stock sizes has not been determined (Bohnsack 1989). Experimentally demonstrating improved survival at artificial reefs would provide indirect evidence to support the production hypothesis (Bohnsack 1989). Barshaw and Spanier (1994) demonstrated that artificial reefs provided protection from predation on the Mediterranean slipper lobster, *Scyllarides latus* (Latreille). They tethered slipper lobsters inside and outside of an artificial reef and observed significantly higher mortality in lobsters tethered outside the reef compared to those tethered within the reef. Small fish may also gain protection from predation by association with artificial reefs (Mottet 1981). Associations with artificial reefs has been described for several demersal gadid species including pollock, *Pollachius pollachius* (Linnaeus); saithe, *Pollachius virens* (Linnaeus) (Wyche 1984, Smith et al. 1993, Sarno et al. 1994); and pouting, *Trisopterus luscus* (Linnaeus), (Fowler et al. 1999). Juvenile Atlantic cod appear able to assess refuge potential of available habitats and make a selection based on the safety these habitats offer (Gotceitas et al. 1995). Juvenile Atlantic cod abundance (Tupper & Boutilier 1995, Gotceitas et al. 1997) and survival (Linehan et al. 2001) are increased in high complexity areas. Juvenile Atlantic cod were observed in increased numbers near artificial reefs in my study. Association of juvenile Atlantic cod with artificial reefs may have been due to reduced predation risk near reefs, though this was not demonstrated in my study. However, if these juvenile cod select habitats that reduce risk of predation, (e.g. Gotceitas et al. 1997, Gregory & Anderson 1997, Linehan et al. 2001) then association with artificial reefs could increase survival of juvenile Atlantic cod and increase production of this species.

Juvenile Atlantic cod show high site fidelity and remain in the same general area for several months, maintaining a distribution of activity in a central area (Hawkins et al. 1980, Clark & Green 1990, Cote et al. 2001a). Juvenile Atlantic cod tagged in inshore waters of Scotland in summer remained there through the winter or returned to the same areas the following year (Hawkins et al. 1985). Juvenile Atlantic cod observed at my artificial reefs were likely of local origin and not just transient. If reefs provided protection from predators for juvenile Atlantic cod and increased their survival, there should be a net increase in the abundance of this species on a regional basis.

Increased habitat complexity, such as that offered by artificial reefs, may improve survival of juvenile Atlantic cod and thus increase Atlantic cod production. As habitat complexity increases, predator efficiency and risk of predation decreases (Cooper & Crowder 1979, Stein 1979, Coull & Wells 1983, Johns & Mann 1987, Russo 1987, Gotceitas & Colgan 1989, Nelson & Bonsdorff 1990, Mattila 1992, Gotceitas & Brown 1993, Gotceitas et al. 1995, 1997, Tupper & Boutilier 1995, 1997). Connell and Jones (1991) examined the effect of complexity on the survivorship of juvenile fish by manipulating substrate complexity. They increased complexity of simple habitats by adding boulders and reduced the complexity of heterogeneous habitats by removing boulders. They observed that mortality of juvenile *Forsterygion varium* (Whitley and Phillipps), a bennioid fish, was much greater in low complexity habitats. Low complexity treatments had 100% mortality while high complexity treatments experienced only 13% mortality. Coull and Wells (1983) suggested that a complexity threshold must be reached beyond which predation is significantly reduced. If prey are able to discriminate levels of habitat complexity, it should be expected that prey select habitats above the threshold level required to reduce predator foraging success (Gotceitas & Colgan 1989). These authors suggested that the threshold level of habitat complexity required to reduce foraging efficiency could be calculated. Gotceitas et al. (1997) suggested that there is a threshold level of complexity with stem density of eelgrass required to reduce predator efficiency on age 0 juvenile Atlantic cod. Gotceitas and Colgan (1989) suggested that determining threshold values should be useful in conservation or management programs concerned with habitat improvement or manipulation.

2.4.2 Artificial reef research

There are few artificial reef studies to date that test hypotheses and employ replicated manipulations with controls. Bohnsack and Sutherland (1985) reviewed 413 artificial reef references from various sources. They reported that artificial reefs were effective in attracting fish and were an important fishery management tool. However, most published papers were descriptive or were limited to discussion of reef construction. Conclusions were often based on little scientific evidence. Few studies employed quantitative experimental methods and many did not include valid controls, an observation supported by Lindberg (1997). Since the Bohnsack and Sutherland (1985) review I examined 445 abstracts referencing artificial reefs since 1986. Forty percent of these references were from workshops, symposiums, conferences, technical consultations, annual meetings, or proceedings from meetings. Many of the references were reviews of artificial reef programs in various US states and other countries or examined the feasibility of reef projects. Twenty-three percent of all the references related specifically to fish. Though some fish references examined seasonal or diel patterns, none quantified the scale that artificial reefs affect the density of particular fish species.

Gascon and Miller (1981) is the only artificial reef study conducted in Canadian waters which has been published in the scientific literature. They studied the structure of a temperate nearshore fish community in British Columbia. They examined patterns of fish community development, seasonal recruitment, and predictability on artificial reefs. In contrast to tropical areas, fish communities on their reefs formed temporally stable, predictable, and uniform communities. Individual fish showed limited home ranges with restricted movement within the vicinity of reefs. They did not quantify the change in habitat complexity resulting from artificial reef deployment, determine the scale of effect of reefs on each fish species, or examine diel patterns in fish density.

2.4.3 Day and night habitat associations

Juvenile Atlantic cod seek refuge from predators in areas of increased habitat complexity when predation risk is high (Linehan et al. 2001). Stein (1979) argued that organisms vulnerable to predation occupy habitats with greater complexity during periods when predators are more efficient. Being vigilant has energetic costs and organisms should decrease vigilance if it is unwarranted (Lima & Dill 1990). Small fishes should exhibit anti-predator behaviour to reduce their vulnerability without incurring undue vigilance costs (Linehan et al. 2001). It may be more efficient and possibly safer in the long-term for juvenile Atlantic cod to adopt anti-predator behaviour that requires less vigilance and less energy expenditure by appropriate substrate selection (Gregory &

Anderson 1997). The usage of refuge habitat tends to be reduced when predation risk is low (Walters & Juanes 1993).

Fish density around reefs changes due to daily activity patterns of fish (Mottet 1981). Associated with day and night shifts in activity are daily migrations for fish between safe resting areas and exposed feeding areas (Hixon 1991). Some fish have well-defined times when they leave to search for food, moving away from reefs at dawn and dusk to feed (Mottet 1981). Pouting were observed near artificial reefs during the day, but leave reefs after dusk and returned before dawn (Fowler et al. 1999). Fish density commonly increases over sandy areas at night when visual predators are less active (Helfman 1993). Similarly, diurnally active lake fishes tend to rest at night in relatively exposed areas (barren or sparsely vegetated bottoms or clearings) (Helfman 1993). Smith et al (1993) found that saithe were often more evenly distributed at night but were aggregated during the day. My study supports diel patterns of habitat usage for juvenile cod and cunners. In autumn 1999, juvenile Atlantic cod associated with artificial reefs during the day. At night juvenile Atlantic cod did not associate with complexity, but were instead dispersed over noncomplex substrate. This suggests that juvenile Atlantic cod may associate with complexity during the day when risk of predation to visual predators is highest. At night, when vision becomes limited and the efficiency of older conspecifics and other predators is reduced, juvenile Atlantic cod would not need to associate with complexity. Cote et al. (2001b) found that the density of juvenile Atlantic cod was high over complex substrates at dusk but their distributions shifted to more open areas such as sand or gravel at night. They suggested that cod migrate to low complexity habitats at night when visual predators are less efficient. I suspect that juvenile Atlantic cod associate with artificial reefs during the day because they provide shelter from predators but shortly after dusk they leave to forage. At night juvenile Atlantic cod disperse when predation risk is low returning to areas of high complexity at dawn.

Fish abundance in artificial reef habitats appears to be determined more by predation protection from increased complexity than increased food availability (Bohnsack 1991). Juvenile Atlantic cod may spend some of the day foraging in exposed areas and resting in refuge areas (Clark & Green 1990). Most fish spend their day foraging or avoiding predators and may separate the day into an active foraging phase and a relatively inactive phase linked to predator avoidance (Helfman 1993). Cote et al. (2001a) found that juvenile Atlantic cod were considerably more active in the day than at night, but showed high site fidelity. Stomach content analyses indicate that many fish species do not use artificial reefs for feeding (Mottet 1981). Pouting form non-feeding schools near artificial reefs during the day (Fowler et al. 1999). Artificial reefs do not have to provide food to attract fish but they would need to be in close proximity to foraging areas (Mottet 1981). Therefore, juvenile Atlantic cod may move between areas of increased habitat complexity during the day foraging periodically as they move within a local area. During these foraging movements juvenile Atlantic cod may school to reduce predation risk. This may explain the observations of occasional small schools (tens to hundreds) of age 1-2 juvenile Atlantic cod by Keats (1990) and schools of less than 10 age 2-3 cod during the day in my study.

The distribution and habitat use of juvenile Atlantic cod appears to change depending on the time of day and season. Several studies have found increased numbers of juvenile Atlantic cod in the shallow waters at night (Keats 1990, Methven & Bajdik

1994, Gibson et al. 1996, Grant & Brown 1998, Cote et al. 2001a, 2001b, Linehan et al. 2001). Juvenile Atlantic cod also showed increased density at night relative to the day in my study suggesting increased nocturnal activity. Pihl (1982) demonstrated that juvenile Atlantic cod migrated into a shallow water bay in Sweden at dusk and migrated back out at dawn. Gibson et al (1996) found that some fish species were more abundant in shallow water at night and suggested that this indicated an inshore migration at dusk and offshore migration at dawn. They found this pattern was most obvious for Atlantic cod; whiting, *Merlangius merlangus* (Linnaeus); poor cod, *Trisopterus minutus* (Linnaeus); and haddock, *Melanogrammus aeglefinus* (Linnaeus). They suggested this may be a characteristic behavioural pattern for juvenile Gadidae in general. Tagging studies of juvenile Atlantic cod in Newfoundland have concluded that cod migrate from deep cold water during the day to warm shallow water at night during the summer, but in the autumn move into shallow water and remain there throughout the day and night (Clark & Green 1990, 1991, Cote et al. 2001a). This seasonal pattern would explain the observations of several juvenile Atlantic cod during the day in the autumn of 1999 while none were observed until late September in 2000. Despite the fact that the autumn data (1999) and the summer data (2000) were collected in separate years, the results are consistent with seasonal patterns observed for this species (Clark & Green 1990, 1991, Cote et al. 2001a).

Temporal movements of juvenile Atlantic cod may help to optimize foraging and minimize predation risk. Grant and Brown (1998) found that age 1 Atlantic cod fed predominantly at night during the late summer through the autumn. Clark and Green (1990) suggested that the daily activity of juvenile Atlantic cod changes in September from nocturnal to diurnal behaviour. During this time they found that Atlantic cod actively feed over sandy areas during the day and move to rocky areas at night that might provide shelter from predation. Keats and Steele (1992) found that age 1 juvenile Atlantic cod feed throughout the day in the spring and summer but stomachs are most full in the early evening and there was little feeding at night in the summer. Alderstein and Wellman (2000) analyzed stomach contents of Atlantic cod, predominantly <40 cm, over a 24 h period and found a diel variation in weight of stomach contents. Findings suggested that Atlantic cod fed more intensely at dawn and dusk and that this diel pattern was fairly general for the North Sea in the spring and summer. Pihl (1982) suggested that the feeding activity by juvenile Atlantic cod at dawn and dusk seemed more pronounced in coastal populations and may be associated with their migration into shallow water areas. Clark and Levy (1988) predicted that the ratio of mortality risk to feeding time for juvenile sockeye salmon, *Oncorhynchus nerka* (Walbaum), was minimized at the intermediate light intensity levels of dawn and dusk which they called "antipredator windows". Their model was based on the assumption that variations in light intensity directly influenced vertical migrations of species that locate their prey by sight and was not an incidental cue to achieve bioenergetic benefits. Løkkeborg and Fernö (1999) observed that acoustically tagged adult Atlantic cod showed relatively low activity at night and suggested this pattern might be connected to their reduced ability to detect visual stimuli. Atlantic cod are primarily visual predators and use smell and taste when vision is limited (Brawn 1969). If Atlantic cod are primarily visual predators it is reasonable to suggest that they feed during daylight to maximize foraging efficiency. Juvenile cod are preyed upon by adult cod, which are also visual predators. Therefore, juveniles may forage opportunistically during the day when predation risk is high and increase foraging in the early evening when visual stimuli may reduce success of conspecific predators despite the limitation to their own visual stimuli. There would be little feeding by either juvenile or adult Atlantic cod at night when vision is restricted.

2.4.4 Seasonal habitat associations

Differences in habitat associations of juvenile Atlantic cod may be related to the time of year that studies are conducted. Gotceitas et al (1997) observed that juvenile Atlantic cod age 1-3 associated mainly with rock or macroalgae habitat but the depth and/or temperature influenced habitat use. Gregory and Anderson's (1997) study of juvenile Atlantic cod habitat was conducted in late April. Ambient water temperature from 25 to 75 m in their study area was -1.0°C, at which time age 1 juveniles were found over a gravel substrate. They believed that fish were in a 'winter mode'. The observation of comparatively fewer juvenile Atlantic cod in warmer, shallow water close to shore suggested these fish preferred the lower temperature conditions at the time of their study. Cote et al. (2001b) found that the abundance of age 1-2 Atlantic cod decreased in shallow waters in Newman Sound after mid-October and likely reflected migration to deeper winter habitats. My study was conducted during summer and autumn, which may explain the discrepancy between observations of habitat association of age 1 Atlantic cod with those of Gregory and Anderson (1997). Age 1 Atlantic cod in my study were observed near the artificial reefs. Fish observed by Gregory and Anderson (1997) were in winter mode and were moving slowly, which may represent a period of reduced predation risk.

Age 1 Atlantic cod in their study may have been less vigilant, less motivated, or slow to seek shelter.

Juvenile Atlantic cod have not been observed to make diel migrations in the winter or spring. Dalley and Anderson (1997) observed no diel variation in catch rates of juvenile Atlantic cod during December and January. This observation may have been due to the somewhat deeper areas where they sampled (minimum depth ~60 m). They suggested that diel behaviour in ages 0 and 1 Atlantic cod may be restricted to shallow inshore habitats and in the December to January period they may adopt a winter behaviour. Gregory and Anderson (1997) found no quantitative differences in activity or distribution and little change in frequency of Atlantic cod observations between night and day. They suggested that juvenile Atlantic cod did not undergo diel movements at spring water temperatures (~ -1.0°C).

2.4.5 Temperature effects on distribution

Changes in seasonal and temporal distributions of juvenile Atlantic cod may be linked to temperature. Wigley and Serchuk (1992) studied distributions of juvenile Atlantic cod <37 cm in the Georges Bank-Southern New England region and found that juvenile Atlantic cod changed depth locations seasonally to maintain preferred temperatures. Clark and Green (1990) suggested the diel migration of juvenile Atlantic cod was also driven by temperature. They stated that there is a thermocline during the summer months in shallow coastal Newfoundland waters that breaks down in mid September after which the water column becomes isothermal. They suggested that it would be energetically beneficial for these fish to move to deep colder waters during nonforaging hours while a thermocline existed because juvenile Atlantic cod are nocturnal during the summer months and are relatively inactive during the day. When the thermocline breaks down, juvenile Atlantic cod no longer benefit from shoreward migration and remain in shallow waters (Clark & Green 1990). In the laboratory, examination of temperature preferences of juvenile Atlantic cod indicated that fish held in heated water made choices similar to those held in ambient temperature seasonally (Clark & Green 1991). These authors maintained that seasonal variation in temperature preference was not due to changes in ambient temperature but occurred in an anticipatory manner, allowing juvenile Atlantic cod to maintain a physiologically optimum temperature at a seasonally appropriate level. However, they were unable to determine whether the seasonal changes in behavioural patterns were caused by endogenous annual cycles, external cues (e.g. water chemistry) or a change in day length. Tat'yankin (1974) suggested that seasonal differences in preferred temperatures in fish are not only dependent on the environmental temperature but depend on other seasonal factors. Clark and Green's (1991) temperature preference experiments supported the energetics hypothesis that shoreward migration is energetically efficient as long as there is thermal stratification but migration ceases as stratification breaks down. However, they did not test whether juvenile Atlantic cod showed diel temperature preferences, which would have given more support for their hypothesis.

Temperature-depth profile data and cod observations in my study in 2000 supported Clark and Green's (1990) hypothesis that diel shoreward migration of juvenile Atlantic cod ceases when the water column becomes isothermal. The first observations of juvenile Atlantic cod in 2000 were not until late September after the thermocline had broken down (mid-September). However, I could not determine whether the breakdown of the thermocline caused the cessation of the shoreward migrations of juvenile Atlantic cod, or if migrations ceased due to an environmental cue corresponding with the breakdown of the thermocline.

Several studies have shown evidence contrary to the energetics hypothesis proposed by Clark and Green (1990). Analysis of stomach contents of age 1 juvenile Atlantic cod captured during the late spring and summer showed there was continuous feeding throughout the day; gut contents were highest in the early evening, and there was little feeding at night, during the summer (Keats & Steele 1992). This suggests that the nocturnal migration of juvenile Atlantic cod into shallow waters was not for the purpose of feeding as Clark and Green (1991) had proposed. Grant and Brown (1998) obtained similar results for age 0 juvenile Atlantic cod but this pattern continued into the autumn. Also, Linehan et al. (2001) demonstrated that predation by juvenile Atlantic cod (13.0 - 28 cm SL) on tethered prey (age 0 *Gadus* spp.) was higher during the day than at night. Observations of shoreward migration of juvenile Atlantic cod by Cote et al. (2001a, 2001b) could not be explained by physiological energetics. Movements of juvenile Atlantic cod were all above the thermocline, which did not break down until after the tagged juvenile Atlantic cod had stopped shoreward migration in the autumn (Cote et al. 2001a, 2001b). Cote et al. (2001b) suggested that the diel movement of juvenile Atlantic cod might be due to increased feeding opportunities and/or reduced predation risk. My data supports the nocturnal shoreward migration of juvenile Atlantic cod while the thermocline exists, but neither supports nor contradicts the energetics hypothesis.

My study appears to represent the only multiyear study in cold marine waters on artificial reefs examining abiotic factors, which have been all but ignored (Sanders et al. 1985). Sanders et al. (1985) represents one of the few studies to examine the effects of long-term abiotic factors for artificial reefs in temperate waters. In their study, temperature seemed to be the most important factor influencing the density of fishes near reefs. However, fish density could have been influenced by visibility or light intensity. All three factors may have covaried and they were unable to conclude whether temperature itself was the most important factor in determining species diversity and density. I could not determine what motivated the seasonal distribution patterns of juvenile rock cod in my study but distribution patterns may be similar to that for juvenile Atlantic cod. Cunners in contrast appears to be more directly influenced by temperature itself. Bradbury and Green (1997) found that cunners go into torpor when the water temperature is around 5°C in Newfoundland waters, which would explain their absence during surveys in my study at the end of October 1999. Water temperature in late October 1999 was close to 5°C, as estimated from the thermometer on the diving console of P. Sargent.

2.4.6 Before and after reef removals

The effect of reef removal on the density of demersal species was most evident for cunners. The low number of observations of both juvenile *Gadus* species during the before and after reef removal portion of my study was insufficient for accurate interpretation. The removal of the artificial reefs resulted in a decrease in cunner density in the centre of the transects where the reefs had been present. Some of the decrease in cunner observations may have been due to a decline in water temperature between these survey periods. There appeared to be an overall decline in cunner numbers at all sites between the before and after reef removal surveys.

2.4.7 Scale of reef effect

Several studies have determined the distribution of increased fish abundance from artificial reefs by fishing at various distances from the reefs (Mottet 1981) but my study is the first to quantify the SA for individual species. Mottet (1981) defined the "enhanced fishing zone," as the distance within which fish attracted to an artificial reef can be caught and determined this to extend at least 100 m from the reef. The enhanced fishing zones are usually not circular because fish often congregate either upcurrent or downcurrent from the attracting reef (Mottet 1981). Mottet (1981) did not explain how this zone was determined. If the method of fishing employed bait, fish may have been attracted from distances greater than they normally tend to associate with the reefs. The estimation of the SA for each demersal species in this study was considerably less than 100 m (15 - 20 m). This may be due to the relatively small size of my artificial reefs. The demersal species in my study may show closer associations with complexity than those described by Mottet (1981), or because no bait was used for this estimation. The scale of effect for juvenile cod species in my study is comparable to that of another demersal gadid, saithe. Wyche (1984) observed that saithe spent 85% of their time schooling around the perimeter of an artificial reef. They swam up to 10 m out over the adjoining sandy substrate but never moved more than 20 m away from the reef. Fowler et al. (1999) observed that pouting distributed differently around artificial reefs depending on current speed. At current speeds >0.3 m/s, pouting congregated near the reefs, but when currents were <0.1 m/s pouting were more dispersed, were further from the reefs and were higher in the water column. It was not known how each species in my study distributed with the currents and how the transects were oriented in relation to the prevailing currents at artificial reef sites. The effects of currents on the distribution and habitat selection of nearshore Newfoundland demersal species should be given future consideration.

2.4.8 Substrate complexity

In artificial reef studies increased complexity is assumed and rarely quantified (Bohnsack 1991). Artificial and natural reefs are made up of different structural elements in different combinations and are thus difficult to quantify and most are impossible to compare (Bohnsack 1991). However, fractal dimensions are mathematical expressions of substrate ruggedness that are independent of the nature of the substrate (Gee & Warwick 1994). Fractal dimensions allow comparisons of complexity between different habitats (Gee & Warwick 1994). This study used both fractal dimensions and rugosity indices to quantify the complexity added by artificial reefs in relation to the surrounding substrate.

When substrate complexity was estimated by fractal dimensions and rugosity indices at the two largest step sizes over a 10 m section of the transects and the whole transect (80 m) there were no significant differences in complexity between reef, adjacent control and distant control transects. At the 10 m scale it was expected that artificial reefs would contribute greatly to the complexity of that section of transect. However, at the scale of the whole 80 m transect, as expected, the reef transects were not significantly different from the adjacent control and distant control transects. This was expected because the larger the scale of view, the smaller a single complex object becomes overall. Differences in complexity between types of transects may not have been detected because the comparisons were made using substrate complexity measure estimates based only on the two largest step sizes.

The fractal dimension (*D* = 1.16) calculated for the middle 10 m sections of reef transects in this study compares with the fractal dimension calculated by Bradbury et al. (1984) using their largest step sizes to determine the complexity of a coral reef. They used seven step sizes and found at the finest scale (10 cm) the fractal dimension of the natural reef was *D* = 1.1 but *D* changed to 1.05 over the next 4 larger scales (20, 50, 100, and 200 cm) and rose to about 1.15 for the two largest scales (5 and 10 m). They suggested that the ranges of scales coincided with that of the three different reef structures. The branches and convolutions of the individual structure of the coral colonies were in the range of 10 cm; the distribution of sizes of living adult coral colonies were in the range of 20-200 cm; and the biological structures such as spurs, grooves, and buttresses were in the range of 5-10 m. The artificial reefs in my study had complexity comparable with that of the geomorphological structures such as spurs, grooves, and buttresses on coral reefs.

The complexity of a habitat, as determined by its fractal dimension, may increase or decrease depending on the magnification or scale of measurement at which it is viewed (Bradbury et al. 1984, Gee & Warwick 1994). There was no apparent change in fractal dimensions within each type of transect estimated for the non-kelp sites over for the range of step sizes used in my study. The middle 10 m section of the reef transects was fractal (*D* = 1.16). Therefore, along these 10 m sections the distance measured changed as the step size changed. The adjacent control and distant control transects had fractal dimensions of 1.00 and 1.01, respectively. These transects were essentially Euclidean in that the distance measured within these 10 m sections of transect did not change when measured at different step sizes. No significant difference in complexity was detected between the kelp bed transects and their respective non-kelp transect types. However, the kelp itself was not directly measured. This was because the devices used to measure substrate complexity compressed the kelp reducing its natural complexity. If the kelp had been quantified by fractal dimensions and added to that of the underlying substrate there may have been a significant difference in complexity between kelp and non-kelp transects.

Complexity of the transects was also measured by vertical relief and substrate rugosity indices using the two largest step sizes. Rugosity and vertical relief are highly correlated on coral reefs and substrate rugosity is correlated with fish species richness (Luckhurst & Luckhurst 1978). However, for demersal fish vertical relief does not seem to be important (Mottet 1981). The profile (Bohnsack & Sutherland 1985) or horizontal spread (Grove & Sonu 1985) of an artificial reef may be more important than the height for demersal species.

2.5 Conclusions and recommendations

I have demonstrated that artificial reefs increase the local density of marine demersal species in the nearshore waters of Newfoundland. Artificial reefs significantly increased habitat complexity at small scales. Juvenile Atlantic cod and rock cod show increased numbers with habitat complexity such as created by artificial reefs. Juvenile Atlantic cod appear to use habitat complexity in shallow water (< 20 m) during the day in the autumn (late Sept. - end of Oct.). During the day, in the summer, they are in deeper water. They increase in density at night when predation risk is low and do not associate with complexity during this time. Juvenile rock cod appear to use habitat complexity in shallow water in the summer in the day but in the autumn their substrate preference seemed to change as they were observed in a kelp area. Juvenile rock cod did not increase greatly in number at night but like juvenile Atlantic cod dispersed showing no association with habitat complexity. There were differences in substrate usage seasonally between juvenile *Gadus* species but both used complexity during the day at some part of the year. Cunners also used habitat complexity provided by artificial reefs but unlike juvenile *Gadus* spp. showed no seasonal differences in habitat usage between summer and autumn. Cunners were influenced by water temperature. Cunners, unlike juvenile *Gadus* spp. are diurnal and become inactive at night.

The association of juvenile *Gadus* species with increased complexity in this study has been assumed to relate to their need to reduce predation risk and seek refuge. However, another explanation for the use of artificial reefs by these species may be increased food availability at these sites. Therefore, a critical question that must be considered with respect to the use of artificial reefs is what makes them attractive; food, shelter, or both.

Bohnsack and Sutherland (1985) made 29 recommendations for future artificial reef research, of which several were followed in my study. A carefully controlled experimental study was conducted to test several hypotheses with controls and replication. Quantitative data were collected and the complexity of the reefs was quantified. The depth of use, size of the reefs, reef design, composition, surrounding substrate and dates of deployment were described. The variation in usage of the artificial reefs with behaviour for individual species was examined. The use of artificial reefs for the purposes of improving recruitment was also examined.

Several recommendations of Bohnsack and Sutherland (1985) should be followed in future artificial reef research. Most artificial reef construction and research has been conducted in Japan and the United States but with very different approaches (Bohnsack & Sutherland 1985). Work in the United States has been directed towards harvesting existing resources while Japan is directed towards enhancing fishing grounds. Bohnsack and Sutherland (1985) recommended a technical knowledge transfer on artificial reefs from Japan. They suggested that many recommendations from Japan could be used as testable hypotheses in other areas of the world. The construction of artificial reefs is still more of an art than a science and future research should examine how to construct effective, economical, long lasting, and easily handled structures (Bohnsack & Sutherland 1985). The optimal reef size, design, density and configuration for a particular habitat should be determined. Studies suggest that artificial reef location, its surrounding habitat, and reef size are important factors and may be more important than habitat complexity in determining population abundance (Bohnsack 1991). In addition, the economics of long-term versus short-term strategies and the economics of building prefabricated reefs as opposed to using waste materials should be examined (Bohnsack & Sutherland 1985). Research should examine how to increase production of more desirable species, in this case juvenile Atlantic cod. Bohnsack and Sutherland (1985) recommended that the means by which reefs attract fish should be determined as it may lead to designs to increase attractiveness. Often fisheries managers believe that artificial reefs work but do not know how they work, even though understanding how they work is critical for proper management strategy (Bohnsack & Sutherland 1985). More studies should examine the effects of artificial reefs on the surrounding biota. Before artificial reefs are employed, a comprehensive plan for reef development and deployment should be established (Bohnsack & Sutherland 1985).

The density and substrate association of juvenile Atlantic cod are influenced by many factors including season, time of day, habitat complexity, predation risk, and age class. If artificial reefs are to be used to increase density of juvenile Atlantic cod they should be positioned to increase the chances of usage. The best positioning for these artificial reefs may depend on the size and structure of the reef used and on the depth and substrate on which it is placed. Juvenile Atlantic cod associate with artificial reefs in shallow water in the autumn, but it may be prudent to also examine whether juvenile Atlantic cod will use reefs at deeper depths during the summer.

The potential use of artificial reefs for the enhancement of commercial species in Canadian waters requires further research. The Japanese national budget for the construction of enhanced fishing grounds during a seven year period ending in 1982 was $326,000,000. This compares to little, if any, federal funding to date for artificial reef research in Canada.

Chapter 3 General Discussion

3.1 Ideal free distribution (IFD)

The IFD model predicts that organisms distribute between resource sites to match the distribution of resources (Kennedy and Gray 1993). Kennedy and Gray (1993) re-examined the model and found that the current form of IFD does not accurately depict the distribution of foraging animals, which may be explained by violation of assumptions of the model. The presence of predators is one of several factors that may affect the distribution of organisms (Kennedy & Gray 1993). Utne (1994) examined body length of perch, *Perca fluviatilis* (Linnaeus), to determine if it influenced the trade-off between utilizing feeding habitats and avoiding predation. She found that perch did not distribute between the feeding patches according to the IFD model and the presence of predatory pike, *Esox lucius* (Linnaeus), significantly influenced perch distribution. All perch reduced the proportion of visits to the risky feeding patch, with smaller perch making significantly fewer visits to the risky patch than larger ones. However, the smaller perch increased their number of visits to the risky patch when the food ratio was increased consistent with the predictions of IFD. Duren and Glass (1992) examined the IFD model for juvenile Atlantic cod with a food abundance ratio of 2:1 and found there were on average 2.5 times as many fish present in the rich patch compared to the poor patch. However, this was examined in a laboratory and influences of potential predators were excluded. If food availability is equal between reef and control transect, I would speculate that juvenile cod observed during my study distributed due to the effects of predator avoidance as opposed to the distribution of resources alone.

3.2 Size-specific effects of habitat complexity

Habitat complexity may have size-specific effects and may be perceived differently depending on the size of the organism observing it. A fractal dimension describes the complexity of an object, which relates to the way an animal living on this object perceives that habitat. For example, the circumference of a tree trunk will increase as the scale (step size) at which it is measured decreases. This will have consequences for the way the tree trunk appears to creatures of different sizes (Sugihara & May 1990). Gee and Warwick (1994) examined macrofauna and meiofauna in relation to the fractal dimensions of several species of macroalgae. They found that by equating the measurement scale (step size) with a measure of animal body size, the implications were that the same seaweed may have less habitat for large amphipods (macrofauna) and more habitat for small harpacticoid copepods (meiofauna). Similarly, Morse et al. (1985) studied fractal dimensions of vegetation in relation to density of arthropods. They proposed that by substituting an animal's body length for the step size (λ) in equation 1 (in Chapter 2) and assuming that animals see and use their environment in proportion to their own body length, their perspective of the environment can vary depending on their size. They suggested that for a homogeneous fractal surface with a fractal dimension of 1.5, a 3 mm long organism may perceive its environment to be an order of magnitude more extensive than organisms 30 mm perceive the same area. Beck (1998) suggested that most indices of habitat complexity other than fractal dimensions are unable to account for the potential size-specific effects of structural complexity on species. This indicates that fractal dimensions are biologically appropriate measures to compare results of artificial reef studies.

The idea of size-specific effects of habitat complexity and the way in which habitat complexity is perceived by an organism may be applied to the complexity of artificial reefs. The increase in habitat heterogeneity provided by artificial reefs results in greater density of fish (Guzmán et al. 1988). Artificial tire reefs may provide nursery areas for commercial species and could increase the survival of their juvenile stages (Guzmán et al. 1988). Chua and Chou (1994) compared tire pyramid and concrete module artificial reefs in Singapore to see if barren seafloor areas could be enhanced and made more productive. They found that concrete modules were more effective than tire pyramids at increasing fish density per unit volume but the tire pyramids, with smaller openings, restricted the size of fish that could shelter in them and so only juveniles and smaller adults were observed. These tire reefs excluded larger fish and provided more protection for smaller fish. A similar conclusion was reached by Higo et al. (1980). The complexity of these artificial tire reefs may appear high from the point of view of the smaller fish offering more available habitat, yet to larger fish it may appear much less complex when viewed at their proportional scale. Hole size is an important component of shelter quality and fish density varies with the abundance of holes that correspond with the size of the fish (Hixon & Beets 1989). Hixon and Beets (1989) also found a significant relationship between the density of large fish and the maximum number of small fish. As the number of large holes and the corresponding density of large fish increased, the density of small fish decreased. Therefore, to maximize abundance of juvenile Atlantic cod, artificial reefs should be constructed with hole sizes that correspond with the sizes of these fish.

3.3 Age-dependent habitat associations

Habitat association by juvenile Atlantic cod is age-dependent. Perry and Smith (1994) found that Atlantic cod did not consistently associate with certain depths in the spring or summer and they could not distinguish between temperature and salinity as the single factor influencing their distributions, possibly because of age-related effects. In laboratory studies, ages 0, 1 and 3 Atlantic cod tend to select different substrates (Gjøsæter 1987, Gotceitas & Brown 1993, Gotceitas et al. 1995, Fraser et al. 1996). Age 0 Atlantic cod preferred cobble to artificial kelp as a refuge from predation (Gotceitas et al. 1995). Predators could not get at juvenile Atlantic cod hidden in the interstitial spaces of cobble, while those hidden in the artificial kelp were accessible (Gotceitas et al. 1995). Therefore, cobble would appear to be a better refuge for age 0 Atlantic cod. Artificial kelp was only selected when cobble was not available and juvenile Atlantic cod seemed to associate with fleshy macroalgae primarily for cover from predators (Gotceitas et al. 1995). In the spring in Placentia Bay, Newfoundland, age 1 Atlantic cod were found predominantly over low relief gravel substrates (Gregory & Anderson 1997). In contrast, in the summer and autumn Gotceitas et al. (1997) observed that age 1 Atlantic cod concentrated in areas of coarse substrate with macroalgae. All age 1 Atlantic cod observed in my study were found in close association with the artificial reefs and were not observed over a sand substrate. This may indicate that these fish were being vigilant for predators. In the laboratory, Fraser et al. (1996) found that age 1 Atlantic cod preferred sand to gravel but sought refuge in the interstitial spaces of cobble when an age 3 Atlantic cod was present. They also showed that when age 0 and age 1 juveniles were together, age 0 Atlantic cod avoided age 1 Atlantic cod by selecting the more complex substrate. However, when both were presented with an age 3 Atlantic cod (conspecific predator), both selected similar habitat to avoid these predators. In the field, Gregory and Anderson (1997) found no indication that young juvenile Atlantic cod avoided areas occupied by older juveniles given that they observed young juveniles within 10 m of older juveniles. They found that age 2-3 juveniles associated with coarse substrate and high bathymetric relief and often individual substrate features, such as single boulders or crevices, and increased activity at increasing distance from such features. Gotceitas et al. (1997) found that age 1-3 Atlantic cod in Trinity Bay, Newfoundland associated mainly with a habitat of rocks or macroalgae but the depth and/or temperature at which this habitat occurred influenced its use by these fish. In the laboratory, Gotceitas et al. (1995) examined habitat selection of age 0 Atlantic cod in the presence of passive and active age 3 Atlantic cod predators. They found that the age 3 Atlantic cod avoided artificial macroalgae when foraging but associated with it when they were passive. In the laboratory, age 3 Atlantic cod (the predator in the study) showed no consistent preference between sand and gravel, but preferred sand when presented with sand and cobble (Fraser et al. 1996). In all of the laboratory studies the younger juveniles were prey while the age 3 juveniles were predators. No study tested habitat selection when all these age classes are potential prey. An age 3 Atlantic cod might behave as age 0 and 1 Atlantic cod under threat of predation. During my study age 2-3 juvenile Atlantic cod were observed in schools together on several occasions in Newman Sound. Age 2-3 Atlantic cod in my study were found concentrated near the artificial reefs. Because juvenile Atlantic cod are capable of identifying predation risk and respond in ways that improve their survival

(Gotceitas et al. 1995, Fraser et al. 1996), differences in habitat associations in the field may be a result of this assessment of predation risk.

Juvenile Atlantic cod may only use a kelp substrate occasionally as refuge. Keats et al. (1987) observed that when juvenile Atlantic cod ages 1-2 moved into shallow inshore waters in Conception Bay, Newfoundland in the spring they associated with beds of the macroalgae, *Desmarestia* spp., and were rarely observed in the adjacent barren areas. Similarly, I have observed juvenile Atlantic cod ages 0-1 associate with *Desmarestia* spp. growing on sand substrate in Conception Bay and Newman Sound, and growing on bedrock in Torbay, Newfoundland. I have also observed age 1-2 Atlantic cod associate with *Alaria esculenta* (Linnaeus) in Logy Bay, Newfoundland. Keats et al. (1987) occasionally observed juvenile Atlantic cod near large boulders covered with large macroalgae, around patches of ephemeral macroalgaes, and near large crevices and boulder patches. In contradiction to this, Keats (1990) found a lack of association of age 1-2 juvenile Atlantic cod with algal beds during the day in Newman Sound. In Placentia Bay, Newfoundland Gregory and Anderson (1997) also found that juvenile Atlantic cod did not show a preference for macroalgae cover. In Newman Sound, Cote et al. (2001b) observed that juvenile Atlantic cod associate with the kelp, which they had identified as *Agarum cribosum*, (Bory), more frequently during the day compared to the night. Data from the artificial reef in a bed of *L. longicruris* showed that juvenile Atlantic cod were rarely observed in this kelp. When they were observed, they associated more closely with the artificial reef than the surrounding *L. longicruris*. The reef may have offered more suitable habitat or higher complexity than that of *L. longicruris* for juvenile Atlantic cod ages 1-3.

Substrate type, threat of predation, and the presence of conspecific Atlantic cod all influence habitat use by juvenile Atlantic cod (Fraser et al. 1996). Juvenile Atlantic cod associate with structurally complex habitats as a means of reducing risk to predation. The ability of juvenile Atlantic cod to identify predation risk and respond in a manner that increases their chances of survival has been demonstrated (Fraser et al. 1996). Predators not only influence diel patterns of foraging of their prey, but may also reduce food consumption, in turn reducing growth, fecundity, and fitness (Stein 1979). The intensity of these patterns seems related to vulnerability. Prey should shift their microdistribution or exhibit escape behaviour when the potential for attack from a predator is high because time spent hiding is time lost for activities such as foraging (Stein 1979). Consequently, association of juvenile Atlantic cod with artificial reefs may indicate a threat of predation. Observed differences in patterns of activity of age groups of Atlantic cod suggested a difference in predator avoidance behaviour (Gregory & Anderson 1997). Gregory and Anderson (1997) observed that age 1 Atlantic cod are more mottled in colour and seemed to rely more on crypsis, while the older uniform-coloured juvenile Atlantic cod associated with habitat complexity. In my study, juvenile Atlantic cod (age 1-3) were most commonly observed in association with increased habitat complexity of the artificial reefs but were on occasion observed over sand or light cobble substrates. Observations of juvenile Atlantic cod in association with the artificial reefs may have been due to a predation threat. The observation that age 1 Atlantic cod were observed only near the artificial reefs may suggest that age 1 Atlantic cod are more vulnerable to predation and are more likely to associate with complexity. Age 2-3 Atlantic cod are larger and may be less vulnerable to predation and may have weaker associations with habitat complexity.

This idea is consistent with observations for another gadid species, the pouting. Fowler et al. (1999) observed that 86% of the pouting that associated with artificial reefs were age 0, followed by age 1 (7%), age 2 (5%) and age 3 (2%).

3.4 Age-dependent temporal distribution patterns

Seasonal and temporal differences in distribution patterns of juvenile Atlantic cod depend on age class. Organisms select habitats that promote growth and reproduction and reduce the risk of mortality, but habitat requirements change spatially and temporally as individual requirements change so that habitat suitable for one life stage may not suit another (Gibson et al. 1996). The centre of the distribution of age 0 Atlantic cod in Newfoundland coastal waters in the autumn during the 1990s was at bottom depths of 4-7 m (Methven & Schneider 1998). This environment has lower predation risk than deeper waters (Linehan et al. 2001). During the winter, age 0 juvenile Atlantic cod were almost exclusively found inshore in the large coastal bays, age 1 were dispersed onto shelf areas at depths up to 400 m, age 2 Atlantic cod were more frequently inshore and age 3 were predominantly offshore (Dalley & Anderson 1997). Wigley and Serchuk (1992) also found a separation in depth distribution with age classes for part of the year. Age 1 and 2 Atlantic cod were found at significantly shallower depths than age 3 and 4 Atlantic cod and the mean depths of age 3 and 4 Atlantic cod were also significantly different. However, in autumn surveys (late Sept. - late Oct.) age 1-3 Atlantic cod were observed at similar depths but were still significantly different from that of age 4. The authors suggested that the co-occurrence of age 1-3 fish might relate to diet and Atlantic cod may change depth locations to maintain preferred temperatures. However, seasonal shifts in concentration did not suggest a major change in substrate preference by Atlantic cod. Tat'yankin (1974) suggested that the age related differences in distribution may be a result of physiological changes in the growing organism. A result of these changes is a corresponding change in habitat requirements of the organism. In contrast to juvenile Atlantic cod, rock cod show no differences in depth distribution with age class. Morin et al. (1991) found that the sizes of rock cod caught in coastal and offshore waters in James Bay were similar, suggesting no size-based distribution differences. However, this has yet to be determined in coastal waters of Newfoundland.

3.5 Species-dependent habitat associations

Juvenile gadid species associated with habitat complexity during the day throughout the summer and autumn periods, but juvenile Atlantic cod and rock cod did not share common habitat during these periods. Juvenile rock cod, like Atlantic cod, appeared to have seasonal differences in their association with habitat complexity. Rock cod were observed near artificial reefs during the summer in 2000 but not in the autumn of 1999. In contrast, juvenile Atlantic cod were observed near artificial reefs in the autumn of 1999 but not in the summer of 2000. The exception to this observation was that rock cod were observed near the reef in the bed of *L. longicruris* in the autumn of 1999 and the summer of 2000. Only one juvenile Atlantic cod was observed in this kelp bed site and it was near the reef. This suggests that juvenile rock cod are more likely to associate with *L. longicruris* than juvenile Atlantic cod. Further, the common association of age 0 juveniles of both these species with eelgrass, *Zostera marina*, in shallow Newfoundland waters (Ings et al. manuscript, personal observation) changes as they get older. Thus there may be segregation in habitat preferences between gadid species at different times of the year. Both species use habitat complexity, but the specific cue that changed each species' habitat association on a diel and seasonal basis was not determined. During the day, in the summer juvenile Atlantic cod moved to deeper water so that habitat complexity in shallow water was available to juvenile rock cod. However, when Atlantic cod move into shallow water in the autumn, Atlantic cod may outcompete juvenile rock cod for shelter, which would force rock cod to seek shelter elsewhere. Juvenile rock cod might also change their habitat association seasonally by the same cues used by Atlantic cod to avoid competition for habitat. The seasonal distribution of rock cod appears to be related to temperature (Morin et al. 1991). Thus, the seasonal change in habitat association of rock cod observed in my study may be due to temperature.

During the night, juveniles of both *Gadus* species disperse and show no preference for habitat complexity. Unlike juvenile Atlantic cod, the number of juvenile rock cod observed did not increase greatly at night relative to the day. In contrast to this finding, Morin et al. (1991) found in the summer in James Bay that the number of rock cod caught in baited traps at night in shallow waters was significantly greater than during the day. This pattern suggests that rock cod feed during the night. However, like juvenile Atlantic cod, this pattern may be influenced by season. Mikhail and Welch (1989) found in Hudson Bay that there were no significant differences in catches of rock cod with depth during the day in the spring and suggested that rock cod were relatively sedentary. Data from my study indicates that rock cod do not undergo a diel shoreward migration like that observed for juvenile Atlantic cod in Newfoundland waters. Juvenile rock cod may simply aggregate near habitat complexity during the day and disperse at night.

However, if these two gadid species have similar metabolisms, one would expect that if Clark and Green's (1990, 1991) energetics hypothesis for juvenile Atlantic cod was correct then it would be energetically advantageous for juvenile rock cod to migrate to colder, deeper water during the day, especially if they feed at night.

My study found that cunners associated with habitat complexity during the day. Tupper and Boutilier (1997) also found a positive correlation between substrate rugosity (i.e. complexity) and cunner density. The number of cunners in my study was influenced by habitat complexity even more than juvenile *Gadus* spp. Cunners were more commonly associated with habitat of lower complexity than juvenile *Gadus* species. Cunners were often observed near kelp or large cobble whereas juvenile Atlantic cod were rarely observed in these areas. I did not determine whether this association of cunners was due to increased shelter or availability of food provided by this complexity. Cunners may be more motivated to associate with complexity for increased availability of food given that predation on adult cunners in Newfoundland is low (Bradbury & Green 1997, J. M. Green, personal comm.). Cunners showed no fear of divers during surveys and readily approached them during my study. Another explanation for their association with habitat complexity may be that there is reduction in energy expenditure associated with this behaviour (Bradbury & Green 1997).

Cunners associated with lower complexity than that of juvenile *Gadus* species. Some cunners were observed at the ends of the reef transects in both years. The association of the cunners with the first 30 m of the adjacent control transects can be explained by the presence of large cobble at the Heffems Cove East adjacent control transect in the first 30 m. This region of the transect was noticeably more complex than the other control transects, as confirmed by a statistical comparison of complexity measured by fractal dimensions. *L. longicruris* was present near the other end of the reef transect at the Heffems Cove East site and cunners may have associated with the presence of this kelp. In 2000, *L. longicruris* had spread over the Heffems Cove East site, and the observed distribution of cunners corresponded to this kelp distribution, suggesting that they associate with *L. longicruris*. Cunners were not observed on almost half of the Little Minchins Cove reef transect in 1999. There was no *L. longicruris* for the first 20 m of this transect after which the density of cunners increased in correspondence with increased kelp density.

Cunners did not change habitat association patterns between summer and autumn periods, but their behaviour was influenced by temperature. During surveys on July 17, 2000 water temperature was 4-5°C and cunners were sometimes observed to be inactive, resting on the bottom next to small rocks on low complexity substrate. During later surveys, when water temperature was warmer, cunners were more active and associated with higher substrate complexity including the artificial reefs. The density of cunners declined in correspondence with the decreasing temperature in autumn 1999, and no cunners were observed during the final surveys in late October when the water temperature was close to 5°C. Cunners are inactive when water temperature was below 5°C (Bradbury & Green 1997). In autumn, cunners go into torpor where they become dormant and stop feeding. They overwinter in the same inshore waters where they are observed throughout the year, beneath rocks or in deep rock crevices (Dew 1976). In Newfoundland waters cunners may remain in torpor for six months or more (Bradbury & Green 1997). Cunners showed strong associations with habitat complexity in 1999 and

2000 though corresponding peaks in numbers with complexity were more pronounced in 2000. This may be due in part to the fact that there were four surveys in 2000 as opposed to three in 1999. However, temperature was likely a contributing factor in the differences between these study periods. There were no cunners observed in the final survey of 1999 when the temperature was close to 5°C. Thus, cunners were only observed in two surveys in 1999 while in 2000 they were observed during all four surveys. This difference in survey numbers in which cunners were observed may account for the fact that there were almost four times as many cunners observed in 2000 (192) as in 1999 (50).

Cunners show a diel pattern of density opposite to that of juvenile Atlantic cod. The number of cunners observed at night decreased dramatically in my study. Using scuba transect surveys Methven et al. (2001) also observed that cunners were more abundant during the day than at night. Cunners are a diurnal species and are nocturnally quiescent (Bradbury & Green 1997). The only cunners observed in my study at night were non-active within the artificial reefs.

3.6 Effects of artificial reefs on other species

Density of organisms other than Atlantic cod, rock cod, and cunners increased with habitat manipulations in my study. Two American lobsters, *Homarus americanus* (Milne Edwards), three toad crabs, *Hyas araneus* (Linnaeus), several radiated shannies, *Ulvaria subbifurcata* (Storer), several age 0 *Gadus* spp., a sculpin, *Myoxocephalus* sp., a sea raven, *Hemitripterus americanus* (Gmelin), and a pout, *Lycodes* sp., were observed within, on top of, or in depressions beneath cinder blocks used to anchor the artificial reefs and mark the ends of the transects. Another 15 toad crabs were observed clinging to the sides or top of artificial reefs. The rope attaching together the surface buoy and cinder block at the end of transects also affected species density. Several age 0 *Gadus* spp. and at least two female lumpfish, *Cyclopterus lumpus* (Linnaeus), were observed hovering near the rope at least 7 m from the bottom. These latter observations may have been due to foraging behaviour of these fish feeding on various organisms growing on the rope near the surface. Increased habitat complexity affected invertebrate and fish distributions.

3.7 Reef surveying methods

Scuba visual surveys were employed during my study because they allowed non-destructive behavioural observations of demersal species in association with their habitat, free from the effects of baiting. Two methods of sampling have been commonly employed to assess fish assemblages at artificial reefs; fishing (destructive), and visual censuses by diving or video recording (non-destructive) (Charbonnel et al. 1995). Willis et al. (2000) compared underwater visual surveys, angling, and baited underwater video methods of measuring relative density of fishes in a New Zealand marine reserve. They found that the visual surveys gave the least reliable measure of density for one fish species, because of the extreme variability in responses of these fish to divers. The other methods employed bait, which may have attracted fish from distances greater than that surveyed by divers. They suggested that non-baited cameras may be less intrusive but required more field time and required more expensive equipment to provide testable results. Mottet (1981) suggested that fixed cameras and diver observations are limited in the area that can be covered and recommended echo sounders to determine how fish schools distribute around the reefs. However, this method does not provide information for individual species. Direct visual censuses allow recording of behaviour in association with the artificial reef and causes little disturbance so that replication is possible (Keats 1990, Charbonnel et al. 1995). Gregory and Anderson (1997) suggested that using scuba in nearshore studies is advantageous in describing associations of individual fish with habitat. Video recording was not used in my study, as it would have been too difficult to record the data required to estimate rates of attenuation of each species from the reefs. Keats (1990) found no evidence of attraction or repulsion of juvenile Atlantic cod during scuba surveys. Cote et al. (2001b) found that juvenile Atlantic cod could be approached by divers close enough to count, but would retreat as the diver advanced. Similarly, Wyche (1984) observed that saithe allowed divers to approach within 2 m and were undisturbed if divers made no sudden movements. Cote et al. (2001b) suggested that diver avoidance may have influenced the observed trends in diel abundance and habitat density of juvenile Atlantic cod. Juvenile cod are rarely observed during the day over substrates such as sand or gravel (Cote et al. 2001b). They suggested these fish may have inhabited these areas but dispersed before they could be detected by divers. Juvenile Atlantic cod may be sensitive to sound, especially in open areas if they are vigilant to the threat of predation. Small schools of juveniles had been observed during my study in Newman Sound as divers descended but were rarely seen while swimming along the bottom over non-complex substrates. Juvenile Atlantic cod ages 0-1 observed over macroalgal substrates during the day in Conception Bay and Logy Bay, NF allowed divers within close proximity (2-3 m), and closer proximity (1-2 m) when divers held their breath (personal observation). Exhaling produced noise from the release of bubbles and caused juvenile cod to retreat to 2-3 m distance from divers. Juvenile Atlantic cod may avoid sources of loud sounds when over a substrate that offers little refuge. I would recommend that future studies might use rebreathers for scuba surveys as they reduce the amount of air bubbles released and are thus quieter and would cause less disturbance to sound sensitive species thereby increasing the accurateness of fish surveys. Alternatively echo sounders could be used over artificial reefs accompanied with the use of an underwater camera to determine what species are present and in what proportions.

3.8 Reef construction

Materials that would otherwise be discarded have potential to enhance the marine environment and reduce the problem of disposal on land (Mottet 1981). Artificial reefs of old tires appear to be a cost-effective method to improve fishing and dispose of tires (Jensen et al. 1980). However there is concern that adding manmade structures to the natural marine environment may increase pollution as these materials break down. Leaching of chemicals from reefs are a concern (Kim 2001). One must consider the materials being used to construct artificial reefs because toxic pollutants may be introduced into the food chain (Mottet 1981). The problem with scrap materials is their durability. Old cars and household appliances have a short life (6 to 10 years) in seawater and add undesirable metals to the marine environment as they break down (Jensen et al. 1980). However, materials such as concrete rubble and tires have greater potential (Mottet 1981). Old car tires were used in the construction of the artificial reefs of this study, which do not disintegrate in the marine environment and provide a good substrate for the growth of algae and invertebrates (Fast & Pagan 1974). However, the reefs in my study also had rebar frames, which showed considerable rusting even during the period of this study. For long-term artificial reef projects it would be better to use materials which are less prone to corrosion. Another challenge in using these materials is developing methods to create piles high enough to be attractive to fish yet resist dispersal by storms (Mottet 1981). Also, despite the fact that many scrap materials are free, the labour and costs of reef construction may make it less costly in the long-term to use materials that are more durable that can be manufactured into the most effective reef designs (Mottet 1981). Mottet (1981) suggested fibreglass reinforced plastic (FRP) as possible material for reef construction because it is strong, non-toxic and does not corrode in salt water. Another advantage of FRP is that it can be used to make all components so the entire structure has the same durability. FRP is relatively expensive, but its strength allows construction with little material (Mottet 1981).

Optimal reef size is another important consideration for artificial reef construction. It is most efficient to construct a reef with an optimal size that maximizes the production per unit volume of the reef (Ogawa et al. 1977). This optimal size will vary depending on the model employed but the models analysed by Ogawa et al. (1977) suggested the optimal size for an artificial reef was 2000 m^3. However, for practical purposes they suggested it is important to realize the relation between production per unit volume of reefs and the size of the reefs for each area where the reefs are to be distributed.

3.9 Reef spacing

The spacing of the reefs is another important aspect of placement that must be determined in order to maximize the density of juvenile Atlantic cod and minimize costs of artificial reef construction and deployment. The distance within the SA estimated for each demersal species where 95% of observations of each species were expected was calculated. This distance may be used to determine the optimal spacing of similar artificial reefs. For juvenile Atlantic cod this distance was found to be approximately 15 m. The optimal spacing of these reefs would then be twice this distance between two such reefs so that the SA has minimal overlap. These particular reefs should therefore be spaced 30 m from one another to optimize the number of juvenile Atlantic cod present and reduce production costs of artificial reefs. For juvenile rock cod the calculated integral for 95% of the observations was around 20 m so artificial reefs should be spaced 40 m apart for this species and for cunners reefs optimal spacing would be 30 m. These distances would change depending on the reef design and the substrate on which they are placed. This was indicated because the SA for cunners at the reef in the kelp bed was considerably larger than that calculated for non-kelp sites. This study was the first to calculate optimal spacing distances for artificial reefs.

If artificial reefs are to be used to enhance the production of a species it would seem wise to establish a marine reserve with restricted fishing in the area in which they are used. Guzmán et al. (1988) strongly recommended that artificial reefs be used for conservation of tropical marine resources subject to heavy exploitation, and/or poor management. Many artificial reef studies have discussed their results in relation to using them for fishing grounds (Dewees & Gotshall 1974, Parker et al. 1974, Jensen et al. 1980, Laufle & Pauley 1985, Sanders et al. 1985, Hixon & Beets 1989, Chua & Chou 1994, Gregg 1995). Using artificial reefs as fishing grounds would be inappropriate for some species because artificial reefs may aggregate the dwindling remnants of a stock (Grossman et al. 1997). Using these reef areas as a fishing ground may further the destruction of the species (Grossman et al. 1997). Pitcher et al. (2000) employed quasi-spatial ecosystem simulations, using a modified ECOSIM method to predict benefits from artificial reefs (AR) within a marine protected area (MPA) in the South China Sea of Hong Kong harbour using catch and biomass data by species and fishery sector. They predicted that despite the increased fishing power of the Hong Kong fleet, a 10-20% MPA/AR system would produce significant benefits within 10 years. They cautioned however, that species that are highly migratory and have increased exchange rates between MPA and non-MPA areas will have reduced benefits from the MPA since those fish that migrate outside the MPA can be caught. Guénette et al. (2000) examined the potential use of marine reserves in the management of the northern (Atlantic) cod fishery in Newfoundland using a modelling approach. They determined that even if 80% of the fishing grounds had been marine reserves, the stocks could not have been rebuilt using the reserves alone unless they had been accompanied with reduced fishing effort in the 1980s. They suggested that moderate sized reserves (20%) accompanied with temporal closures to trawls and gill nets could have prevented the stock collapse. Fish migration is another factor limiting the success of marine reserves for Atlantic cod. If one were to consider the use of an MPA/AR system for juvenile Atlantic cod their temporal migrations must be taken into consideration. It might be advised to place ARs in several areas known to be frequented by juvenile cod at different times of the year. MPAs could be instituted for those reefs during the time of year that they are in use by this species. Pitcher et al. (2000) made four design recommendations for the MPA/AR system in Hong Kong. These were (1) permanent no-take commercial fishing zones with a no-anchor zone in the immediate AR area; (2) no-take corridors between ARs to allow fish migration among ARs; (3) a fished AR to encourage support from the fishers' community, provide a contrast in data and controls for monitoring; and (4) a managed buffer zone with rotating fishing sectors.

References

Alderstein, S. A. and H. C. Welleman. 2000. Diel variation of stomach contents of North Sea cod (*Gadus morhua*) during a 24-h fishing survey: an analysis using generalized additive models. Can. J. Fish. Aquat. Sci. **57**: 2363-2367.

Anon. 1993. Artificial reef tested in beach erosion control project. Public Works. **124**: 34-35.

Artificial Reefs Inc. 2001. Fish Haven. http://www.artificialreefs.com/fullscreen-launch.htm.

Barrett, R. T., N. Røv, J. Loen, and W. A. Montevecchi. 1990. Diets of shags *Phalacrocorax aristotelis* and cormorants *P. carbo* in Norway and possible implications for gadoid stock recruitment. Mar. Ecol. Prog. Ser. **66**: 205-218.

Barshaw, D. E. and E. Spanier. 1994. Anti-predator behaviours of the Mediterranean slipper lobster, *Scyllarides latus*. Bull. Mar. Sci. **55**: 375-382.

Beck, M. W. 1998. Comparison of the measurement and effects of habitat structure on gastropods in rocky intertidal and mangrove habitats. Mar. Ecol. Prog. Ser. **169**: 165-178.

Bogstad, B., G. R. Lilly, S. Mehl, O. K. Pálsson, and G. Stefánsson. 1994. Cannibalism and year-class strength in Atlantic cod (*Gadus morhua* L.) in Arcto-boreal ecosystems (Barents Sea, Iceland, and eastern Newfoundland) ICES Mar. Sci. Symp. **198**: 576-599.

Bohnsack, J. A. 1991. Habitat structure and the design of artificial reefs. pp. 413 - 426. *In* Bell, S. S., E. D. McCoy, and H. R. Mushinsky (eds.). 1991. Habitat structure: The physical arrangement of objects in space. Chapman and Hall, New York. 438p.

Bohnsack, J. A. and D. L. Sutherland. 1985. Artificial reef research: A review with recommendations for future priorities. Bull. Mar. Sci. **37**: 11-39.

Bohnsack, J. A., A-M. Eckland, and A. M. Szmant. 1997. Artificial reef research: Is there more than the attraction-production issue? Fisheries **22**: 14-16.

Bowen, W. D., J. W. Lawson, and B. Beck. 1993. Seasonal and geographic variation in the species composition and size of prey consumed by grey seals (*Halichoerus grypus*) on the Scotian Shelf. Can. J. Fish. Aquat. Sci. **50**: 1768-1778.

Bradbury, C. and J. M. Green. 1997. Daily and seasonal activity patterns of female cunner, *Tautogolabrus adspersus* (Labridae), in Newfoundland. Fish. Bull. **95**: 646-652.

Bradbury, R. H., R. E. Reichelt, and D. G. Green. 1984. Fractals in ecology: methods and interpretation. Mar. Ecol. Prog. Ser. **14**: 295-296.

Bradford, M. J. 1992. Precision of recruitment predictions from early life stages of marine fishes. Fish. Bull. **90**: 439-453.

Brawn, V. M. 1969. Feeding behaviour of cod (*Gadus morhua*). J. Fish. Res. Bd. Canada. **26**: 583-596.

Brock, R. E. 1994. Beyond fisheries enhancement: Artificial reefs and ecotourism. Fifth International Conference on Aquatic Habitat Enhancement. Bull. Mar. Sci. **55**: 1181-1188.

Brown, R. G. B. and D. N. Nettleship. 1984. Capelin and seabirds in the northwest Atlantic. pp. 184-194. *In* Nettleship D. N., G. A. Sanger, and P.F. Springer. (eds.) 1984. Marine birds: their feeding ecology and commercial fisheries relationships. Can. Wildl. Serv. Spec. Publ. 220 p.

Buckley, R. M. and G. J. Hueckel. 1985. Biological processes and ecological development on an artificial reef in Puget Sound, Washington. Bull. Mar. Sci. **37**: 50-69.

Bugrov, Y. Y. 1994. Fish-farming cages and artificial reefs: Complex for waste technology. Fifth International Conference on Aquatic Habitat Enhancement. Bull. Mar. Sci. **55**: 1332.

Chandler, C. R., R. M. Sanders, Jr., and A. M. Landry, Jr. 1985. Effects of three substrate variables on two artificial reef fish communities. Bull. Mar. Sci. **37**: 129-142.

Charbonnel E., P. Francour, J. G. Harmelin, and D. Ody. 1995. Problems in sampling and censusing artificial reef associated fish assemblages. Biol. Mar. Mediterr. **2**: 85-90.

Chua C. Y. Y. and L. M. Chou. 1994. The use of artificial reefs in enhancing fish communities in Singapore. Hydrobiologia. **285**: 177-187.

Clark, D. S. and J. M. Green 1990. Activity and movement patterns of juvenile Atlantic cod, *Gadus morhua*, in Conception Bay, Newfoundland, as determined by sonic telemetry. Can. J. Zool. **68:** 1434-1442.

Clark, D. S. and J. M. Green. 1991. Seasonal variation in temperature preference of juvenile Atlantic cod (*Gadus morhua*), with evidence supporting an energetic basis for their diel vertical migration. Can. J. Zool. **69**: 1302-1307.

Clark, C. W. and D. A. Levy. 1988. Diel vertical migrations by juvenile sockeye salmon and the antipredator window. Am. Nat. **131**: 271-290.

Connell, S. D. and G. P. Jones. 1991. The influence of habitat complexity on postrecruitment processes in a temperate reef fish population. J. Exp. Mar. Biol. Ecol. **151**: 271-294.

Cooper, W. E. and L. B. Crowder. 1979. Patterns of predation in simple and complex environments. pp. 257-267. *In* Stroud, R. H. and H. Clepper. 1979. Predator - Prey Systems in Fisheries Management: International Symposium on Predator-Prey Systems in Fish Communities and their Role in Fisheries Management, Atlanta, Georgia, July 24-27, 1978. Sport Fishing Institute, Washington, D. C. 504p.

Coull, B. C. and J. B. J. Wells. 1983. Refuges from fish predation: experiments with phytal meiofauna from the New Zealand rocky intertidal. Ecology **64**: 1599-1609.

Cote, D., D. A. Scruton, G. H. Neizgoda, R. S. McKinley, D. F. Rowsell, R. T. Lindstrom, L. M. N. Ollerhead, and C. J. Whitt. 2001a. A coded acoustic telemetry system for high precision monitoring of fish location and movement: application to the study of nearshore nursery habitat of juvenile Atlantic cod (*Gadus morhua*). MTS Journal. **32**: 54-62.

Cote, D., S. Moulton, D. A. Scruton, and R. S. McKinley. 2001b. Microhabitat use of juvenile Atlantic cod in a coastal area of Bonavista Bay, Newfoundland. Trans. Amer. Fish. Soc. **130**: 1217-1223.

Daan, N. 1973. A quantitative analysis of the food intake of North Sea cod, *Gadus morhua*. Neth. J. Sea Res. **6**: 479-517.

Dalley, E. L. and J. T. Anderson. 1997. Age-dependent distribution of demersal juvenile Atlantic cod (*Gadus morhua*) in inshore/offshore northeast Newfoundland. Can. J. Fish. Aquat. Sci. **54** (Suppl. 1) 168-176.

Dew, C. B. 1976. A contribution to the life history of the cunner, *Tautogolabrus adspersus*, in Fishers Island Sound, Connecticut. Chesapeake Sci. **14**: 101-113.

Dewees, C. M. and D. W. Gotshall. 1974. An experimental artificial reef in Humboldt Bay, California. Calif. Fish and Game. **60**: 109-127.

Downing, N., R. A. Tubb, C. R. El-Zahr, and R. E. McClure. 1985. Artificial reefs in Kuwait, Northern Arabian Gulf. Bull. Mar. Sci. **37**: 157-178.

Fast, D. E. and F. A. Pagan. 1974. Comparative observations on an artificial tire reef and natural patch reefs off southwestern Puerto Rico. Proceedings of an International Conference on Artificial Reefs, Houston, Texas. Center for Marine Resources, Texas A&M University. pp. 49-50.

Fowler, A. J., A. C. Jensen, K. J. Collins, and I. P. Smith. 1999. Age structure and diel activity of pouting on the Poole Bay artificial reef. J. Fish. Biol. **54**: 944-954.

Fraser, S., V. Gotceitas, and J. A. Brown. 1996. Interactions between age-classes of Atlantic cod and their distribution among bottom substrates. Can. J. Aquat. Sci. **53**: 305-314.

Gascon, D. and R. A. Miller. 1981. Colonization by nearshore fish on small artificial reefs in Barkley Sound, British Columbia. Can. J. Zool. **59**: 1635-1646.

Gee, J. M. and R. M. Warwick. 1994. Metazoan community structure in relation to the fractal dimensions of marine macroalgae. Mar. Ecol. Prog. Ser. **103**: 141-150.

Gibson, R. N., L. Robb, M. T. Burrows, and A. D. Ansell. 1996. Tidal, diel and longer term changes in the distribution of fishes on a Scottish sandy beach. Mar. Ecol. Prog. Ser. **130**: 1-17.

Gjøsæter, J. 1987. Habitat selection and inter year class interaction of young cod (*Gadus morhua*) in aquaria. Flødevigen rapportser. 1, 1987: 27-36.

Gotceitas, V. and J. A. Brown. 1993. Substrate selection by juvenile Atlantic cod (*Gadus morhua*): effects of predation risk. Oecologia **93**: 31-37.

Gotceitas, V. and P. Colgan. 1989. Predator foraging success and habitat complexity: quantitative test of the threshold hypothesis. Oecologia **80**: 158-166.

Gotceitas, V., S. Fraser, and J. A. Brown. 1995. Habitat use by juvenile Atlantic cod (*Gadus morhua*) in the presence of an actively foraging and non-foraging predator. Mar. Biol. **123**: 421-430.

Gotceitas, V., S. Fraser, and J. A. Brown. 1997. Use of eelgrass beds (*Zostera marina*) by juvenile Atlantic cod (*Gadus morhua*). Can. J. Aquat. Sci. **54**: 1306-1319.

Grant, S. M. and J. A. Brown. 1998. Diel foraging cycles and interactions among juvenile Atlantic cod (*Gadus morhua*) at a nearshore site in Newfoundland. Can. J. Fish. Aquat. Sci. **55**: 1307-1316.

Gregg, K. L. 1995. Comparisons of three manufactured artificial reef units in Onslow Bay, North Carolina. N. Am. J. Fish. Man. **15**: 316-324.

Gregory, R. S. and J. T. Anderson. 1997. Substrate selection and use of protective cover by juvenile Atlantic cod *Gadus morhua* in inshore waters of Newfoundland. Mar. Ecol. Prog. Ser. **146**: 9-20.

Gregory, R. S., J. T. Anderson, and E. L. Dalley. 1996. Use of habitat information in conducting assessments of juvenile cod abundance. NAFO. SCR DOC. 96/23.

Gregory, R. S., J. T. Anderson, and E. L. Dalley. 1997. Distribution of juvenile cod (*Gadus morhua*) relative to available habitat in Placentia Bay, Newfoundland. NAFO Sci. Coun. Studies. **29**: 3-12.

Grossman, G. D., G. P. Jones, and W. J. Seaman, Jr. 1997. Do artificial reefs increase regional fish production? A review of existing data. Fisheries **22**: 17-21.

Grove, R. S. and C. J. Sonu. 1985. Fishing reef planning in Japan. pp. 187-251. *In* D'Itri, M. (ed.) 1985. Artificial reefs: marine and freshwater applications. Lewis Publishers, Chelsea, Michigan. 589p.

Guénette, S., T. J. Pitcher, and C. J. Walters. 2000. The potential of marine reserves for the management of northern cod in Newfoundland. Bull. Mar. Sci. **66**: 831-852.

Guillen, J. E., A. A. Ramos, L. Martinez, J. L. Sanchez Lizaso. 1994. Antitrawling reefs and the protection of *Posidonia oceania* (L.) Delile meadows in the western Mediterranean Sea: Demand and aims. Fifth International Conference on Aquatic Habitat Enhancement. Bull. Mar. Sci. **55**: 645-650.

Guzmán, H. M, J. Campos, C. Gamboa, and W. A. Bussing. 1988. Un arrecife artificial de llantas: su potencial para el manejo de pesquerías. An. Inst. Cienc. del Mar y Limnol. Univ. Nal. Autón, México. **15**: 249-254.

Hammill, M. O. and G. B. Stenson. 2001. Estimated consumption by harp seals (*Phoca groenlandica*), hooded seals (*Cystophora cristata*), grey seals (*Halicoerus grypus*) and harbour seals (*Phoca vitulina*) in Atlantic Canada. J. Northw. At. Fish. Sci. **26**: 1-23.

Hawkins, A. D., G. G. Urquhart, and G. W. Smith. 1980. Ultrasonic tracking of juvenile cod by means of a large spaced hydrophone array. *In* Amlaner, G. J. and D. W. MacDonald. (eds.) 1980. A handbook on biotelemetry and radio tracking. Pergamon Press, Oxford. 804p.

Hawkins, A. D., N. M. Soofiani, and G. W. Smith. 1985. Growth and feeding of juvenile cod (*Gadus morhua* L.). J. Cons. Int. Explor. Mer. **42**: 11-32.

Helfman, G. S. 1993. Fish behaviour by day, night and twilight. pp. 480-512. *In* Pitcher, T. J. 1993. Behaviour of teleost fishes. 2nd Ed. Chapman and Hall. New York. 715p.

Higo, N., H. Hashi, I. Takahama, S. Tabata, M. Nagashima, S. Sakono, T. Kamimizutaru, and T. Yamasaki. 1980. On the fish gathering effect of the artificial reefs ascertained by the diving observation - VII - At the off sea of Makurazaki City. Mem. Fac. Fish., Kagoshima Univ. **29**: 51-63.

Hixon, M. A. 1991. Predation as a process structuring coral reef communities. pp. 475-508. *In* P. F. Sale. (ed.) The ecology of fishes on coral reefs. Academic Press, New York, N. Y. 754p.

Hixon, M. A. and J. P. Beets. 1989. Shelter characteristics and Caribbean fish assemblages: experiments with artificial reefs. Bull. Mar. Sci. **44**: 666-680.

Hower, A. E. 1998. Combining wave energy and artificial reef technology for sustainable coastal resource development. Oceans 98 - Engineering for sustainable us of the oceans: conference proceedings 28 September - 1 October 1998, Nice (France) IEEE/OES **3**: 1691-1695.

Ings, D. W., D. C. Schneider, R. S. Gregory, and V. Gotceitas. Manuscript. Coastline complexity and vegetation determine density of small fish in temperate nursery habitats.

Jenkins, S. A. and D. W. Skelly. 1994. Development of a soft reef sandlbar for offshore wave dissipation. Fifth International Conference on Awuatic Habitat Enhancement. Bull. Mar. Sci. **55**: 1338.

Jensen, P. A., I. F. Walker, J. M. Falk, W. R. Hall Jr., and H. H. Seymour. 1980. Artificial reefs for Delaware? Marine Advisory Service. Sea Grant College Program. University of Delaware. 36p.

Jessee, W. N., A. L. Carpenter, and J. W. Carter. 1985. Distribution patterns and density estimates of fishes on a southern California artificial reef with comparisons to natural kelp-reef habitats. Bull. Mar. Sci. **37**: 214-226.

Keats, D. W. 1990. A nocturnal inshore movement of juvenile cod *Gadus morhua* L. in eastern Newfoundland. J. Exp. Mar. Biol. Ecol. **139**: 167-173.

Keats, D. W. and D. H. Steele. 1992. Diurnal feeding of juvenile cod (*Gadus morhua*) which migrate into shallow water at night in Eastern Newfoundland. J. Northw. Atl. Fish. Sci. **13**: 7-14.

Keats, D. W., D. H. Steele, and G. R. South. 1987. The rôle of fleshy macroalgae in the ecology of juvenile cod (*Gadus morhua* L.) in inshore water off eastern Newfoundland. Can. J. Zool. **65**: 49-53.

Kennedy, M. and R. D. Gray. 1993. Can ecological theory predict the distribution of foraging animals? A critical analysis of experiments on the Ideal Free Distribution. OIKOS **68**: 158-166.

Kim, C. G. 2001. Artificial reefs in Korea. Fisheries **26**: 15-18.

Kock, R. L. 1982. Patterns of abundance in reef fishes near an artificial reef at Guam. Env. Biol. Fish. **7**: 121-136.

Krebs, C. J. 1994. Factors that limit distributions: Interrelations with other organisms. pp. 75-92. *In* C. J. Krebs. (ed.) 1994. Ecology: the experimental analysis of distribution and abundance. 4th Edition. Harper Collins College Publishers. New York, NY. 801p.

Laihonen, P., J. Haenninen, J. Chojnacki, and I. Vuorinen. 1997. Some prospects of nutrient removal with artificial reefs. European artificial reef research. Proceedings of the 1st EARRN conference, Ancona, Italy, March 1996. Southampton Oceanography Centre, Southampton, UK pp. 85-96.

Laufle, J. C. and G. B. Pauley. 1985. Fish colonization and materials comparisons on a Puget Sound artificial reef. Bull. Mar. Sci. **37**: 227-243.

Lima, S. L. and L. M. Dill. 1990. Behavioral decisions made under risk of predation: a review and prospectus. Can. J. Zool. **68**: 619-640.

Lindberg, W. J. 1997. Can science resolve the attraction-production issue? Fisheries **22**: 10-13.

Linehan, J. E., R. S. Gregory, and D. C. Schneider. 2001. Predation risk of age 0 cod (*Gadus*) relative to depth and substrate in coastal waters. J. Exp. Mar. Biol. Ecol. **263**: 25-44.

Løkkeborg, S. and A. Fernö. 1999. Diel activity pattern and food search behaviour in cod, *Gadus morhua*. Environ. Biol. Fish. **54**: 345-353.

Luckhurst, B. E. and K. Luckhurst. 1978. Analysis of the influence of substrate variables on coral reef fish communities. Mar. Biol. **49**: 317-323.

Mandelbrot, B. B. 1982. The fractal geometry of nature. W. H. Freeman and Company, San Francisco.

Mark, D. M. 1984. Fractal dimension of a coral reef at ecological scales: a discussion. Mar. Ecol. Prog. Ser. **14**: 293-294.

Mattila, J. 1992. The effect of habitat complexity on predation efficiency of perch *Perca fluviatilis* L. and ruffe *Gymnocephalus cernuus* (L.) J. Exp. Mar. Biol. Ecol. **157**: 55-67.

McGurrin, J. M. and A. J. Fedler. 1989. Tenneco II artificial reef project: An evaluation of Rigs-to-Reefs fisheries development. Bull. Mar. Sci. **44**: 777-781.

Methven, D. A. and C. Bajdik. 1994. Temporal variation in size and abundance of juvenile Atlantic cod (*Gadus morhua*) at an inshore site off eastern Newfoundland. Can. J. Fish. Aquat. Sci. **51**: 78-90.

Methven, D. A. and C. McGowan. 1998. Distinguishing small juvenile Atlantic cod (*Gadus morhua*) from Greenland cod (*Gadus ogac*) by comparing meristic characters and discriminant function analyses of morphometric data. Can. J. Zool. **76**: 1054-1062.

Methven, D. A. and D. C. Schneider. 1998. Gear-independent patterns of variation in catch of juvenile Atlantic cod (*Gadus morhua*) in coastal habitats. Can. J. Fish. Aquat. Sci. **55**: 1430-1442.

Methven, D. A., R. L. Haedrich, and G. A. Rose. 2001. The fish assemblage of a Newfoundland estuary: diel, monthly and annual variation. Est. Coast. Shelf Sci. **52**: 669-687.

Mikhail, M. Y. and H. E. Welch. 1989. Biology of Greenland cod, *Gadus ogac*, at Saqvaqjuac, northwest coast of Hudson Bay. Environ. Biol. Fishes. **26**: 49-62.

Minitab. 1992. Release 9.1.

Morin, B., C. Hudon, and F. Whoriskey. 1991. Seasonal distribution, abundance, and life-history traits of Greenland cod, *Gadus ogac*, at Wemindji, eastern James Bay. Can. J. Zool. **69**: 3061-3070.

Morse, D. R., J. H. Lawton, M. M. Dodson, and M. H. Williamson. 1985. Fractal dimension of vegetation and the distribution of arthropod body lengths. Nature **314**: 731-733.

Mottet, M. G. 1981. Enhancement of the marine environment for fisheries and aquaculture in Japan. Washington Dept. Fish., Tech Rep. 69. 176p.

Munoz-Perez, J. J., J. M. G. Mas, J. M. Naranjo, E. Torres, and L. Fages. 2000. Position and monitoring of anti-trawling reefs in the Cape of Trafalgar (Gulf of Cadiz, SW Spain). Bull. Mar. Sci. **67**: 761-772.

Myers, R. A. and N. G. Cadigan. 1993. Density-dependent juvenile mortality in marine demersal fish. Can. J. Fish. Aquat. Sci. **50**: 1576-1590.

Nelson, W. G. and E. Bonsdorff. 1990. Fish predation and habitat complexity: are complexity thresholds real? J. Exp. Mar. Ecol. **141**: 183-194.

Ogawa, Y., S. Takeuchi, and A. Hattori. 1977. An estimate for the optimum size of artificial reef. Bull. Jap. Soc. Fish. Ocean. **30**: 39-45.

Parker, Jr., R. O., R. B. Stone, C. C. Buchanan, and F. W. Steimle, Jr. 1974. How to build artificial reefs. Fishery Facts 10. U.S. Seattle, WA. 47p.

Perry, R. I. and S. J. Smith. 1994. Identifying habitat associations of marine fishes using survey data: an application to the Northwest Atlantic. Can. J. Fish. Aquat. Sci. **51**: 589-602.

Pihl, L. 1982. Food intake of young cod and flounder in a shallow bay on the Swedish west coast. Neth. J. Sea. Res. **15**: 419-432.

Pitcher, T. J., R. Watson, N. Haggan, S. Guénette, R. Kennish, U. R. Sumaila, D. Cook, K. Wilson, and A. Leung. 2000. Marine reserves and the restoration of fisheries and marine ecosystems in the South China Sea. Bull. Mar. Sci. **66**: 543-566.

Quigel, J. C. and W. L. Thornton. 1989. Rigs to reefs - a case history. Bull. Mar. Sci. **44**: 799-806.

Reef Ball Development Group Ltd. 2001. Reef Ball Development Group Ltd. Internet Brochure. http://www.reefball.com/reef.htm.

Reggio, V. C. 1989. Petroleum structure as artificial reefs: A compendium. OCS/MMS-89/0021: MMS/GM-89-0021 174pp.

Rhodes R. J., M. Bell, and R. S. Pomeroy. 1994. Scuba diver expenditures associated with South Carolina's artificial reefs. Bull. Mar. Sci. **55**: 1350.

Russo, A. R. 1987. Role of habitat complexity in mediating predation by the gray damselfish *Abudefduf sordidus* on epiphytal amphipods. Mar. Ecol. Prog. Ser. **36**: 101-105

Salvanes, A. G. V., D. L. Aksnes, and J, Giske. 1992. Ecosystem model for evaluating potential cod production in a west Norwegian fjord. Mar. Ecol. Prog. Ser. **90**: 9-22.

Sanders, Jr., R. M., C. R. Chandler, and A. M. Landry Jr. 1985. Hydrological, diel and lunar factors affecting fishes on artificial reefs off Panama City, Florida. Bull. Mar. Sci. **37**: 318-328.

Sarno, B., C. W. Glass, G. W. Smith, A. D. F. Johnstone, and W. R. Mojsiewicz. 1994. A comparison of the movements of two species of gadoid in the vicinity of an underwater reef. J. Fish. Biol. **45**: 811-817.

SAS Institute Inc. 1999. SAS User's Guide: Statistics, version 8.0. SAS Institute Inc., Cary, NC

Scott, W. B. and M. G. Scott. 1988. Atlantic fishes of Canada. Can. Bull. Fish. Aquat. Sci. No. **219**.

Smith, G. W., C. W. Glass, A. D. Johnstone, and W. R. Mojsiewicz. 1993. Diurnal patterns in the spatial relationships between saithe, *Pollachius virens*, schooling in the wild. J. Fish. Biol. **43** (Suppl. A): 315-325.

Spanier, E., M. Tom, and S. Pisanty. 1985. Enhancement of fish recruitment by artificial enrichment of man-made reefs in the southeastern Mediterranean. Bull. Mar. Sci. **37**: 356-363.

Stein, R. A. 1979. Behavioral response of prey to fish predators. pp. 343-353. *In* Stroud, R. H. and H. Clepper. (eds.) 1979. Predator-Prey Systems in Fisheries Management. Sport Fishing Institute, Washington, D.C.

Sugihara, G. and R. M. May. 1990. Applications of fractals in ecology. TREE. **5**: 79-86.

Tat'yankin, Y. V. 1974. The distribution of juvenile cod [*Gadus morhua morhua* (L.)] pollock [*Pollachius virens* (L.)] and haddock [*Melanogrammus aeglefinus* (L.)] under temperature gradient conditions. J. Ichthyol. (Engl. Transl.). **114**: 1002-1009.

Tupper, M. and R. G. Boutilier. 1995. Effects of habitat on settlement, growth, and postsettlement survival of Atlantic cod (*Gadus morhua*). Can. J. Fish. Aquat. Sci. **52**: 1834-1841.

Tupper, M. and R. G. Boutilier. 1997. Effects of habitat on settlement, growth, predation risk and survival of a temperate reef fish. Mar. Ecol. Prog. Ser. **151**: 225-236.

Utne, A. C. W. 1994. Size-dependent risk taking in perch. J. Fish Biol. **45** (Suppl. A): 255

Walters, C. J. and F. Juanes. 1993. Recruitment limitation as a consequence of natural selection for use of restricted feeding habitats and predation risk taking by juvenile fishes. Can. J. Fish. Aquat. Sci. **50**: 2058-2070.

Wigley, S. E. and F. M. Serchuk. 1992. Spatial and temporal distribution of juvenile Atlantic cod *Gadus morhua* in the Georges Bank-Southern New England region. Fish. Bull. US **90**: 599-606.

Willis, T. J., R. B. Millar, and R. C. Babcock. 2000. Detection of spatial variability in relative density of fishes: comparison of visual census, angling, and baited underwater video. Mar. Ecol. Prog. Ser. **198**: 249-260.

Wyche, C. J. 1984. Observations on the behaviour of a saithe (*Pollachius pollachius* (L.)) school on a temperate reef. Prog. Under. Sci. **9**: 87-98.