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Ecological impacts of shrimp trawling on juveniles of selected benthic fish species in the southwestern Gulf of Maine

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Ecological impacts of shrimp trawling on juveniles of selected benthic fish species in the southwestern Gulf of Maine

Abstract
The ecological impacts of shrimp trawling on juveniles of selected benthic fish were studied in the southwestern Gulf of Maine. The fish community and selected environmental variables were sampled at two sites in summer and winter during 1998 and 1999. The community was composed of 24 species in 12 families. Site 1 was dominated by longhorn sculpin, Myoxocephalus octodecemspinosus, while American plaice, Hippoglossoides platessoides, was the dominant species in Site 2. Red hake, Urophycis chuss was abundant at both sites. There were also seasonal differences in the flux of fish at the two sites.

Correlation analysis showed the abundance of M. octodecemspinosus (Site 1) and H. platessoides (Site 2) to be predictable from environmental variables ordered into principal components (PC1--PC3) as predictors. These PCs represented a depth and sediment composite, a temperature and salinity composite, and an epibenthic macroinvertebrate composite, respectively. There was a strong overall correlation of the two species with PC2 ($r_c = 0.921$). This was largely attributed to a significant correlation of the abundance of M. octodecemspinosus with this PC ($F[1,4] = 21.703, p = 0.010$).

American plaice (H. platessoides), silver hake (Merluccius bilinearis), winter flounder (Pseudopleuronectes americanus), and Atlantic cod (Gadus morhua) were the four species selected for study. Only H. platessoides and M. bilinearis were caught in sufficient numbers to enable the correlation of their abundance with environmental variables ordered into PCs. The result was not significant ($R_{Adj.2} = 0.0; r_c = 0.8545$). Spatial and temporal differences in abundance of the two species tested with MANOVA were also not significant ($F[2,4] = 2.7915, p = 0.1742$; and $F[2,4] = 4.9998, p = 0.0816$, respectively).

The population structure of H. platessoides analyzed with length frequency analysis and scale-age determination revealed four age classes, Age 0+, 1+, 2+ and 3+, to be present in the study area. The diet of H. platessoides was also analyzed to understand potential impacts of trawling on its trophic ecology. Ampharetid polychaetes, other polychaetes, amphipods, bivalves and cyclopoid copepods were the dominant prey taxa. These taxa were also abundant in the environment.

A number of recommendations were proposed from these results on how such studies could increase our understanding of fishing gear impacts on benthic habitats, enabling appropriate management decisions to be made.

Keywords
Biology, Ecology, Biology, Oceanography

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ECOLOGICAL IMPACTS OF SHRIMP TRAWLING ON JUVENILES OF SELECTED BENTHIC FISH SPECIES IN THE SOUTHWESTERN GULF OF MAINE

BY

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DISSERTATION

Submitted to the University of New Hampshire in Partial Fulfillment of the Requirements for the Degree of

Doctor of Philosophy
in
Zoology

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04/27/01
Date
DEDICATION

A program of active graduate education was not fully developed at the University of Papua New Guinea (UPNG) at the time I graduated. Hence, to fulfill my interest towards such an education, I had to join the race with many from my generation of UPNG graduates to study abroad. There were many hurdles to overcome: academic challenges; difficulties of separation from family and friends; and the stresses of immersing into a foreign social and cultural environment.

My endurance in this race, ultimately for the Ph. D., would not have been possible without my parents' positive influences on me during my early childhood that shaped my character. My late father inspired me through his skills of self-control and perseverance in pursuing difficult tasks until satisfactory results were achieved. My mother, inspired by her own childhood as an orphan after the early loss of my grandmother, founded in me the value of self-respect, trust and determinism as key to being successful in life. These values influenced my attitude towards my education and founded the strategies with which I pursued this race.

I dedicate this dissertation in memory of my late father, Gerard, and my mother, Catherine Mungkaje, who has promised to keep in good health to see me get my Ph.D. Without their positive influences, I would have dropped out of the race long before I reached this finishing line.
ACKNOWLEDGEMENTS

This dissertation would not have come to existence without the constant guidance, encouragement and suggestions of Professor Hunt Howell under whose supervision this work was undertaken. He also offered many hours of help on field sampling cruises, and with the loading and unloading of heavy equipment and samples that followed those cruises. I thank him for all those contributions.

Other members of my dissertation committee also offered suggestions and comments that guided this work to a success. Professor Larry Harris gave me access to his library of benthic ecology texts and provided advice on strengthening the statistical aspects of the benthic sampling design. Associate Professor Ray Grizzle's help with resources and expertise on identifying macroinvertebrate samples was very encouraging during the early phase of this project, since my experience in benthic ecology was limited. Assistant Professor Rich Langan offered many critical suggestions on aspects of the project. Professor Ann Bucklin steered me into exploring the possibility of using multivariate statistics in making sense of the multivariate data set obtained in the study; data analysis would have been chaotic and statistically ineffective without it. My sincere thank you to them all for their input.

My confidence with multivariate data analysis was enhanced by a multivariate statistics course that I took at the National Conservation Training Center of the U.S. Fisheries and Wildlife Service, Shepherdstown, West Virginia. I would like to thank Dr. Alan Temple who approved my enrollment on the course.
At the start of this project Dave Goethel's knowledge of the shrimp fishing areas at the location of this study helped Professor Howell and I to identify the study area. Dr. Chris Neefus provided advice on statistical issues relevant to the sampling design of this project. I thank them both for their help at this critical phase of the project.

Offshore field studies at this latitude where weather conditions are always harsh with sudden disruptions to planned sampling cruises make field sampling often logistically difficult. Getting help for fieldwork under these conditions is not easy. The data used in this dissertation would not have been collected without the deck and shore help offered by Susan Reynolds, Elizabeth Fairchild, Nick King, Bruno Chazaro, Megan Tyrrell, James Sulikowski, Travis Schoppmeyer, Nathan Rennels and Noel Carlson. Captain Paul Pelletier and mate Kenneth Houtler of the *R/V Gulf Challenger* offered safe and efficient shipboard sampling expertise during all cruises. I thank them all for those long hours.

As in any graduate research project, one does not always have all the necessary expertise and equipment that are needed. Dr. Larry Ward, Dr. Alan Baker, and Dr. Edward Tillinghast lent me equipment to collect and process sediment samples. Dr. Larry Ward and Dr. Alan Baker allowed me access to their laboratories to perform organic content analysis. Dr. Ted Loder gave me the permission to use a CTD unit from his laboratory to collect temperature and salinity data, and his graduate student, Erin Penfold, showed me how to use the accompanying software to download and process the data. Manya Hult was always helpful in lending me teaching equipment from the Zoology Department, and also in advising me of the correct procedures for disposing chemical and biological wastes. My appreciation and thanks is due to them all.
This work would not have been undertaken without funding from a number of sources. Grants from the Center for Marine Biology, University of New Hampshire (UNH) funded all of the fieldwork, and grants to Professor Howell were used to acquire laboratory consumables and equipment. Tuition, book and stipend costs were covered under a HEP Scholarship to me from the Government of Papua New Guinea, and financial support to my family was provided by the Staff Development Unit, University of Papua New Guinea. The UNH Graduate School provided travel grants for presenting the results of this research at two scientific conferences. These sources of financial support are gratefully acknowledged.

Besides academic routines, there were administrative aspects and deadlines to deal with. I thank the following for their time in performing those tasks on a regular basis. Barbara Millman and Diane Lavalliere of Zoology Department were always helpful in sorting administrative matters within the department. Kathleen Sellew and Rita Snider of the Midwest Universities Consortium for International Activities, Inc. (MUCLA), University of Minnesota, and Marehari Liriope, of the Office of Higher Education, Papua New Guinea Commission of Higher Education, Port Moresby, for their time in managing all administrative aspects my HEP fellowship. Leila Paje-Manalo and Liisa Reimann of the Office of International Students and Scholars, UNH, were always very helpful in providing advice and assistance with immigration matters.

Central to this success is my family. I would like to thank my wife Martha, and daughters Catherine and Grace, for their constant love, support and encouragement. They always provided a caring environment that facilitated my physical and mental recuperation, and were always tolerant whenever I was unable to attend to their needs.
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ABSTRACT

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by

Augustine Japeni Mungkaje

University of New Hampshire, May 2001

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Correlation analysis showed the abundance of M. octodecemspinosus (Site 1) and H. platessoides (Site 2) to be predictable from environmental variables ordered into principal components (PC1-PC3) as predictors. These PCs represented a depth and sediment composite, a temperature and salinity composite, and an epibenthic macroinvertebrate composite, respectively. There was a strong overall correlation of the two species with PC2 ($r_c = 0.921$). This was largely attributed to a significant correlation of the abundance of M. octodecemspinosus with this PC ($F_{[1,4]} = 21.703, p = 0.010$).
American plaice (*H. platessoides*), silver hake (*Merluccius bilinearis*), winter flounder (*Pseudopleuronectes americanus*), and Atlantic cod (*Gadus morhua*) were the four species selected for study. Only *H. platessoides* and *M. bilinearis* were caught in sufficient numbers to enable the correlation of their abundance with environmental variables ordered into PCs. The result was not significant ($R^2_{Adj} = 0.0; r_c = 0.8545$). Spatial and temporal differences in abundance of the two species tested with MANOVA were also not significant ($F_{[2,4]} = 2.7915, p = 0.1742$; and $F_{[2,4]} = 4.9998, p = 0.0816$, respectively).

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A number of recommendations were proposed from these results on how such studies could increase our understanding of fishing gear impacts on benthic habitats, enabling appropriate management decisions to be made.
INTRODUCTION

Exploitation of natural fish stocks can be targeted at a single species or a group of related species based on criteria such as size, abundance, economic value and marketability. Depending on what the target species are, fishermen select and use appropriate gears to harvest the selected stocks. Trawls, demersal handlining, horizontal and vertical longlining, gillnets, baited traps, and seines are examples of the gears used by commercial fishermen. A good description of the range of conventional forms of commercial fishing gears and species targeted by these respective gears is given in Huntsman (1948). Iversen (1996) listed and described examples of the kinds of fishing gear specifically used in the United States to exploit marine finfish and shellfish.

With the advancement in modern technology enabling the building of bigger, mechanized and powerful vessels, many of these gears used for commercial purposes, especially in industrial fisheries operated by large fishing companies, have been modified into much larger, heavier, and cost effective forms. Besides contributing to the decline in most of the world's major fish stocks (Beverton, 1994; Knauss, 1994; Botsford et al., 1997; Fogarty and Murawski, 1998; Pauly et al., 1998) by increasing fishing power, these developments, especially mobile forms such as trawls and dredges, come with another price tag. The destruction to benthic habitats by such heavy gear and the negative impacts this can have on the benthos (Caddy, 1973; Langton and Robinson, 1990; Messier et al., 1991; Bergman and Hup, 1992; Brylinsky et al., 1994; Kaiser and Spencer, 1996; Collie et al., 1997; Engel and Kvitek, 1998) and benthic fish populations (Auster, 1998; Jennings and Kaiser, 1998; Auster and Langton, 1999) is now being recognized.
**Impacts of Mobile Gear**

Trawls and dredges have been found to harm the benthic environment by causing physical damage to its biotic and abiotic components (Caddy, 1973; Langton and Robinson, 1990; Messieh *et al.*, 1991; Bergman and Hup, 1992; Brylinsky *et al.*, 1994; Kaiser and Spencer, 1996). In Europe, the weight and tow speed of some of these gears have been increased since the 1980s due to more powerful vessels (Rogers *et al.*, 1998).

Gears such as hydraulic dredges are capable of excavating furrows 1 m wide and 40 cm deep (Beukema, 1995, cited in Rogers *et al.*, 1998) and scallop dredges can excavate down to 10 cm deep in soft sediment (Kaiser *et al.*, 1996). Similarly, otter trawl doors can excavate 4.5-10 cm deep into the sediment (Brylinsky *et al.*, 1994; Kaiser *et al.*, 1996; Gordon *et al.*, 1998; Pilskaln *et al.*, 1998; DeAlteris *et al.*, 1999). Trawling can result in scraping and ploughing of the substrate, resuspension of sediment (Churchill, 1989), destruction of the benthos and their smothering from the dumping of processed wastes, post-fishing mortality, and long-term trawl-induced changes to the benthos (Messieh *et al.*, 1991; Jones, 1992; Rogers *et al.*, 1998). Changes to the texture of sediments from suspension can affect the distribution of adult and larval stages of benthic invertebrates (Messieh *et al.*, 1991). This study also listed reducing food quality for filter feeders, and damage to gills and spawning areas for benthic invertebrates as potential impacts.

Evidence of physical displacement of adult benthic polychaetes, 25-30 m into the water column from sediment resuspension has also been reported (Pilskaln *et al.*, 1998).

Additionally, chemical alterations such as the mobilization of heavy metals can result from sediment resuspension. Furthermore, sediment resuspension can also change the normal pathways of remineralization and nutrient dynamics at the sediment-water
interface. This results largely from burying organic matter from the more aerobic environments at the interface to the more anaerobic environments in the deeper layers beneath, thereby changing the relative importance of the roles of eukaryotic and prokaryotic microorganisms in nutrient remineralization (Pilskaln et al., 1998).

With knowledge of trophic pathways existing within both the water column and the benthic environment in an impacted area, one can perceive the potential perturbation these impacts can cause to the structure of the respective communities, either directly or indirectly, affecting processes at the community and ecosystem levels. Depending on the nature of both natural and anthropogenic perturbations, resultant impacts can affect existing trophic pathways in a bottom-up or top-down manner (Hunter and Price, 1992; Menge, 1992; Power, 1992). Perturbations impacting higher trophic levels leading to either direct or indirect influence on the structure of lower trophic levels induce a top-down impact, while the converse induces a bottom-up impact (Menge, 1992). A classic example of a top-down impact transmitted trophically was that due to the effects of a top predator, the starfish Pisaster ochraceus, on lower trophic levels on a benthic intertidal community in Washington, U.S.A. (Paine, 1966; 1980). Grosholz et al. (2000) also reported evidence of a similar impact on a coastal marine food web in California by an introduced green crab, Carcinus maenas, monitored over a 9-year period. A meta-analysis of freshwater pelagic food webs confirmed that zooplankton abundance was regulated by consumer (top-down) control, while phytoplankton biomass was under strong resource (bottom-up) control (Brett and Goldman, 1997). In the fisheries context, direct species removal by fishing in marine systems has been found to exert a top-down effect (Parsons, 1996; Fogarty and Murawski, 1998).
Impacts of mobile gear can exert both top-down and bottom-up impacts. Top-down impacts such as a reduction in juvenile fish recruitment can occur due to destruction of habitat attributes that enhance feeding and growth (Sogard, 1992; Gibson, 1994; Tupper and Boutlier, 1995). Likewise, it can also result from an increase in predatory risks resulting from a reduction in the refuge role of physical habitat attributes (Gotceitas and Brown, 1993; Auster et al., 1994; Gregory and Anderson, 1997). Bottom-up impacts can arise from disturbances such as sediment resuspension leading to the mobilization of inorganic nutrients, or the burying of organic matter resulting in the alteration of its remineralization pathways (Pilskaln et al., 1998), affecting processes that channel energy to higher trophic levels. Because such impacts in any fisheries are indirect, they are only detectable at higher levels of ecological organization after a time lag. Consequently, they are often difficult to associate directly with fishing activities. Therefore, they are often overlooked in management regulations.

In many instances community and ecosystem level analyses are precluded by either non-availability of appropriate data or good data being expensive (Larkin, 1996), and difficult (Botsford et al., 1997) to obtain. The generation of suitable hypotheses, and the execution of sensitive experimental field studies designed with appropriate spatial and temporal scales taken into account (Langton et al., 1994), would be necessary to trace the nature of bottom-up impacts of fishing activities in marine ecosystems. In systems where some data exist, these data can serve as the basis to develop suitable theoretical and conceptual models (Auster, 1998) with which to make further predictions on the nature of potential impacts.
Approaches to Fisheries Management

Traditional Fisheries Management Practices

Traditionally, fisheries management has mainly focused on direct impacts of fishing on fish stocks. These management methods include monitoring: 1) catch per unit effort (CPUE) data; 2) estimates of recruitment based on adult stock biomass and fecundity; 3) numbers of fishing licenses in any one fishery; and 4) seasonal closures to areas known or assumed to be spawning and nursery grounds (Hilborn and Walters, 1992; Wallace et al., 1994; Botsford et al., 1997). Nearly all of these management strategies focus at lower levels of ecological organization, usually at the level of the population of the exploited species, and are concerned more with the top-down impacts of fishing (Larkin, 1996).

This approach to fisheries management seems insufficient to accommodate the complexity of interrelationships between the physical and biological attributes of marine ecosystems (Hutchings and Myers, 1994). There has been a rapid decline in some of the world’s major fisheries in recent years. Half the fish stocks are fully exploited and another 22% are overexploited (Botsford et al., 1997). There has also been a shift in global fisheries landings from large piscivorous species to smaller planktivorous species and invertebrates in the last 45 years, especially in the Northern Hemisphere (Pauly et al., 1998). A steady decline has been reported in the major groundfish stocks (cod, haddock and yellowtail flounder) off the northeastern United States since 1978 (Anthony, 1993). Atlantic cod stocks in Newfoundland and Labrador have declined in the last decade (Hutchings and Myers, 1994; Walters and Maguire, 1996), and a similar trend has been shown for the North Sea cod stocks recently (Cook et al., 1997). A recent empirical
analysis (Hutchings, 2000) showed that some of the world’s heavily exploited gadid and flatfish stocks may be in danger of extinction.

**Community and Ecosystem Considerations in Fisheries Management**

It is becoming imperative that the simplistic traditional approaches of fisheries management need to be complemented with new, ecologically intricate, and realistic ones (Botsford *et al.*, 1997; Roughgarden, 1998). Hutchings and Myers (1994), assessing the possible causes for the collapse of the Atlantic cod (*Gadus morhua*) fishery of Newfoundland and Labrador, have suggested that aspects of community ecology that influence a species' abundance should also be considered in the management of exploited populations. A recent study addressing a similar issue in freshwater systems has recommended that habitat restoration efforts to enhance fish production should focus more on the community and ecosystem level (Minns *et al.*, 1996). Such studies should aim to incorporate relevant natural processes of ecological systems, focusing more on community- and ecosystem-level aspects of marine and freshwater ecosystems. The United States has taken a major step by mandating this approach in managing its fisheries resources through its 1996 Magnuson-Stevens Fishery Conservation and Management Act (Benaka, 1999; Fluharty, 2000; Rosenberg *et al.*, 2000).

An important step in this direction has been the determination and delineation of some of the world’s principal large marine ecosystems (Larkin, 1996). An illustration of forty-nine of these large marine ecosystems is given in Sherman (1994), where the Gulf of Maine is described as one of the subsystems of the Northeast United States Continental Shelf Ecosystem. Appropriate spatial and temporal scales must also be considered in these efforts (Lewis *et al.*, 1996). Furthermore, a more multi-disciplinary approach
should be encouraged in such studies, allowing for a mutual exchange of the necessary data required to better understand the processes involved in a more holistic sense (Botsford et al., 1997; Rose, 2000). The Global Ocean Ecosystems Dynamics (GLOBEC) project is one such example (U.S. GLOBEC, 1992; Reeve, 1993; Sherman, 1994). A research program under this project is being conducted on the Georges Bank ecosystem (Frady, 1996; Wiebe et al., 1996) to better understand the interrelationships between the biological and physical processes of this system.

Field studies done at the ecosystem-level to predict the trophic dynamics of fish communities due to fishing pressure are rare (Botsford et al., 1997; Crowder and Murawski, 1998). It is apparent from a recent review that despite numerous studies on impacts of mobile gear in general, empirical data linking habitat and survival of late-juvenile and adult fish populations is lacking (Auster and Langton, 1999). This makes predictions of ecological impacts from the use of such gear on benthic fish populations difficult. Consequently, management decisions have to take conservative approaches.

Gomes and Haedrich (1992) used the “press perturbation” technique (Bender et al., 1984) in a modeling study to predict the dynamics of community structure from knowledge of the food web structure of fish communities of the Grand Banks of Newfoundland. They simulated the result of removing species on the community, as would occur through natural processes and fisheries exploitation, and predicted the consequent changes in the community structure. Their models were simplified to include seven of the most persistent species on the Grand Banks (six fish species and sea birds) from a 17-year data series smoothed over an annual time scale in each model. Seventeen models out of the forty generated were selected, on the basis of their local stability, as
possible representations of the true structure of fish communities in this system. It was not possible to conclusively predict which of these seventeen represented the true structure of the community in the simulations. They concluded that although their simulated models were better than traditional single species fisheries models, the simplification of their system in the models possibly contributed to their inability to predict the correct model structure after species removal. However, they felt that the inclusion of additional complexity to increase realism to the models would only increase the inability to predict the true model structure.

This result reflects the complex nature of processes involved at the community and ecosystem level, and consequently the uncertainty in predicting the outcome of the net influence of these processes on community structure. This is inevitable since ecological complexity demands careful modeling (Rose, 2000) with adequate parameterization in the models in order to represent ecological processes more realistically. With caution on increased variance in parameter estimation associated with complex models, the development of innovative quantitative techniques to adequately deal with model uncertainty (see Hilborn and Mangel, 1997; Burnham and Anderson, 1998) would enable ecologists to describe ecological phenomena more realistically from their data. With the development of personal computers, faster computers are now becoming readily accessible to most ecologists. Multivariate data sets can be easily stored, processed and analyzed, allowing hypotheses at the community and ecosystem level to be objectively formulated and tested.

When undertaking studies of the impact of mobile gear on marine ecosystems careful considerations must be given to both qualitative and quantitative trophic
interconnections of the constituent species. Sensitive sampling designs and robust statistical techniques to enable separating impacts due to mobile gear (Kaiser, 1998; Engel and Kvitek, 1998) from similar impacts due to natural perturbations (Hall, 1994; Watling, 1996) must be planned within appropriate temporal and spatial scales of study (Langton et al., 1994; Langton, 1998). This will improve the certainty with which impacts due to mobile gear on benthic communities can be detected, and furthermore, allow for predictions of the nature of impacts expected in similar systems where direct data may not be readily available (Auster, 1998). It is with a strong predictive ability that efficient management decisions in regulating the operation of ecologically harmful fishing gear can be made over broad spatial scales.

Focus of this Study

The indirect impacts of fishing can be best studied over appropriate spatial (km² to ≥ 100 km²) and temporal (days to years) scales (Langton, et al., 1994), focusing on the community and ecosystem levels of ecological organization. Studies of fisheries systems at these levels should consider biological information of the species in relation to the broader influences of abiotic (physical) and biotic (ecological) processes relevant to the system at these higher levels of ecological organization.

A number of important ecological processes are known to marine ecologists and fisheries scientists to ultimately affect populations of aquatic species. Spawning, dispersal and recruitment operate more at the population level, while processes such as competition and predation have a greater influence at the community level. A diagrammatic conceptualization of biotic and abiotic processes relevant to each of the levels of
ecological organization viz. individual, population, community and ecosystem levels, is illustrated in a review on sustainability, yield and health of coastal ecosystems (Sherman, 1994). Relevant physical processes include changes in the daylength, the degree of solar insolation, and vertical mixing and horizontal circulation of water masses. Influenced by variables such as the amount of solar radiance, temperature, salinity, density and wind forcing, mixing can suspend and redistribute nutrients within the water column. This would trigger the onset of biological processes such as increased primary productivity (blooms), spawning and recruitment in the ocean, especially in the seasonal environments of temperate and boreal marine systems. Such coupling between the physical and biological processes has been shown to explain long-term dynamics of marine fish stocks in many regions of the world (Mann, 1993), and is the basis of the "match-mismatch" hypothesis (Cushing, 1990).

**Life History Considerations**

According to the theory of “saltatory ontogeny” with reference to the early life history of fish (Balon, 1984, 1990, 1999), different developmental stages signify “periods” of threshold in ontogenesis when truly functional stages are distinguishable. These periods of threshold in development are accompanied by dramatic changes in size, shape, structure, physiology and behavior (Fuiman and Higgs, 1997). Ecologically, this means that the processes listed above will affect different life history stages differently (Rose, 2000). Juveniles differ from larval and adult life history stages both morphologically and ecologically. They are sufficiently developed to respond to biotic and abiotic processes better than larval stages, but not as effectively as adults. Therefore, they may cope more successfully with the influences of physical factors such as thermal
stress than the larval stages, through behavioral responses such as active avoidance. However, they may not be as effective as adults in competitive and predatory interactions. Ultimately, the survival of the adult and juvenile stages is the product of their respective age-specific survival probabilities (Hutchings, 1997). This has been the basis of research that seeks to understand age-specific survival in early life stages of fish (Houde, 1987, 1996) and its consequence on recruitment variability.

With such understanding, it is possible to identify different ecological processes that can influence the ecology of juvenile fish, either at the population, community or ecosystem level. If the nature of the influence of these ecological processes on the juvenile stages or "periods" (Balon, 1984, 1990, 1999) of species of interest can be established quantitatively, it should be possible to get a better insight into the critical processes that influence recruitment success of this life history stage. Being comparatively less passive and susceptible to adverse conditions of the physical environment than the larval stage, the juvenile stage has a higher survivorship, making it a promising stage to study ecological processes that have applied relevance to fisheries management.

**Rationale for Studying Trawling Impacts with Juvenile Fish**

The destruction of the small-scale topographic complexity of the benthic environment by trawls and dredges (Caddy, 1973; Langton and Robinson, 1990; Messieh et al., 1991; Brylinsky et al., 1994) is a principal source of ecological impact arising from these fishing methods. This is largely due to a reduction in the nursery and refuge function of these features to juvenile fish. My focus on juvenile fish is based on a number of reasons.
Firstly, traditional assessment of recruitment is based on the ‘fecundity and biomass-of-parent-stock’ approach to forecast the abundance of fish stocks (Hjort, 1914; Ricker, 1954; Myers and Barrowman, 1996; Cook et al., 1997). This approach does not resolve differences in mortality and survival rates between the larval and juvenile life history stages. Analyses of recruitment time series for seventeen fish populations, largely of gadids and pleuronectids, from five regions have indicated that the juvenile stage was very important for population regulation in most of these species (Myers and Cadigan, 1993). Therefore, for proper management it is important to consider the two stages separately based on their ontogeny and differences in habitat requirements, since different processes are expected to affect each stage differently, both qualitatively and quantitatively.

Secondly, unlike egg and larval stages, juveniles are less susceptible to mortality due to density-independent factors (largely abiotic ones) and their abundance tend to correlate well with that of their subsequent adult abundance (Rose, 2000). By definition, juveniles are individuals that have overcome two major selective forces on this life history stage: 1) overwinter mortality of temperate species, associated with starvation and extremes of physical variables, and 2) predation (Sogard, 1997). Therefore, new insights to enhancing natural fish stocks may be attained by studying the ecology of juvenile fish, since they have succeeded in one of the first natural screening processes.

Thirdly, if the destruction of benthic habitats through shrimp trawling has any adverse impacts at all on benthic fish communities, juveniles would be more at risk. This is the life history stage during which benthic fishes initially establish important ecological connections with the benthic environment (Lough et al., 1989; Levin, 1991; Love et al.,...
1991; Gotceitas and Brown, 1993; Lough and Potter, 1993; Auster et al., 1994; Gregory and Anderson, 1977; Grant and Brown, 1998). Densities of post-larval silver hake *Merluccius bilinearis*, have been found to be positively correlated with amphipod tubes (a benthic microtopographic feature) in the northern Mid Atlantic Bight (Auster et al., 1994). The successful settlement of juveniles onto benthic habitats is critical to their subsequent recruitment success into local fisheries as sub-adults and adults. A positive relationship between settlement and recruitment has been shown in the cunner *Tautogolabrus adspersus*, a typical benthic species, in the Gulf of Maine (Levin, 1996).

Fourthly, bycatch of finfish in shrimp trawling have been reported to consist largely of small-sized individuals (Murray et al., 1992), with a good proportion of these individuals consisting generally of juveniles and sub-adults. Analysis of length frequency distribution of discarded fish from the shrimp fishery in the southwestern Gulf of Maine indicated a similar pattern (Howell and Langan, 1987, 1992).

The fifth reason is that juvenile fish, having endured the transition from the more passive mode of existence as larvae in the plankton to a more active one as benthic juveniles, can be considered to possess evolutionary qualities that contributed to this successful transition. From an applied perspective if enough were known about their ecology, their probability of becoming adults could potentially be enhanced by appropriate scientific and technological interventions. Examples could include programs such as the designation of marine protected areas in locations known to be spawning and nursery grounds, and the installation of artificial structures in sub-optimal habitats to enhance their suitability as fish habitats.
Finally, if such programs demonstrate adult populations to be strongly correlated with increases in the survivorship of juvenile stages, and the processes critical to this relationship are understood, new insights into wild stock enhancement using juveniles reared artificially can be attained. Better release strategies and protocols can then be developed to enhance their probability of survival to maturity once released. Active research in this direction is beginning to gain momentum (Polovina, 1991; Fairchild, 1998; Tanaka et al., 1998).

**Objective of this study**

The principal objective of this study is to investigate the ecology of juveniles of selected fish species within the southwestern Gulf of Maine ecosystem with respect to any impacts of commercial trawling for the northern shrimp *Pandalus borealis*. It aims to test the hypothesis that shrimp trawling gear cause physical disturbances and damage to the benthic environment (biotic and abiotic), impacting its habitat and trophic attributes important to resident juvenile fish populations. This can contribute to further declines in their abundance, besides that due to fishing and natural mortality. Separating these sources of mortality is not easy. Some knowledge of their relative magnitudes with respect to different life history stages can allow some insight into their relative importance as sources of mortality.

Based on their reported predominance in bycatch in this fishery in a previous study (Howell and Langan, 1992) the selected fish species are: 1) American plaice (*Hippoglossoides platessoides*); 2) winter flounder (*Pseudopleuronectes americanus*); 3) silver hake (*Merluccius bilinearis*); and 4) Atlantic cod (*Gadus morhua*). This selection allows juveniles of both predominantly bottom-associated species (1&2) and species that...
have a tendency to migrate diurnally off the bottom (3&4) to be represented in the study. Any ecological impacts from this fishing method on juveniles of these species are investigated, and the mechanisms by which any detected impacts may be transmitted ecologically will be described. The results could then be used to demonstrate how broader environmental and habitat management practices are vital, and can be useful as fisheries management tools.

To achieve this objective, a number of key questions were asked in order to integrate the theoretical basis of the hypothesis into an empirical framework. These questions are:

1. Are there any significant differences in attributes of the benthic habitat consisting of: i) sediment organic content; ii) sediment particle size composition; iii) microtopographic features between the regularly trawled site, and the site where this method of fishing is less frequent; and iv) abundance and composition of benthic invertebrates?
2. Does the abundance of juveniles of the selected fish species correlate with differences in the above habitat attributes?
3. Does diet in the juvenile fish, an important indicator of the nursery function of juvenile habitats, vary significantly between the two sites?
4. Do the observed dietary patterns of the juvenile fish correlate with the abundance of potential benthic-invertebrate food sources?
5. What interpretations can be made of the ecological roles of the above habitat attributes to the selected juvenile fish between the 'trawl' and 'no trawl' sites?
6. What recommendations for fisheries management can be made from these results?
To best answer these questions the study took a multivariate approach. Juveniles of the four species listed above of benthic fish were sampled as the principal dependent (response) variables. Several independent (explanatory) variables consisting of: 1) substrate type and composition; 2) substrate organic content; 3) benthic invertebrate abundance; 4) depth; 5) temperature; 6) salinity; 7) diet; and 8) trawling activity were also measured. The answers to the above questions, through the analysis of these variables, comprise the content of the subsequent chapters of this dissertation.

Chapter 1 describes the composition of the overall fish community in the study area, and its spatial and temporal variation in composition and community structure. The second chapter specifically describes the spatial and temporal distribution patterns, and the abundance of juveniles of the four fish species. These patterns are analyzed with respect to relevant explanatory variables. Given sufficient sample sizes of juvenile *H. platessoides* by sampling date and site compared to the other three species, its population structure, habitat and trophic ecology is described with respect to relevant explanatory variables in Chapter 3. Finally, with the foundation laid by the previous chapters, Chapter 4 integrates and synthesizes all the relevant data and results from these chapters and examines broader ecological impacts of shrimp trawling on juvenile benthic fish within a 'trawling-impact' context. It also provides a list of recommendations for potential application in fisheries management, using a community and ecosystem approach.
CHAPTER 1

FISH COMMUNITY OF THE SOUTHWESTERN GULF OF MAINE

Abstract

The overall fish community in the study area was composed of 24 species in 12 families. There was a clear distinction in the composition of the fauna at the two sites. Site 1 had a greater diversity of fish than Site 2. It was dominated by longhorn sculpin, Myxocephalus octodecemspinosus, while American plaice, Hippoglossoides platessoides was dominant in Site 2. Red hake, Urophycis chuss was equally dominant at both sites. Based on temporal persistence, different species were classified as rare or transient, seasonal, or resident within each site. Besides this spatial difference in the fish fauna, there was a greater influx of fish at the two sites in summer than in the winter. This seasonal dynamic was more equitable in Site 2 than in Site 1, as indicated by the more stable trend in the species equitability index, E.

Of the four species selected for this study, juvenile H. platessoides were the most abundant, followed by silver hake, Merluccius bilinearis. They were both more abundant in Site 2 than in Site 1. Winter flounder, Pseudopleuronectes americanus was rare, caught usually in summer, while only one Atlantic cod, Gadus morhua was caught at Site 1 in the summer of 1998.

Correlation analysis showed that the abundance of the two predominant species, M. octodecemspinosus (Site 1) and H. platessoides (Site 2) was predictable by a model.
that consisted of eight environmental variables ordered into three principal components (PC1-PC3) as predictors. The principal components represented composite variables accounting for a depth-sediment type attribute, a temperature-salinity attribute, and an epibenthic macrofauna attribute, respectively. Of these composite variables, the overall correlation of both species was high with PC2 ($r_c = 0.921$). However, PC2 significantly explained the variation in the abundance only of *M. octodecemspinosus* ($F_{1,4} = 21.703$, $p = 0.010$), and not *H. platessoides* ($F_{1,4} = 0.235$, $p = 0.653$).

**Introduction**

The direct impact of removing fish biomass through fishing is well known in fisheries science (Beverton, 1994; Botsford, et al., 1997; Fogarty and Murawski, 1998; Pauly et al., 1998). It is the leading cause of overexploitation of nearly 30% of 176 major fish stocks currently tracked by FAO (Knauss, 1994). On the other hand, certain types of fishing gear can themselves cause indirect adverse impacts on fish populations through habitat destruction (Caddy, 1973; Langton and Robinson, 1990; Messieh et al., 1991; Bergman and Hup, 1992; Brylinsky et al., 1994; Kaiser and Spencer, 1996). Jennings and Kaiser (1998) recently reviewed our knowledge of these two types of impact of fishing on the world's marine ecosystems.

Studies of indirect impacts of fishing, especially those due to the impact of mobile fishing gear, have almost always assessed its impact on benthic invertebrate communities (Bergman and Hup, 1992; Kaiser and Spencer, 1996; Brylinsky, et al., 1994; Collie et al., 1997; Engel and Kvitek, 1998; Simboura et al., 1998). Such studies are done with the understanding that benthic macroinvertebrates are the principal food sources of many
benthic fish and any impact on their populations will indirectly affect benthic fish populations too. However, few studies (e.g. Philippart, 1998) have assessed the direct impact of fishing gear on fish communities.

Fish community studies are popular in assessing aquatic pollution and habitat degradation in general. Because fish are sensitive to most forms of human disturbance, and also represent multiple trophic levels, they are a good taxon to assess anthropogenic impacts in aquatic environments (Harris, 1995). A number of studies were done in the 1970s to investigate the impact of organic and thermal pollution in estuarine and coastal marine ecosystems (Bechtel and Copeland, 1970; Haedrich and Haedrich, 1974; Livingston, 1975; Hilman et al., 1977). More recent fish-community impact studies in marine ecosystems include investigations to determine the impact of sewage outfall (Musick et al., 1996; Otway et al., 1996), toxic pollutants (Hartwell et al., 1997; 1998; Summers et al., 1995; Strobel et al., 1999), and thermal pollution from nuclear power generation (Thoernqvist, 1999). Depending on the scope and the goals of such studies, the focus can often be on whole communities of fish and benthic invertebrates or on an objectively selected subset of species within these two major taxonomic groups.

To study anthropogenic or natural perturbations in ecological systems, where biotic and abiotic components are both impacted, requires studies to be undertaken at community or higher levels of ecological organization. Such an approach in studying the impacts of mobile fishing gear such as trawls will enable an adequate understanding of any adverse impact of the gear with habitat components. Sampling multiple variables, both biotic (e.g. benthic invertebrate communities) and abiotic (physical and chemical), will provide the necessary data to analyze directly the processes involved, or where this is...
not possible, provide some insight into any underlying processes that might be involved. Subsequently, the mechanisms involved in transmitting the impact(s) can be understood, revealing ecological routes through which this takes place within the ecosystem. The functional role of the impacted habitat to the ecology of the fish communities, and broader ecological processes within the respective ecosystems, can then be understood, enabling suitable management decisions to be made. Such an approach is the broad objective of this study.

An ecological "community" can be defined as an assemblage of two or more species with different kinds of population fluctuation patterns and interactions with each other (Whittaker, 1975; Morin, 1999). Results from biogeographic studies support the notion that a community is not simply an assembly of species that colonized a habitat and adapted to the physical environment, but is also an entity that resulted from the population dynamics and interaction among its constituent species (Roughgarden, 1989). Interspecific interactions such as competition, predation and mutualism (Morin, 1999), together with the influences of the abiotic environment (Dayton, 1984; Roughgarden, 1989), are fundamental to understanding many of the processes responsible for the temporal and spatial structure existing in ecological communities. However, the specific mechanisms involved may vary among communities (Dayton, 1984).

The community is the interface between population and higher levels of ecological organization, and studying it would enable data relevant at both population and community levels to be collected concurrently, allowing a more realistic interpretation of the processes investigated. A common approach for community-level studies is the use of diversity indices (Peet, 1974; Magurran, 1988; Krebs, 1989; Morin,
1999) to study the structure of a given community. Many of these indices account for both species richness and the relative abundance of all the constituent species within a community (Poole, 1974; Magurran, 1988). Results of such studies allow ecologists to either interpret ecological processes that may be responsible for the observed structure of a given community (Whittaker, 1975; Morin, 1999), or enable them to formulate hypotheses to further test ecological theory (Morin, 1999).

Analyzing the structure of the benthic fish community in my study area will enable some understanding of the potential mechanisms that may be involved in its establishment. This will allow me to explain the distribution and abundance patterns of the four species studied with respect to the abiotic and biotic variables measured in my study within the specific context of any impacts of shrimp trawling. Consequently, the impacts of mobile gear, and any other natural or anthropogenic perturbation, on benthic invertebrate and fish communities can be better understood. This will contribute to increasing the awareness and understanding of the important role of habitat attributes in enhancing benthic fish recruitment, and potentially incorporate this understanding in better fisheries and habitat management policies.

**Materials and Methods**

**Study Area**

The study area was located in the southwestern Gulf of Maine, south of the Isles of Shoals, New Hampshire, covering a total area 4 km x 4 km (Figure 1.1). Two study sites, each 1 km x 1 km, were selected within this area during the summer of 1997 using a 1: 80,000 scale nautical chart (Chart No.13278, Portsmouth to Cape Ann) (US National
Figure 1.1. Map of the study area (large square) showing study sites (smaller filled squares) inside the large square. The top filled square is Site 1 and the bottom one is Site 2. The first four depth contours are in 5 m intervals while the rest are in 10 m intervals. (Source: Edited version of a base map obtained from Dr. Frank Bub of EOS, University of New Hampshire).
Oceanic and Atmospheric Administration (NOAA), 1993), and by consulting Dave Goethel, an experienced local fisherman. Site 1 (42° 55.36' N, 70° 36.90' W) was located within an area not trawled by shrimp trawlers because the size of the towable area at this site was too small for commercial shrimp trawling (Goethel, pers. comm.), which is usually 2-3 hours long (Ross and Hokenson, 1997). It was classified as the 'no trawl' site.

Site 2 (42° 53.77' N, 70° 37.96' W) was located within the fishing ground and classified as the 'trawl' site.

Site 2 was located within the inshore margin of the winter fishery for the northern shrimp, *Pandalus borealis*, in the southwestern Gulf of Maine. This area is part of a larger fishing ground within the Gulf of Maine that extends from the Stellwagen Bank off Massachusetts to the waters of central Maine, (Haynes and Wigley, 1969; Howell and Langan, 1992; Ross and Hokenson, 1997). This fishery occurs typically from December to May, and has been in existence since 1935 (Haynes and Wigley, 1969). It is based on the inshore spawning migration of the ovigerous females (Haynes and Wigley, 1969). In recent years a "rolling closures" fisheries management policy had been put in place. The policy requires inshore fishing grounds for groundfish to be progressively closed during the spring months with increasing latitude along the New England coast. Consequently, some New Hampshire fishermen cue in on these in-coming shrimp stocks during February and March (Dobbs, 2000) to maintain their fishing incomes.

Furthermore, the general bathymetric and substrate information for the two sites were checked from a chart of the 1:100,000 scale "Bathymetric Fishing Map" series (NOAA, 1986) in order to ensure that the two sites were similar in all physical aspects as
much as possible, except trawling activity. The two sites were located between the 60 m and 70 m bathymetric contours (Figure 1.1). Site 1 was located in shallower water at an average depth of 65 m, while Site 2 was in an average depth of 71 m (Table 1.1). The two sites were separated horizontally by 1.19 km, and interspersed between them was a stretch of shallower water of 37.8 m at its shallowest depth.

Geologically, the study area is within Unit 3 of sedimentary units described by Birch (1984, 1989, 1990) for this area, containing surface sediments consisting largely of glacial-marine deposits that are part of the Presumpscot Formation. These sediments are presumed to be Wisconsinan to Holocene in age (Birch, 1990).

**Benthic Habitat Attributes**

Before any impact of shrimp trawling on juvenile fish can be analyzed, it was necessary to quantify natural habitat attributes that can also influence distribution and abundance patterns of fish communities. Consequently a number of variables that were considered to constitute important habitat features were measured.

**Sediment Particle Size.** Sediment particle-size composition was analyzed from samples obtained with a *Shipek* grab sampler. Given the high mud content, a technique based on the "Wet-Sieve, Moisture-Replicate" method described in Folk (1980) suitable for sediments containing > 40 % mud was used. Two replicates of 15 g samples, accurate to 0.1g, from each of three grabs for each site were weighed on an electronic balance (*OHAUS*® Precision Standard Electronic Balances: *TS series*). They were then wet-sifted to separate the gravel, sand and mud fractions, using sieves of appropriate meshes. These particle size categories were separated according to the *Wentworth* size classes (gravel ≥ 2.0 mm; < 2.0 mm sand ≥ 0.0625 mm; mud < 0.0625 mm) (Wentworth, 1922; Folk...
1980). Subsequently, the respective fractions were carefully back-washed from the sieves into pre-weighed beakers that were dried in an oven at 50 °C for six hours. They were allowed to stand for six hours to allow sedimentation, and then the water was carefully siphoned out. The beakers with the sediment fractions were then re-dried at the same temperature for a further six hours. After this period the beakers, now with the respective sediment fractions, were re-weighed. The dry weight of each particle size fraction was determined by difference, using the beaker-only and beaker-and-sediment dry weights, and from these, the percentage of each fraction was calculated. A mean percentage for each size fraction was calculated from the two replicates and these were recorded as the estimate of the proportion of gravel, sand and mud in each grab. All such means for three grabs per site were used to calculate a grand mean per site for each sampling date. Where parametric statistical techniques were used in subsequent statistical analyses, an arc-sin transformation (\( \sin^{-1}(\sqrt{p}) \)), where \( p \) = proportion of particle size category) (Sokal and Rohlf, 1995; Zar, 1996; Underwood, 1997) was done to render the data appropriate for such analyses.

**Temperature and Salinity.** Temperature and salinity data with depth were collected with a *SEACAT SBE 19-03®* conductivity, temperature and depth (CTD) recorder, one cast per site per sampling date at about the same time on each date. The data was downloaded from the CTD in the laboratory using the accompanying software, *SEASOFT* (Version 4.218), run from a *DELT®* PC computer and saved as raw data files (*.HEX files). Using instructions in the software manual (Sea-Bird Electronics, Inc., 1996), these files were used to select the down-cast part of the file (the up-cast is usually too fast to be reliable) was then averaged into depth bins. The means for each bin for the
variables measured were then saved as converted files with an appropriate file extension (*.CNV files). These files were finally imported into Excel and used for further statistical and graphical analysis.

**Epibenthic Macroinvertebrates.** Certain benthic biotic features also constitute habitat components useful to juvenile fish, either as features suitable for refuge, background camouflage, and as food sources. The presence of these organisms can also serve as indicators of bottom type. Hence, all such material incidentally caught in the trawls were also recorded, as a means to get an idea of the bottom environment from which the fish were sampled. Sea stars, brittle stars, mysids, pandalid shrimps and the sand shrimp, *Crangon septemspinosa* were the principal groups incidentally caught. Sea stars and *C. septemspinosa* were used in a correlation analysis with other environmental variables to see if any relationships existed between the predominant fish species and the selected environmental variables representing both the biotic and abiotic attributes of the benthic habitat. This was done by using their means per three trawls for each sampling date for each site to first determine composite variables together with other environmental variables in a principal component analysis (PCA), as described in detail below.

**Fish Sampling**

Fish were sampled from the two sites during the winter and summer of 1998 and 1999. Winter samples were taken in late February while the summer samples were taken in late July of each year. A 16 foot (4.88 m) otter trawl with 2 inch (6.10 cm) mesh, and a cod end mesh of 1 1/2 inch (4.57 cm) with an inner lining of 1/4 inch (0.76 cm) was used. Each trawl was towed from the stern of the *R/V Gulf Challenger* using a hydraulic winch.
The duration of each tow, measured as the time elapsed between the locking of the winch to the beginning of hauling, was 10 minutes at a speed of 2 knots (1.03 m/s). Given this duration, speed and a ground rope width of 4.88 m, each tow is calculated to sweep an area of 3,015.84 m². To visually comprehend this figure in terms of physical space, it is an area equivalent to half the size of an average soccer field.

After 10 minutes, the net was winched in and guided onto the deck sample-sorting area by two persons, each taking one of the two otter boards. The catch was emptied into fish boxes and all fish and incidentally caught material (benthic invertebrates and other material such as gravel) were sorted and bagged into labeled (sampling site, trawl number and sampling date) heavy-duty trash bags. Upon return to shore, the samples were transported to the laboratory and frozen immediately.

**Fish Processing**

In the laboratory, fish were thawed at room temperature and rinsed clean with tap water. Fish were sorted into plastic trays and identified to species. Their weight, standard length, and either total or fork length, whichever was appropriate, were measured. Length was measured with a standard fish measuring board, made of a 1 m ruler fixed onto an L-shaped board. Weight for each fish was measured after blotting dry any excess water, using an *OHAUS®* electronic balance, accurate to 0.1 g. Subsequently, all the study species were dissected to collect gut, otolith and scale samples. Where possible, especially for sub-adult fish, sex and gonad development stage was recorded by quickly viewing samples under a dissecting microscope. The data was recorded onto data sheets and later entered as *EXCEL* spreadsheet data files.
Data Analysis

The structure of the fish community in the study area was analyzed by summarizing and performing various analyses on the raw catch data presented in Table A1 (Appendix). To simplify this dataset by pooling replicate trawls of each sampling date (i.e. season) within each site, it was necessary to test if pooling was justified. Count data often fit a Poisson distribution, hence, the total number of fish caught per trawl was square-root transformed ((n+1)^0.5) to meet the ANOVA assumption of independence between the mean and variance (Sokal and Rohlf, 1995; Zar, 1996; Underwood, 1997). Using a balanced ANOVA design, where the factors; site, year, and season were fixed while trawl (nested within site) was treated as a random variable, the analysis was performed with the MINITAB statistical package. The effect of all three factors, as well as interactions among them, were tested. There were no significant differences in total numbers of fish caught among the trawls (F[4] = 0.68; p = 0.621), justifying pooling the catches of replicate trawls (three per sampling date per site) for further analysis.

The pooled data were examined closely to analyze which species in the community were numerically dominant each season at each site and which were not, in order to characterize the fish fauna at each site. To ensure objectivity in this process a cluster analysis was also done with the SYSTAT statistical software package. A dendrogram produced by the analysis and the pooled catch data were both used to determine which species characterized the fish communities at the two study sites.

To determine temporal and spatial dynamics in community structure, the species richness index, S, the Shannon-Wiener (also Shannon-Weaver in some texts) diversity index, H', and evenness index, E, were calculated. For each sampling date and site these...
indices were calculated using a macro written with **MINITAB**. The specific formula for these indices were as follows:

\[
S = \text{total number of species recorded for each sample};
\]

\[
H' = - \sum p_i \ln p_i \quad \text{(Equation 1.1)}
\]

where \( p_i = \text{the proportion of the } i\text{th species in the sample} \) \((p_i = n_i / N)\);

\( n_i \) is the total number of individuals of the \( i\text{th} \) species;

\( N \) is the total number of individuals of all species in the sample.

\[
E = H' / \ln S \quad \text{(Equation 1.2)}
\]

where, the natural log of the number of species in the sample, is equal to the maximum diversity of the community attainable if all the species in the community were equally represented numerically (i.e. \( \ln S = H_{\text{max}} \)) (Peet, 1974).

The values for each index were then plotted with **EXCEL** to reveal site and seasonal trends in the changes in the three indices. Comparisons of any differences and similarities between the two sites for each sampling season in conjunction with pooled catch data were done to enable interpretations of the spatial and temporal patterns of the structure of the fish community in the study area.

In addition to the above analyses, a canonical correlation analysis (CANCORR) was done to analyze if the observed temporal and spatial patterns in the abundance of the resident species were correlated to any environmental factors. Generally, correlation is a technique intended to measure the degree of association between variables on the left side of the correlation equation with those on the right. Often there is no *a priori* intention of predicting those on the left from those on the right (Kachigan, 1982; Sokal and Rohlf, 1995; Zar, 1996). However, for clarity and reference to the respective sides of the
correlation equation, and consistent with usage in Tabachnick and Fidell (1996), I will use the " DVs" and " IVs" terminology to refer to variables on the left and right hand side of the correlation model respectively, used in this analysis.

Prior to the analysis, the normality of variables was tested in MINITAB using the Anderson-Darling normality test, a goodness-of-fit test based on an empirical cumulative distribution function (Minitab, 1998), even after having appropriately transformed the raw data. The two predominant species in each site were selected as the dependent variables (DVs). Eight environmental variables, both abiotic and biotic, were selected as independent variables (IVs). To select a few components from these variables representing independent sets of highly correlated variables, principal component analysis (PCA) was performed in MINITAB. This was necessary in order to reduce the number of IVs in the model relative to sample size (n = 8 in this case; 2 sites x 2 seasons x 2 years (Table 1.3)). Spurious inflation of the coefficient of determination (R²) can occur as the number of IVs are increased relative to a given sample size, even if no correlation exists between the DVs and IVs (Lewis-Beck, 1980; McCullagh and Nelder, 1989; Mittlböck and Waldhör, 2000). Also, among other techniques, this is one used to minimize multicollinearity among variables (Green, 1979; Brook and Arnold, 1985). This arises as a result of highly correlated variables among and/or between the DVs and IVs entering the model in a CANCORR, or any other multivariate models based on the general linear model (GLM) for that matter, violating the underlying assumptions of this method (Tabachnick and Fidell, 1996; McGarigal et al., 2000). Multicollinearity can also pose a problem to matrix computation because strong correlations among predictor variables can prevent the calculation of the inverse matrices during the analysis, as these
would be singular (i.e. the determinants would be close to zero) (Green, 1979; Poore and
Mobley, 1980; Brook and Arnold, 1985).

Based on their eigenvalues and a scree plot of these values, the first three
principal components of the environment variables were selected. The magnitude of the
loadings on the original variables in these components were used to gain some insight
into what environmental process each component represented in terms of the combined
effect of the original environmental variables that loaded high on the component. The
three principal components were then used as the IVs with the two predominant fish
species from each of the two sites treated as the DVs and analyzed with SYSTAT.

Results

Results of the analysis of sediment particle-size composition from grab samples
taken during each sampling date, two sub-samples analyzed for each of the three grabs
sampled per site per sampling date, are presented in Table 1.1. These data showed that
Site 1 contained more sand (54.96%) than mud (32.38%), while this trend was reversed
in Site 2 (Table 1.1). A fair amount of gravel-sized particles (12.66%) was present in
Site 1, but it was almost absent in Site 2 (Table 1.1). The average depths at the study sites
indicated that Site 2 is deeper than Site 1 by about 6 m on average.

The hydrographic data obtained with the CTD indicated that the water column
was more well mixed in the winter than in the summer (1998 data). Despite differences in
bathymetry and sediment composition between the two sites, there was little
hydrographic difference between the two sites. However, there were differences
temporally. The water temperature was colder, with higher salinity in the winter. This
Table 1.1. Mean depth and sediment particle size (defined below *) composition for the study sites. The particle-size values are grand means (n = 12) and ± standard error of mean (SEM) for duplicate sub-samples from three replicate grabs sampled at each site per sampling date over two winters and two summers in 1998 and 1999. The grand mean of depth of each site is based on means (n = 4) of all depth measurements taken with each of the three replicate grabs for each sampling date pooled by site.

<table>
<thead>
<tr>
<th>Site</th>
<th>Depth (m) (± SEM)</th>
<th>% Gravel (± SEM)</th>
<th>% Sand (± SEM)</th>
<th>% Mud (± SEM)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>64.76 (0.39)</td>
<td>12.66 (4.92)</td>
<td>54.96 (3.38)</td>
<td>32.38 (2.09)</td>
</tr>
<tr>
<td>2</td>
<td>70.84 (1.55)</td>
<td>0.18 (0.08)</td>
<td>42.01 (1.66)</td>
<td>57.81 (1.66)</td>
</tr>
</tbody>
</table>

* The sediment size classes are defined based on particle size classes according to the Wentworth scale (Wentworth, 1922; Folk, 1980): gravel ≥ 2.0 mm; < 2.0 mm sand ≥ 0.0625 mm; mud < 0.0625 mm.
pattern was reversed in the summer (Figure 1.2). Seasonal stratification was also apparent, and occurred at about 15 m, as indicated by the thermocline and halocline for temperature and salinity, respectively (Figure 1.2). An overall summary of the variation in the two variables with depth and site is given in Table 1.2. Both surface and bottom summer temperatures for the two years indicated that 1999 was comparatively warmer than 1998.

Qualitatively, the bycatch data of epibenthic macroinvertebrates caught incidentally in the trawls indicate some spatial differences in distribution of some taxa. There seems to be a greater occurrence of starfish and brittle stars in Site 1 than in Site 2. This pattern is reversed in the occurrence of the sand shrimp, *Crangon septemspinosa* (Table 1.3).

The fish community of the southwestern Gulf of Maine sampled during the summer and winter of 1998 and 1999 was comprised of twenty-four species from twelve families (Table 1.4; Table A1, Appendix). It is apparent from the pooled catch data (Table 1.4) that there were site differences in the relative abundance of most of the species. The within season, between-site, comparison of species abundance showed that there were three major groupings among the fish in the community. Most of the species were either rare or were transients. These were species that were caught once among sampling dates within or between sites. The next group was the seasonals; they were species caught in one or both sites only in the summer or winter. Finally, there were residents. These were species that were caught throughout all the sampling dates (two summers and two winters) at either both or one of the sites. However, their numbers varied both seasonally and spatially. Species that were caught in all except one sampling
Figure 1.2. Profile of water temperature (°C) and salinity (psu) in winter (open squares) and summer (filled diamonds) with depth (decibars) at Site 2 in 1998. Both variables were measured with a CTD unit. The unit for depth is in decibars (1 dbar = 0.99 m). Salinity was measured in "practical salinity units" (psu), which is equal to the conventional unit, ppt.
Table 1.2. Surface and bottom temperature and salinity at the study sites in winter (late February) and summer (late July) of 1998 and 1999. Depth is recorded in decibars (1 dbar = 0.99 m descend in depth). The times at which each CTD cast was made are listed.

<table>
<thead>
<tr>
<th>Site</th>
<th>Season</th>
<th>Time (hrs.)</th>
<th>Depth (dbar)</th>
<th>Temperature (°C)</th>
<th>Salinity (psu)*</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Winter '98</td>
<td>08.10</td>
<td>2</td>
<td>3.26</td>
<td>31.58</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>50</td>
<td>3.55</td>
<td>31.71</td>
</tr>
<tr>
<td></td>
<td>Summer '98</td>
<td>07.39</td>
<td>1</td>
<td>16.28</td>
<td>30.18</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>50</td>
<td>4.86</td>
<td>31.59</td>
</tr>
<tr>
<td>2</td>
<td>Winter '98</td>
<td>09.49</td>
<td>1</td>
<td>3.15</td>
<td>31.24</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>61</td>
<td>3.58</td>
<td>31.72</td>
</tr>
<tr>
<td></td>
<td>Summer '98</td>
<td>09.04</td>
<td>1</td>
<td>17.31</td>
<td>30.04</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>61</td>
<td>4.48</td>
<td>31.59</td>
</tr>
<tr>
<td>1</td>
<td>Summer '99</td>
<td>08.00</td>
<td>1</td>
<td>18.67</td>
<td>31.44</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>50</td>
<td>6.36</td>
<td>32.07</td>
</tr>
<tr>
<td>2</td>
<td>Summer '99</td>
<td>07.54</td>
<td>1</td>
<td>18.79</td>
<td>31.53</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>61</td>
<td>6.12</td>
<td>32.10</td>
</tr>
</tbody>
</table>

+ Data for winter 1999 was lost due to a broken cable connector pin on the CTD.
* psu is "practical salinity units" which is essentially equal to the conventional unit, ppt.
Table 1.3. Total numbers of principal taxa of epibenthic macrofauna caught as bycatch in three replicate trawls of different sampling dates at the two study sites.*

<table>
<thead>
<tr>
<th>Taxa</th>
<th>Common name</th>
<th>Site 1</th>
<th>Site 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Echinodermata</td>
<td>Starfish</td>
<td>1</td>
<td>9</td>
</tr>
<tr>
<td></td>
<td>Brittle stars**</td>
<td>30</td>
<td>207</td>
</tr>
<tr>
<td>Arthropoda (Crustacea)</td>
<td>Northern shrimp</td>
<td>13</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Other pandalid shrimps</td>
<td>21</td>
<td>458</td>
</tr>
<tr>
<td></td>
<td>Sand shrimp (Cragon)</td>
<td>1</td>
<td>4</td>
</tr>
<tr>
<td></td>
<td>Mysid shrimps</td>
<td>10</td>
<td>0</td>
</tr>
</tbody>
</table>

* Being bycatch, it should be borne in mind that otter trawls are not designed to sample most of these taxa.

** Brittle stars were largely associated with the occurrence of gravel (see Table 1.1), where crevices in pieces of gravel were their preferred microhabitat.
Table 1.4. Numbers of each fish species caught by site and sampling date. Data from three replicate trawls were pooled for each sampling date within each site. Diversity indices calculated from this data are included at the bottom of the table.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Rajidae</td>
<td><em>Raja erinacea</em></td>
<td>Little skate</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Clupeidae</td>
<td><em>Alosa aestivalis</em></td>
<td>Blueback herring</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>17</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td><em>Alosa pseudoharengus</em></td>
<td>Alewife</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>3</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td><em>Clupea harengus harengus</em></td>
<td>Atlantic herring</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>3</td>
<td>0</td>
<td>9</td>
</tr>
<tr>
<td>Lophidae</td>
<td><em>Lophius americanus</em></td>
<td>Monkfish</td>
<td>0</td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>3</td>
<td>0</td>
<td>1</td>
<td>7</td>
</tr>
<tr>
<td>Gadidae</td>
<td><em>Enchelyopus cimbrius</em></td>
<td>Fourbeard rockling</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>2</td>
<td>5</td>
</tr>
<tr>
<td></td>
<td><em>Gadus morhua</em></td>
<td>Atlantic cod</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td><em>Merluccius bilinearis</em></td>
<td>Silver hake</td>
<td>0</td>
<td>2</td>
<td>3</td>
<td>6</td>
<td>8</td>
<td>11</td>
<td>0</td>
<td>9</td>
</tr>
<tr>
<td>Pollachius vires</td>
<td>Pollock</td>
<td>0</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Urophycis chuss</td>
<td>Red hake</td>
<td>1</td>
<td>1</td>
<td>32</td>
<td>1</td>
<td>27</td>
<td>7</td>
<td>48</td>
<td>0</td>
<td>27</td>
</tr>
<tr>
<td>Labridae</td>
<td><em>Tautoga ornitis</em></td>
<td>Tautog</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td><em>Tautogolabrus adspersus</em></td>
<td>Cunner</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Zoarcidae</td>
<td><em>Macrozoarcus americanus</em></td>
<td>Ocean pout</td>
<td>0</td>
<td>4</td>
<td>0</td>
<td>12</td>
<td>0</td>
<td>46</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Scolidae</td>
<td><em>Lumpenus lumpreaformis</em></td>
<td>Snakebenny</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td><em>Lumpenus maculatus</em></td>
<td>Daubed shanny</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Cryptacanthodidae</td>
<td><em>Cryptacanthodes maculatus</em></td>
<td>Wrymouth</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Cottidae</td>
<td><em>Hemitripterus americanus</em></td>
<td>Sea raven</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>6</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td><em>Myoxocephalus octodecemspinus</em></td>
<td>Longhorn sculpin</td>
<td>15</td>
<td>102</td>
<td>64</td>
<td>28</td>
<td>14</td>
<td>7</td>
<td>4</td>
<td>0</td>
</tr>
<tr>
<td>Agonidae</td>
<td><em>Aspidophoroides monopterygius</em></td>
<td>Alligatorfish</td>
<td>2</td>
<td>12</td>
<td>2</td>
<td>8</td>
<td>0</td>
<td>6</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Bothidae</td>
<td><em>Scophthalmus aquosus</em></td>
<td>Windowpane</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Pleuronecidae</td>
<td><em>Cryptocaes cynoglossus</em></td>
<td>Grey sole</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>2</td>
<td>1</td>
<td>4</td>
</tr>
<tr>
<td>H. platissoides</td>
<td>American plaice</td>
<td>1</td>
<td>37</td>
<td>8</td>
<td>35</td>
<td>29</td>
<td>127</td>
<td>13</td>
<td>79</td>
<td></td>
</tr>
<tr>
<td>Limanda ferruginea</td>
<td>Yellowtail flounder</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Sum</td>
<td>27</td>
<td>262</td>
<td>84</td>
<td>129</td>
<td>61</td>
<td>238</td>
<td>19</td>
<td>142</td>
<td>6</td>
<td>15</td>
</tr>
<tr>
<td>Diversity Index</td>
<td>S</td>
<td>1.0504</td>
<td>1.8174</td>
<td>0.9589</td>
<td>1.9338</td>
<td>1.3856</td>
<td>1.6108</td>
<td>0.8976</td>
<td>1.4281</td>
<td>E</td>
</tr>
</tbody>
</table>
date at one or both sites were also included in this category. Silver hake, *Merluccius bilinearis*, was a good example of such species.

Of the residents, seven species were recorded in Site 1. They included silver hake, *Merluccius bilinearis*, red hake, *Urophycis chuss*, ocean pout, *Macrozoarces americanus*, sea raven, *Hemitripterus americanus*, longhorn sculpin, *Myoxocephalus octodecemspinosus*, alligatorfish, *Aspidophoroides monopterygius*, and American plaice, *Hippoglossoides platessoides*. Of these species, *M. octodecemspinosus* was the predominant species in the catch (numerically) in both seasons (Table 1.4). In Site 2, only four resident species were caught. They were *M. bilinearis*, *U. chuss*, *M. octodecemspinosus*, and *H. platessoides*. *H. platessoides* predominated in the catches in both seasons at this site (Table 1.4). Although *M bilinearis*, *U. chuss*, *M. octodecemspinosus* and *H. platessoides* were the residents that occurred at both sites, they were more abundant at one of the two sites, except *U. chuss* that was almost equally abundant at both sites.

Results of cluster analysis also confirmed this grouping of taxa, clearly separating the predominant species, *M. octodecemspinosus* (Site 1) and *H. platessoides* (Site 2) (Figure 1.3). The fact that the predominance of *U. chuss* was similar at both sites is also evident in Figure 1.3. Apparently, *M. americanus*, *H. americanus*, and *A. monopterygius*, that were among the resident species of Site 1, also separated out as a cluster in this analysis (Figure 1.3).

To understand the spatial and temporal dynamics of the community structure of the fish community in the two sites, it was necessary to look at the spatial and seasonal pattern in the three diversity indices. These are the species richness index, $S$, the species
Cluster Tree

Figure 1.3. Dendrogram of cluster analysis of all fish species caught in the two study sites based on square-root transformed \((n+1)^{0.5}\) catch data. "Distances" are Euclidean distances based on the average linkage method of defining inter-cluster distance. Species codes for each species used in the cluster tree are composed of the first three letters of their genus and species names, respectively, listed in Table 1.3. The predominant species in Site 1 were: MYOCT = Myoxocephalus octodecemspinosus; MACAME = Macrozoarces americanus; and in Site 2 they were: HIPPLA = Hippoglossoides platessoides; UROCHU = Urophycis chuss (its predominance overlapped with Site 1 as well).
heterogeneity index, $H'$ and the species equitability index, $E$ (Peet, 1974; Magurran, 1988). Besides the spatial difference in the fish fauna described above, there is a greater influx of fish at the two sites in the summer than in the winter. This is apparent from the higher peaks in the species richness index $S$, and the Shannon-Wiener heterogeneity index, $H'$ (Figure 1.4). This seasonal dynamic is more equitable in Site 2 than in Site 1, as indicated by the more stable trend in the index $E$ (Figure 1.4).

Table 1.5 shows the results of the PCA showing the first three principal components that explained a total of 88.6% of the variation in the dimensions of the data set representing the eight environmental variables selected in the analysis. The first principal component (PC1) represented a bottom-characteristics (abiotic) axis in the data; the variables in this group were depth, gravel, sand, and mud. Both depth and mud loaded negatively (i.e. negative correlation) while sand and gravel indicated a positive correlation on the first principal axis (PC1). PC2 represented a water column-characteristics (abiotic, physicochemical) axis, with both temperature and salinity correlated positively on this axis. The third component (PC3) was a bottom epibenthic macroinvertebrate presence (biotic) axis with the abundance of the sand shrimp, *Crangon septemspinosa*, and sea star correlated positively on this axis (Table 1.5).

The results of canonical correlation analysis (CANCORR) of the three principal components as IVs with the two predominant fish species, *M. octodecemspinosus* (Site 1) and *H. platessoides* (Site 2) as the DVs (Table 1.6), showed that the overall model had predictive value. The canonical correlation coefficient ($r_c$) of the model was 0.962. Univariate $F$ tests on the multiple regression coefficients ($\beta$) (Table 1.7) of both DVs in the model were significant; *M. octodecemspinosus*, $F_{[1,4]} = 28.291$, $p = 0.006$, and
Figure 1.4. Spatial and temporal variation in the diversity indices of the fish community in the study area.
Table 1.5. Minitab results of principal component analysis on eight environmental variables to select principal components that encompassed correlated sets of composite variables (PCs) used as IVs in the canonical correlation analysis. Of the eight PCs that resulted, only three were selected based on their eigenvalues and a scree plot of these values. Part I of the table shows the results of the eigenanalysis, and Part II gives the loadings on the environment variables in each PC.

### I. Eigenanalysis of the Correlation Matrix

<table>
<thead>
<tr>
<th>Eigenvalue</th>
<th>4.3389</th>
<th>1.7069</th>
<th>1.0457</th>
</tr>
</thead>
<tbody>
<tr>
<td>Proportion</td>
<td>0.542</td>
<td>0.213</td>
<td>0.131</td>
</tr>
<tr>
<td>Cumulative</td>
<td>0.542</td>
<td>0.756</td>
<td>0.886</td>
</tr>
</tbody>
</table>

### II. Variable PCI PC2 PC3

<table>
<thead>
<tr>
<th>Variable</th>
<th>PC1</th>
<th>PC2</th>
<th>PC3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Depth</td>
<td>-0.473</td>
<td>-0.035</td>
<td>0.030</td>
</tr>
<tr>
<td>Temperature</td>
<td>-0.000</td>
<td>0.659</td>
<td>0.383</td>
</tr>
<tr>
<td>Salinity</td>
<td>-0.041</td>
<td>0.696</td>
<td>-0.040</td>
</tr>
<tr>
<td>Gravel</td>
<td>0.410</td>
<td>0.188</td>
<td>-0.342</td>
</tr>
<tr>
<td>Sand</td>
<td>0.441</td>
<td>-0.144</td>
<td>0.314</td>
</tr>
<tr>
<td>Mud</td>
<td>-0.456</td>
<td>-0.053</td>
<td>0.041</td>
</tr>
<tr>
<td>Crangon</td>
<td>-0.330</td>
<td>-0.052</td>
<td>0.488</td>
</tr>
<tr>
<td>Starfish</td>
<td>0.308</td>
<td>-0.138</td>
<td>0.629</td>
</tr>
</tbody>
</table>
Table 1.6. *SYSTAT* results of model estimation of canonical correlation between the predominant species, *M. octodecemspinosus* (MYOOCT) (Site 1) and *H. platessoides* (HIPPLA) (Site 2) as DVs and the three principal components (PC1-PC3, Table 1.4) as the IVs. The model used was MYOOCT HIPPLA = Constant + PC1 + PC2 + PC3.

<table>
<thead>
<tr>
<th>Aspect of model</th>
<th>β (Standardized)</th>
<th>R</th>
<th>R² (Adjusted)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Regression of DVs with (IVs)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>MYOOCT (PC1)</td>
<td>0.035</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>MYOOCT (PC2)</td>
<td>0.901</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>MYOOCT (PC3)</td>
<td>-0.082</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>HIPPLA (PC1)</td>
<td>0.151</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>HIPPLA (PC2)</td>
<td>-0.239</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>HIPPLA (PC3)</td>
<td>0.100</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Multiple correlation</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>MYOOCT</td>
<td>-</td>
<td>0.928</td>
<td>0.758</td>
</tr>
<tr>
<td>HIPPLA</td>
<td>-</td>
<td>0.319</td>
<td>0.000</td>
</tr>
</tbody>
</table>

β: Multiple regression coefficient.
Table 1.7. *SYSTAT* results of model effect significance tests for the canonical correlation model described in Table 1.5.

<table>
<thead>
<tr>
<th>Effect tested</th>
<th>$r_c$</th>
<th>$\lambda$</th>
<th>F[df]*</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>Constant ($\beta$)</td>
<td>0.962</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>MYOOCCT</td>
<td>-</td>
<td>-</td>
<td>28.291 [1,4]</td>
<td>0.006</td>
</tr>
<tr>
<td>HIPPLA</td>
<td>-</td>
<td>-</td>
<td>9.638 [1,4]</td>
<td>0.036</td>
</tr>
<tr>
<td>Variance overlap</td>
<td>-</td>
<td>0.075</td>
<td>18.506 [2,3]</td>
<td>0.021</td>
</tr>
<tr>
<td>PC1</td>
<td>0.206</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>MYOOCCT</td>
<td>-</td>
<td>-</td>
<td>0.035 [1,4]</td>
<td>0.860</td>
</tr>
<tr>
<td>HIPPLA</td>
<td>-</td>
<td>-</td>
<td>0.101 [1,4]</td>
<td>0.767</td>
</tr>
<tr>
<td>Variance overlap</td>
<td>-</td>
<td>0.957</td>
<td>0.067 [2,3]</td>
<td>0.937</td>
</tr>
<tr>
<td>PC2</td>
<td>0.921</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>MYOOCCT</td>
<td>-</td>
<td>-</td>
<td>21.703 [1,4]</td>
<td>0.010</td>
</tr>
<tr>
<td>HIPPLA</td>
<td>-</td>
<td>-</td>
<td>0.235 [1,4]</td>
<td>0.653</td>
</tr>
<tr>
<td>Variance overlap</td>
<td>-</td>
<td>0.153</td>
<td>8.327 [2,3]</td>
<td>0.060</td>
</tr>
<tr>
<td>PC3</td>
<td>0.214</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>MYOOCCT</td>
<td>-</td>
<td>-</td>
<td>0.182 [1,4]</td>
<td>0.692</td>
</tr>
<tr>
<td>HIPPLA</td>
<td>-</td>
<td>-</td>
<td>0.041 [1,4]</td>
<td>0.849</td>
</tr>
<tr>
<td>Variance overlap</td>
<td>-</td>
<td>0.954</td>
<td>0.072 [2,3]</td>
<td>0.932</td>
</tr>
</tbody>
</table>

$r_c$: Canonical correlation coefficient.

$(\lambda)$: Eigenvalue = the proportion of variance in the DVs that overlapped with IVs (or constant) other than the one whose effect was being tested.

* The third $F$-test in the column for each effect is a multivariate test testing the effect of both DVs simultaneously.
H. platessoides, $F_{[1,4]} = 9.638$, $p = 0.036$ (Table 1.7). The multiple correlation coefficients (R) pertaining to M. octodecemspinosus and H. platessoides were 0.928 and 0.319, respectively. The adjusted squared multiple correlation coefficients ($R^2$) of the model in explaining the variance in the DV, M. octodecemspinosus, was 0.758 and in the DV, H. platessoides, it was 0.0 (Table 1.6).

A multivariate significance test of the IVs in the overall model in adequately accounting for the total variance in the DVs was significant ($\chi = 0.075; F_{[2,3]} = 18.506, p = 0.021$) (Table 1.7). Besides testing the overall significance of the model, the amount of variation in the DVs explained by each IV was also analyzed. Of the three IVs only the second IV (PC2), representing a composite of a temperature-salinity component in PCA (Table 1.5), correlated highly ($r_c = 0.921$) with the DVs. This correlation, tested with a univariate F test, was significant in explaining the variance in M. octodecemspinosus ($F_{[1,4]} = 21.703, p = 0.010$), but not the variance in H. platessoides ($F_{[1,4]} = 0.235, p = 0.653$). This difference in significance between the two DVs was influenced by a fair degree of overlap in the variance of the IV and that of the two DVs since the test of overlap in variance was not significant ($\chi = 0.153; F_{[2,3]} = 8.327, p = 0.060$ (Table 1.7).

**Discussion**

Fish community studies are useful in revealing important natural patterns in distribution and abundance of species both spatially and temporally. Description of these patterns can potentially generate hypotheses about possible processes involved, thereby facilitating the study of mechanisms regulating them. Such an approach in understanding the structure of fish communities is useful in setting a basis from which to assess any
anthropogenic or natural perturbations impacting aquatic habitats. As reviewed in the 
introduction, fish community studies are popular in this regard (Bechtel and Copeland, 
1970; Haedrich and Haedrich, 1974; Livingston, 1975; Hilman et al., 1977; Summers et 
al., 1995; Musick et al., 1996; Otway et al., 1996; Hartwell et al., 1997; 1998; Strobel et 
al., 1999; Thoernqvist, 1999). This popularity is largely due to reasons given in Harris 
(1995) as outlined in the introduction above.

This study investigated whether there is any impact of shrimp trawling activity in 
the study area on the juveniles of selected benthic fish species. Such an objective can 
only be achieved if the broader natural patterns in community structure are understood, 
allowing impacts from perturbations to be assessed. This was the objective of this 
chapter.

The analyses revealed that the fish community within the study area (Figure 1.1) 
in the southwestern Gulf of Maine comprised of 24 species in 12 families. Classified and 
defined on the basis of persistence and abundance, three major groups of fish were 
present at the two study sites at any one time. These groups were rare or transient species, 
seasonal species, and the resident species (see results section for definitions of each 
group). However, the fish assemblages at the two sites within the study area were quite 
different, especially when considering the resident species. A total of seven species 
comprised the resident species in Site 1; *M. bilinearis, U. chuss, M. americanus, H. 
americanus, M. octodecemspinosus, A. monopterygius* and *H. platessoides*. The list for 
Site 2 comprised of only four species; *M. bilinearis, U. chuss, M. octodecemspiosus*, and 
*H. platessoides*. At the two study sites, *M. octodecemspinosus* was predominant at Site 1

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while *H. platessoides*, one of the selected species to investigate impact of trawling, was predominant at Site 2 (Table 1.4).

This difference in composition in the resident fauna at each site was quite marked. Consequently, it was even possible for one to guess which of the two sites were being sampled just by looking at the species composition and relative abundance in the catch. The results of a cluster analysis (Figure 1.3) showed evidence of the existence of such tight groupings of species. At the base of the cluster the two predominant species at the two sites, *M. octodecemspinosus* in Site 1 and *H. platessoides* in Site 2, were separated into different clusters. The fact that *U. chuss* was equally predominant at the two sites separated this species at this initial split in the dendrogram. A cluster at the next level starting from the third species code to the sixth listed from top to bottom on the dendrogram, comprised largely a cluster of resident species coded MACAME, HEMAME, ASPMON, and MERBIL (see Figure 1.3 caption for the method of coding in conjunction with Table 1.4). These species were *M. americanus*, *H. americanus*, *A. monopterygius*, and *M. bilinearis*, respectively. As listed above, the first three were among the resident species in Site 1 only while the fourth is also resident at both sites.

Such species groupings have been identified in many marine fish community studies. Tyler (1971) identified periodic and resident components in communities of Atlantic fishes, attributing much of the explanation of these groupings to seasonal variation in temperature. A study of distribution of juvenile groundfish in nearshore habitats in southwest Nova Scotia, Canada, reported both temperature and salinity acting in concert with water clarity and bottom substrate to influence abundance and distribution (Horne and Campana, 1989). Cluster analysis of demersal fish and macro-epibenthic
invertebrate assemblages at a number of sites in British waters enabled stations with similar species composition and abundance to be identified (Ellis et al., 2000). There was evidence in that study of fish species to form groups that correlated with depth, temperature, and substrate characteristics. A study in Sheepscot Bay, Gulf of Maine (Langton, 1998) demonstrated that small scale habitat features associated with sediment and depth, further correlated with suitable prey abundance, explained the spatial distribution of six dominant benthic fish species in that community. Two further studies done on mid-continental-shelf habitats similar to my study area, one in Oregon (Stein et al., 1992) and another in New York Bight (Steves et al., 1999) also reported similar grouping in the fish fauna with respect to substrate, temperature, salinity and depth regimes.

Canonical correlation analysis on the two predominant species at the two sites with composites of environmental variables ordered into principal components (PC1-PC3, Table 1.5) showed that the model (Table 1.6) had predictive value for both species (Table 1.7). That is, the model had potential to explain the variation in the abundance of predominant fish species as a function of a composite of environmental variables (abiotic and biotic) (Tables 1.5, 1.6 and 1.7). A univariate test of the slopes in the model was significant for both *M. octodecemspinosus* (Site 1), $F_{1,4} = 28.291, p = 0.006$, and *H. platessoides* (Site 2), $F_{1,4} = 9.638, p = 0.036$ (Tables 1.6 and 1.7). The results of the canonical correlation of composite environmental variables represented by the three principal components confirmed a strong correlation ($\tau_c = 0.921$) only in the salinity-temperature composite with the two predominant species (Table 1.7). This strong correlation significantly explained the variation in the abundance of the longhorn sculpin,
M. octodecemspinosus (F\[1, 4]\ = 21.703, p = 0.010) but not the variance in H. platessoides (F\[1,4]\ = 0.235, p = 0.653) (Table 1.7).

This result is not surprising since there is a more pronounced variation in the structure of the fish communities seasonally than spatially (Figure 1.4), and surface temperature and salinity were the two variables that varied dramatically at the two sites on a seasonal basis (Figure 1.2). Compared with the surface, seasonal differences at the bottom of approximately one unit (1 °C for temperature and 0.1 psu for salinity) (Table 1.2 and Figure 1.2) may seem insignificant. However, benthic fish seem to be very sensitive to this magnitude of seasonal change in the two variables at the bottom. This strong seasonal influence of temperature and salinity regimes may have masked any effects from other physical variables such as substrate and depth, whose effects operate on much smaller spatial scales (< 100s of meters) (Langton, 1998). The spatial scale of the sampling unit for fish in this study, the area swept by a 10-minute tow, was 3,015.84 m². This area would be approximately equal to half the size of an average soccer field. Tripling this area would give an estimate of the total area sampled by three replicate tows done randomly at each site per sampling date.

Besides this difference in predominance and species composition between the two sites, species richness and relative abundance was also different at the two sites. Noticeable from the trend in the diversity indices (Figure 1.4), Site 1 had a greater diversity than Site 2, as indicated by the index S in this figure. A temporal (seasonal) pattern was also apparent. Over the two winters and summers sampled, the diversity was usually higher in the summer months than in the winter months at both sites (Figure 1.4). The temporal dynamic was more equitable in Site 2 than Site 1, as indicated by the more
stable seasonal trend in the index E (Figure 1.4). The spatial disparity in this trend may be related to differences in the number of resident species, seven in Site 1 and four in Site 2. Intuitively, the more species there are the greater will be the variability in the influx and efflux among them within the community, both spatially and temporally. This observed seasonal pattern was due both to an influx of individuals of resident species, as well as the occurrence of a number of rare/transient species. The unstable trend in the equitability index E seasonally at Site 1 was due to the fact that this site is numerically dominated by its predominant species, *M. octodecemspinosis* in winter (Table 1.4 and Figure 1.4). This caused a 'swamping' effect on the community by just one species, resulting in a depression in equitability.

'Swamping' also occurred in summer at Site 2 due to an influx by *H. platessoides*, the predominant species at this site. As indicated by size (length) frequency analysis (see Chapter 3), this is due both to an influx of older year classes (Age 3+) from deeper waters, and a strong Age 1+ year class that settled in early fall the previous year. Consequently, its numbers are greatly increased compared to that during the winter (Table 1.4). Unlike Site 1, 'swamping' by the predominant species did not result in a depression in equitability (Figure 1.4) because of a stabilization effect from an influx of transient species, largely clupeids, during the summer (Table 1.4). Clupeids are pelagic species, and if they are excluded as part of the benthic fish community on this basis, I believe the effect will be similar to that at Site 1 in the winter. Recruitment, a process that can result in numerical predominance from strong year classes, can regulate the seasonal and temporal trends in community structure in a manner similar to the seasonal movement of older age classes. Howell and Langan (1992) reported that juvenile year
classes (Age 1+ to Age 2+) constituted a large proportion of this species discarded in the commercial shrimp fishery in this area of the Gulf of Maine. Therefore, recruitment of *H. platessoides*, besides seasonal influx, is an important process in regulating community structure in areas where this species is predominant.

Longhorn sculpin, *M. octodecemspinosus*, and American plaice, *H. platessoides*, being the dominant species at Site 1 and Site 2 respectively, have an important influence on the dynamics of the structure of the community. Where such species occur at a site where environmental impacts occur, the monitoring of their abundance and also a good understanding of their ecology can help to interpret temporal shifts in the community structure; monitoring diversity indices would be a quick way to detect such shifts.

Because *M. octodecemspinosus* was not selected using the criteria based on the abundance of commercial species as bycatch in the shrimp fisheries in this area recorded in previous studies (Howell and Langan, 1987, 1992), it was not included as one of the study species. However, data on its diet was collected in this study and is intended to be published, focusing on aspects of its ecology that are not focused upon in this study.

Also worth noting is the very rare occurrence of Atlantic cod, *Gadus morhua*, a very important commercial species. It is not possible to tell from this data whether this is due to low abundance because a small net was used in this study, and the period of active fishing by the gear was short (10 minutes) since juvenile fish were focused upon in the study. Furthermore, mobility and other behavioral aspects of the species can also influence its vulnerability to the gear.

As stated in the introduction, communities are not simply assemblages of species that have colonized a given habitat and adapted to the prevailing physical environmental...
conditions. They are also dynamic entities with interactions among its component species (Dayton, 1984; Roughgarden, 1989; Morin, 1999). Interspecific interactions such as competition, predation and mutualism play a role in determining the nature of communities (Morin, 1999). Whatever the processes are, the specific mechanisms involved may be different among different communities (Dayton, 1984). It seems that on seasonal time scales population-level processes such as recruitment pulses and ontogenetic differences in seasonal habitat preferences can also play a role in structuring fish communities consisting of species with seasonal reproductive patterns. This will become evident in Chapter 3, where the distribution and abundance patterns of *H. platessoides*, the most abundant species of the four selected study species, in relation to specifically selected environmental variables were analyzed. Understanding such recruitment-related predominance in fish communities had been suggested as an approach that would enable a firmer understanding of the existence of a linkage between the adult and larval habitats, rather than perceiving recruitment as a stochastic factor (Roughgarden, 1989, 1998). A better understanding of such processes interfacing both the population and community levels has useful applications in fisheries management. Finally, as shown in this chapter, a firm understanding of communities and their underlying structure is important in impact studies, or ecological studies focused on selected sets of species.
CHAPTER 2

DISTRIBUTION AND ABUNDANCE OF JUVENILES OF THE SELECTED BENTHIC FISH SPECIES

Abstract

Of the juveniles of the four species selected for this study, only two species occurred in sufficient densities for further analysis. *H. platessoides* had the highest densities at both sites followed by *M. bilinearis*. Both species occurred in greater densities at the two sites in summer than in winter. Univariate F-tests, using MANOVA, of both the spatial and temporal pattern were significant for *H. platessoides* ($F_{[1]} = 6.7518, p = 0.0483$ and $F_{[1]} = 12.4749, p = 0.0167$, respectively) but not for *M. bilinearis* ($F_{[1]} = 3.4323, p = 0.1231$ for both site and season). However, the results of multivariate F-tests testing both species (DVs) simultaneously for site and season effects were non-significant ($F_{[2,4]} = 2.7915, p = 0.1742$ and $F_{[2,4]} = 4.9998, p = 0.0816$, respectively).

There was also no significant correlation between the two species and the four principal components encompassing composites of environmental variables that were measured during the study in the field. The adjusted coefficients of multiple determination for both species were zero ($R^2 = 0.0$) with a canonical correlation coefficient ($r_c$) of 0.8545. The small spatial scale (4 km x 4 km) of the overall study is interpreted as a possible explanation of the lack of correlation between the two fish species and the measured environmental variables.
Introduction

This study investigated the ecological impact of shrimp trawling on juveniles of selected benthic fishes in the southwestern Gulf of Maine. Based on their abundance reported in previous studies (Howell and Langan, 1987, 1992) American plaice (*Hippoglossoides platessoides*), winter flounder (*Pseudopleuronectes americanus*), Atlantic cod (*Gadus morhua*), and silver hake (*Merluccius bilinearis*) were selected in the study. The first two species are flatfishes and the latter two are gadids. This combination allowed species both strongly associated with the bottom substrate and those that live close to the bottom but not usually in contact with the substrate on a prolonged basis respectively, to be included in the study. Their distribution and abundance was studied with respect to trawling activity, season, and a number of environmental variables (sediment organic content, sediment particle size, macroinvertebrate abundance, temperature, and salinity) that were perceived to be important attributes of juvenile fish habitat. Abundance of these species, measured as density, was used to assess their distribution and abundance with respect to trawling activity and the variation among the selected environmental variables. A known trawled site and an adjacent site where no shrimp trawling occurred were sampled during the winter and summer of 1998 and 1999 to do this analysis.

Many benthic fishes have preferences for particular types of habitats characterized by a suite of environmental variables. At the smallest scale, these preferences constitute their microhabitats within which physiological, feeding (Sogard, 1992; Gibson, 1994; Tupper and Boutilier, 1995) and refuge responses (Gotceitas and Brown, 1993; Walters and Juanes, 1993; Auster *et al.*, 1994, 1997; Gregory and Anderson, 1997) are enhanced.
For juvenile fish, such 'comfort zones' enhance growth and survival, and may constitute a suitable nursery habitat. Descriptions of habitat preferences of all of these study species in the Gulf of Maine are given in Bigelow and Schroeder (1953); this text is still currently the popular reference on the biology of fishes in the Gulf of Maine and adjacent areas. Similar descriptions for fishes in Canadian waters of the Atlantic, also applicable to the Gulf of Maine, are given in Scott and Scott (1988).

*H. platessoides* generally prefers fine sandy and mud bottoms (Scott, 1982), and is a "cold water species" (Scott and Scott, 1988). Its preferred depth ranges from 36-713 m, with the depth of occurrence largely influenced by bottom type and temperature (Scott, 1982). Studies on the effect of age, sex and abundance on the bathymetric distribution patterns of this species in the southern Gulf of St. Lawrence in September, based on a long-term data series, showed that their density was strongly related to depth (Swain and Morin, 1997). Highest densities were recorded at the 70-90 m depth range in this study. Both dispersion and depth of occurrence of *H. platessoides* were found to differ seasonally on the northern Grand Banks of Newfoundland (Morgan and Brodie, 1991). Fish were more aggregated in winter than in other seasons. Given 100-200 m as the depth range within which they commonly occur in this area, they preferred depths < 100 m in spring than in other seasons (Morgan and Brodie, 1991). A trend towards higher abundance at deeper stations was reported from trawl surveys along the coast of Maine (Sherman et al., 1993).

The other flatfish species selected in this study, *P. americanus*, prefers more coarser substrate, usually muddy sand, and even on pebbly and gravelly bottoms (Bigelow and Schroeder, 1953). Compared to *H. platessoides*, it is a shallow water
species occurring in inshore areas (Sherman et al., 1993), and also down to depths of about 30 m on the continental shelf (Scott and Scott, 1988). Young-of-the-year fish largely occur in inshore and estuarine areas (Able and Fahay, 1998). It is tolerant to a wide range of temperatures (Scott and Scott, 1988; Armstrong, 1997). At more northern latitudes adults undergo regular offshore migration in winter and move onshore in the summer, in response to pronounced seasonal temperature fluctuations. This pattern of migration with respect to season is reversed at more southern latitudes (Scott and Scott, 1988) due to warmer inshore temperatures at these latitudes during the summer. A winter and early spring spawner, this species is also found inshore during the spawning season (Bigelow and Schroeder, 1953).

Of the two gadids, the Atlantic cod, G. morhua, range in their distribution from inshore waters to the edge of the continental shelf (Scott and Scott, 1988). Fishing was reported to be good for this species at a depth of 3-5 fathoms (5-9 m) in the winter around Ipswich Bay (Bigelow and Schroeder, 1953), south of the area for this study. Adapted for bottom feeding, this species can occur off the bottom in pursuit of prey and can occur down to a depth of 457 m (Scott and Scott, 1988). The depth of occurrence is usually determined by temperature; cooler temperatures (-0.5-10 °C) are preferred. Preference for summer temperatures between 3-4 °C have been reported on the northeastern Scotian Shelf, while 7-8 °C on the southwest (Scott and Scott, 1988). G. morhua prefer pebbly grounds on gravel, sand and gritty clay with broken shell debris (Bigelow and Schroeder, 1953; Scott, 1982). Juveniles have been found to have strong preference for coarse pebble-gravel habitats on Georges Bank (Lough et al., 1989) and in inshore waters of
Newfoundland, Canada (Gregory and Anderson, 1997). In shallower water, occurrence among macroalgae is also a common habitat preference (Bigelow and Schroeder, 1953).

Silver hake, *M. bilinearis*, is a voracious predator feeding on invertebrates, younger conspecifics and other fish (Garrison and Link, 2000). The change in diet from invertebrate to fish is related to age, with younger fish feeding more on invertebrates. Large catches of younger fish have been recorded in areas with abundant shrimp (*Pandalus*) populations. They are benthopelagic, usually occurring off the bottom when feeding (Bigelow and Schroeder, 1953). *M. bilinearis* occur over a wide range of depths with fine sediments (Scott, 1982) while temperature is a major factor that determines its specific depth of occurrence. Among the gadids, it prefers temperatures warmer than that preferred by most members of this group (Scott and Scott, 1988). Occurrence on the Scotian Shelf at temperatures between 6-8 °C have been recorded (Scott and Scott, 1988). A recent study on the continental shelf fish communities of the New York Bight recorded juvenile *M. bilinearis* to be among the dominant species that settled on the outer shelf at a mean depth of 70 m (Steves *et al.*, 1998). At a 55 m deep site in the Middle Atlantic Bight, early juveniles (0-year) of this species occurred in higher densities on sand-silt bottoms with amphipod tube cover and with bottom temperatures in the range of 8.7-11.4 °C (Auster *et al.*, 1994, 1997).

In assessing environmental impact due to mobile gears such as shrimp trawling where bottom habitat features are easily perturbed, one needs to understand the natural patterns of habitat preferences before any impact can be analyzed. Natural variation in environmental factors that are strongly correlated with the distribution of different species, both spatially and temporally, can have a strong influence on their distribution.

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patterns. It is only with a good understanding of such natural habitat preferences of juvenile fish that one is able to interpret any effect on their ecology due to impact of mobile gear on habitat attributes. The variation in abundance of target species that are strongly correlated with natural variations in environmental factors can mask any effects of gear-related perturbations, hindering impacts to be determined conclusively. This is an especially useful consideration in impact studies with low statistical power. An in-depth coverage of this and related issues in designing impact assessment studies are discussed in Schmitt and Osenberg (1996).

This chapter focuses on assessing the variation in the abundance, both spatially and temporally, of the two predominant species out of the four selected in this study. In addition, any relationships with selected environmental variables (both biotic and abiotic) sampled in the study will be analyzed to determine the importance of these variables as habitat attributes.

**Materials and Methods**

**Fish Sample Processing**

Fish were sampled and processed as described in Chapter 1. After species identification, length and weight were measured. Samples of the four study species were then dissected to obtain gut, scales and otolith samples. For the purposes of aging these species, scale samples were taken at the region of the lateral line anterior to the caudal peduncle. The pair of sagittal otoliths was dissected from each specimen and dry-stored in capped glass vials.
The numbers of selected species caught on each sampling date at each site were compiled from the overall catch data for the fish community and tabulated as described in Chapter 1. For the purposes of the analysis in this chapter, abundance was calculated as density (numbers m\(^2\)) of fish with respect to the area swept per 10 minutes tow at 2 knots (1.03 m s\(^{-1}\)). The duration of the tow was the time elapsed between the locking of the winch after deploying the net to that at which hauling began. Using this duration, tow speed and the width of the ground rope (4.88 m), the magnitude of the area swept by each tow was calculated. This calculation was done by algebraically manipulating the basic formula for velocity:

\[ v = ms^{-1} \]

where \( v \) = velocity; \( m \) = distance in meters moved at the given velocity; and \( s \) = time in seconds taken to cover unit distance.

First the equation was re-arranged to get the distance moved in each 10 minute tow (\( m \)) on the left side of the equation. This changed the equation to \( m = v \times s \). Using the averages of tow speed (1.03 m s\(^{-1}\)) and the duration of each tow (10 minutes \(\times\) 60 seconds), the horizontal distance (\( m \)) covered by each tow was calculated. This distance (618 m) was then multiplied by the width of the ground rope (4.88 m) to get the area swept by each tow to be 3,015.84 m\(^2\). This figure was then used to calculate the density of each of the selected species for each sampling date and site using EXCEL before further statistical analyses were performed.

**Sediment Organic Content**

Besides determining sediment particle size composition analyzed in Chapter 1 from sediment samples collected with the Shipek grab sampler at the study sites in both
1998 and 1999, organic content was also determined. Duplicate samples of 2.0-3.0 g wet-weight were obtained from each grab for each sampling date onto pre-weighed aluminum cups. These were dried in an oven at 50.0 °C for four hours. After drying, they were cooled in a desiccator at room temperature and their dry weights were measured with an OHAUS® electronic balance, accurate to 0.001g. The samples were then ashed in a muffle furnace at 500 °C for four more hours. Both the temperature and duration of drying and ashing followed published recommendations (Dean, Jr., 1974; Mook and Hoskin, 1982; Kristensen and Andersen, 1987). After ashing, the samples were again cooled in the dessicator and re-weighed to obtain the ash-free weights. Organic content was then calculated from the dry weights by difference in weight loss due to ashing.

Data Analysis

Based on their overall abundance, only *M. bilinearis* and *H. platessoides* were caught in sufficient numbers throughout the study, justifying further statistical analysis. The data was first graphically analyzed to visualize spatial (between site) and temporal (between season) patterns of abundance of the two species. Subsequently, statistical analyses were done to test if these patterns were significant.

First, multivariate analysis of variance (MANOVA) was done to test the significance of the spatial and temporal variation in abundance revealed in the graphical analysis. Then the densities of the two species (DVs) were correlated, using canonical correlation (CANCORR), with selected environmental variables ordered into principal components (IVs) as described in Chapter 1. Most of the environmental variables used this time were same as those used in Chapter 1 but the IV, sea star, was replaced by organic content. This change was done based on the results of the analysis in Chapter 1.
and the review above of the environmental factors important to the distribution of the two species. Sediment organic content is a good correlate of the abundance of benthic macroinvertebrates, a variable important to many benthic fish species from a trophic perspective. This analysis was done to determine which environmental variables were important in influencing the spatial and temporal patterns of distribution and abundance of *H. platessoides* and *M. bilinearis*, thereby offering some insight into habitat attributes preferred by the juveniles of the two species at the two study sites. The analyses were done using the *MINITAB* and *SYSTAT* statistical packages.

**Results**

Juvenile *H. platessoides* and *M. bilinearis* were most abundant of the four study species sampled in the study (Table 2.1). Their overall spatial and temporal variation in density at the two study sites is summarized in Table 2.2. The density of both species was greater in Site 2 than in Site 1 (Figure 2.1). It is also apparent from Figure 2.1 that temporally, the two species occurred in greater densities in the summer than in the winter at the two sites.

MANOVA generated both univariate and multivariate *F*-tests to statistically test these patterns. The univariate *F*-tests showed that both the spatial (site) and temporal (season) differences in the density of *M. bilinearis* were not statistically significant (*F* [1] = 3.4321, *p* = 0.1231 for both site and season) (Table 2.3), but were both significant for *H. platessoides* (*F* [1] = 6.7518, *p* = 0.0483 and *F* [1] = 12.4749, *p* = 0.0167, respectively) (Table 2.3). However, the results of multivariate *F*-tests of both spatial and seasonal patterns in the density of both species were not significant (*F* [2, 4] = 2.7915, *p* = 0.1742 and...
Table 2.1. Numbers of the four study species caught by site and sampling date. Data of three replicate trawls per sampling date at each site were pooled.

<table>
<thead>
<tr>
<th>Family</th>
<th>Scientific name</th>
<th>Species code*</th>
<th>Site 1 Feb. (W)</th>
<th>Site 1 Jul. (S)</th>
<th>Site 2 Feb. (W)</th>
<th>Site 2 Jul. (S)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gadidae</td>
<td>G. morhua</td>
<td>GADMOR</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>M. bilinearis</td>
<td>MERBIL</td>
<td>0</td>
<td>2</td>
<td>3</td>
<td>6</td>
</tr>
<tr>
<td>Pleuronectidae</td>
<td>H. platessoides</td>
<td>HIPPLA</td>
<td>1</td>
<td>37</td>
<td>8</td>
<td>35</td>
</tr>
<tr>
<td></td>
<td>P. americanus</td>
<td>PSEAME</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>2</td>
</tr>
</tbody>
</table>

W = winter; S = summer.

* Refer to coding scheme used in Figure 1.3 (Chapter 1).
Table 2.2. Mean density (number m$^{-2}$) of American plaice, *H. platessoides*, and silver hake, *M. bilinearis*, by site, year and seasons within year calculated from their respective raw data given in Table 2.1. The data from three trawls, each sweeping an area of 3,015.84 m$^2$, were pooled for each sampling date for each site before calculating the mean densities. Values in parenthesis below each species' entry are mean densities adjusted to the area swept per tow*.

<table>
<thead>
<tr>
<th>Family</th>
<th>Scientific name</th>
<th>Species code</th>
<th>Site 1 1998</th>
<th>Site 1 1999</th>
<th>Site 2 1998</th>
<th>Site 2 1999</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gadidae</td>
<td><em>M. bilinearis</em></td>
<td>MERBIL</td>
<td>0.000022</td>
<td>0.00033</td>
<td>0.000088</td>
<td>0.00122</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>(0.0)</td>
<td>(0.6667)</td>
<td>(2.6667)</td>
<td>(3.0000)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.00066</td>
<td>0.00387</td>
<td>0.00144</td>
<td>0.00873</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>(2.0000)</td>
<td>(11.667)</td>
<td>(4.3333)</td>
<td>(26.333)</td>
</tr>
<tr>
<td>Pleuronectidae</td>
<td><em>H. platessoides</em></td>
<td>HIPPLA</td>
<td>0.00011</td>
<td>0.00088</td>
<td>0.00321</td>
<td>0.00144</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>(0.3333)</td>
<td>(2.6667)</td>
<td>(9.6667)</td>
<td>(4.3333)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.00409</td>
<td>0.00387</td>
<td>0.01404</td>
<td>0.00873</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>(12.333)</td>
<td>(11.667)</td>
<td>(42.333)</td>
<td>(26.333)</td>
</tr>
</tbody>
</table>

W = winter; S = summer.

* The area swept by each tow, calculated to be 3,015.84 m$^2$, is an area half the size of an average soccer field.
Figure 2.1. Average density of American plaice, *H. platessoides*, and silver hake, *M. bilinearis*, by site and season. The data in Table 2.1 was used, with the years (1998 and 1999) pooled. Error bars are standard error of means.
Table 2.3. MANOVA results of significance test of spatial (site) and temporal (season) variation in density (number/3,015.84 m²) of *M. bilinearis* (MERBIL) and *H. platessoides* (HIPPLA).

<table>
<thead>
<tr>
<th>Factor by species</th>
<th>Univariate</th>
<th></th>
<th>Multivariate*</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>F [df]</td>
<td>p</td>
<td>F [df]</td>
</tr>
<tr>
<td><strong>Site</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>MERBIL</td>
<td>3.4323 [1]</td>
<td>0.1231</td>
<td>-</td>
</tr>
<tr>
<td>HIPPLA</td>
<td>6.7518 [1]</td>
<td>0.0483</td>
<td>-</td>
</tr>
<tr>
<td>Both</td>
<td>-</td>
<td>-</td>
<td>2.7915 [2,4]</td>
</tr>
<tr>
<td><strong>Season</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>MERBIL</td>
<td>3.4323 [1]</td>
<td>0.1231</td>
<td>-</td>
</tr>
<tr>
<td>HIPPLA</td>
<td>12.4749 [1]</td>
<td>0.0167</td>
<td>-</td>
</tr>
<tr>
<td>Both</td>
<td>-</td>
<td>-</td>
<td>4.9998 [2,4]</td>
</tr>
</tbody>
</table>

* The multivariate test statistic used was Wilk's Lambda (λ).
$F_{[2,4]} = 4.9998, p = 0.0816$, respectively (Table 2.3). The proportion of unexplained variance of the two DVs with respect to site differences was greater ($\lambda = 0.4174$) than with seasonal differences ($\lambda = 0.2857$) (Table 2.3).

The CANCORR analysis correlating the density per tow of the two species (Table 2.2) as DVs with the four principal components (Table 2.4) as the IVs indicated that the model had no predictive value. The adjusted coefficients of multiple determination for both species were zero ($R^2 = 0.0$) (Table 2.5) with a canonical correlation coefficient ($r_c$) of 0.8545 (Table 2.6). Furthermore, both univariate and multivariate $F$-tests on the constant in the model (i.e. the overall slope ($\beta$)) of the model (Table 2.5) were not significant for both species (Table 2.6). The univariate statistics, testing the slope of the model with respect to the two DVs, $M. bilinearis$ and $H. platessoides$, with each IV term (PCs) in the model were $F_{[1,3]} = 6.5694 (p = 0.0830)$ and $F_{[1,3]} = 3.2012 (p = 0.1715)$, respectively. The multivariate statistics testing the significance of the overall slope of the canonical correlation with both DVs tested simultaneously in the model was $F_{[2,2]} = 2.7049 (p = 0.2699)$ (Table 2.6).

**Discussion**

The distribution and abundance patterns of fish, as in many other organisms, are dependent on biological, ecological and environmental processes operating over different spatial and temporal scales. Interspecific variations in these patterns are dependent on differences in response to these processes among species. As discussed in Chapter 1, this consequently gives a fish community its specific structure over a given space and time.
Table 2.4. MINITAB results of principal component analysis on eight environmental variables to select principal components that encompassed correlated sets of composite variables (PCs) to use as IVs in the canonical correlation analysis. Based on their eigenvalues and a scree plot, the first four PCs that explained much of the variation were selected. Part I of the table shows the results of eigenanalysis and Part II gives the loading on the environmental variables in each PC.

I. Eigenanalysis of the Correlation Matrix

<table>
<thead>
<tr>
<th></th>
<th>PC1</th>
<th>PC2</th>
<th>PC3</th>
<th>PC4</th>
</tr>
</thead>
<tbody>
<tr>
<td>Eigenvalue</td>
<td>4.0743</td>
<td>1.9753</td>
<td>0.8954</td>
<td>0.8226</td>
</tr>
<tr>
<td>Proportion</td>
<td>0.509</td>
<td>0.247</td>
<td>0.112</td>
<td>0.103</td>
</tr>
<tr>
<td>Cumulative</td>
<td>0.509</td>
<td>0.756</td>
<td>0.868</td>
<td>0.971</td>
</tr>
</tbody>
</table>

II. Variable

<table>
<thead>
<tr>
<th>Variable</th>
<th>PC1</th>
<th>PC2</th>
<th>PC3</th>
<th>PC4</th>
</tr>
</thead>
<tbody>
<tr>
<td>Depth</td>
<td>0.486</td>
<td>0.021</td>
<td>-0.040</td>
<td>0.011</td>
</tr>
<tr>
<td>Temperature</td>
<td>-0.029</td>
<td>0.610</td>
<td>-0.310</td>
<td>0.380</td>
</tr>
<tr>
<td>Salinity</td>
<td>0.002</td>
<td>0.624</td>
<td>-0.231</td>
<td>-0.361</td>
</tr>
<tr>
<td>Gravel</td>
<td>-0.442</td>
<td>-0.024</td>
<td>-0.456</td>
<td>-0.111</td>
</tr>
<tr>
<td>Sand</td>
<td>-0.434</td>
<td>-0.102</td>
<td>0.235</td>
<td>0.420</td>
</tr>
<tr>
<td>Mud</td>
<td>0.472</td>
<td>0.068</td>
<td>0.220</td>
<td>-0.194</td>
</tr>
<tr>
<td>Organic</td>
<td>-0.164</td>
<td>0.471</td>
<td>0.700</td>
<td>0.113</td>
</tr>
<tr>
<td>Crangon</td>
<td>0.359</td>
<td>-0.010</td>
<td>-0.217</td>
<td>0.697</td>
</tr>
</tbody>
</table>
Table 2.5. *SYSTAT* results of model estimation of canonical correlation between *M. bilinearis* (MERBIL) and *H. platessoides* (HIPPLA) as DVs and the four principal components (PC1-PC4, Table 2.4) as the IVs. The model used was MERBIL HIPPLA = Constant + PC1 + PC2 + PC3 + PC4.

<table>
<thead>
<tr>
<th>Aspect of model</th>
<th>Strength and estimates of model parameters</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>β (Standardized)</td>
</tr>
<tr>
<td><strong>Regression of DVs with (IVs)</strong></td>
<td></td>
</tr>
<tr>
<td>MERBIL (PC1)</td>
<td>0.0568</td>
</tr>
<tr>
<td>MERBIL (PC2)</td>
<td>-0.5709</td>
</tr>
<tr>
<td>MERBIL (PC3)</td>
<td>-0.0957</td>
</tr>
<tr>
<td>MERBIL (PC4)</td>
<td>0.0307</td>
</tr>
<tr>
<td>HIPPLA (PC1)</td>
<td>0.4067</td>
</tr>
<tr>
<td>HIPPLA (PC2)</td>
<td>-0.2974</td>
</tr>
<tr>
<td>HIPPLA (PC3)</td>
<td>0.0183</td>
</tr>
<tr>
<td>HIPPLA (PC4)</td>
<td>0.0499</td>
</tr>
<tr>
<td><strong>Multiple correlation</strong></td>
<td></td>
</tr>
<tr>
<td>MERBIL</td>
<td>-</td>
</tr>
<tr>
<td>HIPPLA</td>
<td>-</td>
</tr>
</tbody>
</table>

β: Multiple regression coefficient.

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Table 2.6. *SYSTAT* results of model effect significance tests for the canonical correlation model described in Table 2.5.

<table>
<thead>
<tr>
<th>Effect tested</th>
<th>$r_c$</th>
<th>$\lambda$</th>
<th>$F(df)^*$</th>
<th>$p$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Constant ($\beta$)</td>
<td>0.8545</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>MERBIL</td>
<td>-</td>
<td>-</td>
<td>6.5694 [1,3]</td>
<td>0.0830</td>
</tr>
<tr>
<td>HIPPLA</td>
<td>-</td>
<td>-</td>
<td>3.2012 [1,3]</td>
<td>0.1715</td>
</tr>
<tr>
<td>Variance overlap</td>
<td>-</td>
<td>0.2699</td>
<td>2.7049 [2,2]</td>
<td>0.2699</td>
</tr>
</tbody>
</table>

$r_c$: Canonical correlation coefficient.

$(\lambda)$: Eigenvalue = the proportion of variance in the DVs that overlapped with IVs (the constant in this case) other than the one whose effect was being tested.

* The third $F$-test in the column is a multivariate test testing the effect of both DVs simultaneously.
Natural processes such as: 1) spatial and temporal reproduction and recruitment variability (Colton et al., 1979; Longhurst, 1984; Sherman et al., 1984; Roughgarden, 1989, 1998); 2) interspecific interactions (competition, predation and mutualism) among species (Dayton, 1984; Roughgarden, 1989; Morin, 1999); and 3) impact from physical processes such strong storm-related turbulence and currents (Hall, 1994; Watling, 1996), often have a greater impact on marine fish populations. Many species are adapted and resilient to many of these forces. Any anthropogenic impact within such a 'noisy' (high variability) system, where the response of target species to the impact is confounded within a background influence from such natural forces, may not be easily discernible. It would require powerful statistical methods and sampling designs to make impact detection possible (Underwood, 1992, 1993, 1996; Osenberg et al., 1996; Glasby, 1997).

Since the overall objective of this study was to study the impact of shrimp trawling in the southwestern Gulf of Maine on the juveniles of selected species, it was first necessary to analyze any natural differences in spatial and temporal patterns in distribution and abundance among these species. As reviewed in the Introduction to this chapter the species selected in this study show specific preferences to a number of environmental variables that are important to many of the processes listed above. It is only with knowledge of the degree of influence from natural processes in inducing these patterns that one can be able to assess the impact of anthropogenic perturbations such as shrimp trawling meaningfully.

To do this, a combination of graphical analysis and multivariate statistical techniques were used to: 1) detect group differences in patterns associated with factors of interest (space and time); and 2) correlate existing patterns with a range of biotic and
abiotic environmental variables measured in the field. Results of graphical analysis showed apparent spatial and temporal differences in the density of *H. platessoides* and *M. bilinearis* (Figure 2.1), the two most abundant of the four species selected for study (Table 2.1). Both species were more abundant at Site 2, the site at which shrimp trawling is known to occur, than at Site 1 (the 'no trawl' site). Temporally, they both occurred in greater densities at both sites in the summer than in the winter, a non-surprising pattern at boreal/temperate latitudes such as the Gulf of Maine where many biological processes have a strong seasonality. The purpose of the graphical analysis was to reveal patterns of interest between the two species. Hence, a common y-axis was not used since *M. bilinearis* had densities ten times lower than that of *H. platessoides* (Figure 2.1).

To enable some explanation of the role of any of the measured variables in influencing these patterns it was necessary to test, using MANOVA, if these spatial and temporal patterns in abundance were significant. Based on the univariate F-tests from this analysis, both spatial and temporal patterns of *H. platessoides* revealed by the graphical analysis were statistically significant (*F* [1] = 6.7518, *p* = 0.0483 and *F* [1] = 12.4749, *p* = 0.0167, respectively (Table 2.3). However, both patterns for *M. bilinearis* were not significant from the F-tests (*F* [1] = 3.4323, *p* = 0.1231 for both site and season) (Table 2.3).

According to the description of the study sites in Chapter 1 the mean depth in Site 1 (64.8 m) is shallower than that in Site 2 (70.8 m). They also differ in sediment composition in terms of the relative proportions of sand and mud-sized particles. Site 1 has an average sand content of 55 % and 32 % mud, while Site 2 has 42 % and 58 %, respectively. Besides differences in these habitat variables there was a 1 °C difference in
bottom temperature between summer and winter. As reviewed above, all the study species are associated with most of these variables.

From the results of the univariate $F$-test of the MANOVA there might be some degree of association of \textit{H. platessoides} with these variables that might explain its significant temporal and spatial differences in abundance. However, in looking at the multivariate $F$-tests, based on a simultaneous test of both species, the spatial and temporal differences in the two species were not significant ($F_{[2,4]} = 2.7915$, $p = 0.1742$ and $F_{[2,4]} = 4.9998$, $p = 0.0816$, respectively (Table 2.3). The multivariate $F$-test in a MANOVA is usually more conservative than the univariate test, and it is recommended that any interpretations be based on it rather than the univariate $F$-test (Tabachnick and Fidell, 1996) when there is disparity in significance between the two tests as in this case.

Recall from the study area descriptions in Chapter 1 and Figure 1.1 therein, that the study encompassed an area about 16 km$^2$ (4 km x 4 km), with two study sites each 1 km x 1 km in size, on the mid-continental shelf of the southwestern Gulf of Maine. The two study sites were separated by a horizontal distance of 1.19 km with Site 1 being shallower than Site 2 by 5-6 m on average. Such a spatial scale might not be large enough for many of the environmental variables that each species is associated with to differ very much. Hence, their influence on the two species within the spatial scale of the study would be insignificant.

\textit{M. bilinearis}, a voracious predator (Garrison and Link, 2000 (b)), is distributed over a wide range of depths with temperature being a major determinant in its occurrence (Scott and Scott, 1988). Temperature, bottom type, and depth are important variables determining the distribution of \textit{H. platessoides} (Scott, 1982; Scott and Scott, 1988;
Morgan and Brodie, 1991; Sherman et al., 1993). Bottom temperatures between sites analyzed in Chapter 1 (Figure 1.2; Table 1.2) within the depth range (65-71 m on average) of the study sites indicate very little difference spatially but changed by a difference of about 1 °C seasonally. This variable is correlated to an extent with depth. A difference in depth of 5-6 m on average between the two sites seems an insufficient magnitude of change to cause major changes in bottom temperature. As analyzed in Chapter 1, both M. bilinearis and H. platessoides were resident species at both study sites. Hence, the spatial-scale of the study might be small, relative to that in which many variables important to the distribution of the two species are expected to vary significantly. This could be a possible explanation for the two species in not having significant spatial (site) and temporal (season) differences in their density (Table 2.3).

The results of the correlation analysis of the density of the two species (DVs) and the four principal components representing composite variables (IVs) (Table 2.4) indicated that the model tested was not significant ($R^2 = 0.0$ (Table 2.5); $r = 0.8545$, with $F_{[1,3]} = 6.5694$, $p = 0.0830$ and $F_{[1,3]} = 3.2012$, $p = 0.1715$ for M. bilinearis and H. platessoides, respectively (Table 2.6). PC1 represented a depth and bottom type composite while PC2 represented a temperature and salinity composite. These first two principal components together explained 75.6% of the total variation in the principal component analysis (Table 2.4). The result of this correlation seems to also indicate a lack of significant difference in the key variables known to affect the distribution of the two species within the spatial scale of the study. It appears to offer support for the 'spatial-scale' explanation given above rather than to imply that no relationship exists between the two species and the composite variables (PC1-PC4) in the model.
Impact studies can be of two kinds. A "field assessment" study is one where a site with a known perturbation was studied relative to one where the specific perturbation was absent. On the other hand, a "field experimental" study would be one in which impact and control sites are experimentally manipulated by the researcher(s) and the effect of the experimentally-induced impact on the DV(s) of interest are tested (Osenberg and Schmitt, 1996).

This study was a field assessment study. Site 2 was selected within a broad strip of mud-bottomed seabed in which trawling was known to occur, but located adjacent to Site 1 (the 'no trawl' site). A horizontal distance of 1.19 km separated the two sites, a distance selected based on the range within in which the spatial variation in physical variables such substrate and depth can be expected to be minimum (Langton, 1998). Interspersed between the two sites was also an area of shallower water (37.8 m). This was expected to serve as a barrier between the two sites and mutually exclude any 'edge effect' of the two sites on each other's fauna. Furthermore, this spatial choice of the location of the two study sites was also made to ensure constancy among many of the environmental variables within the study area, in order to allow any difference in abundance between sites to be attributed to trawling activity. The above results seem to confirm that these objectives were achieved.

The result of this analysis hints to a very important consideration necessary in designing a field experimental impact study; natural variations in variables important to DV(s) (say abundance of some fish species of interest) among experimental units must be kept constant as much as possible. One strategy often used to achieve this is the selection of an appropriate spatial scale in which any influences of key environmental variables...
among the experimental units are comparable. This is a necessary prerequisite in order to ensure any difference in the DV(s) that result (s) from the experimental manipulation to be detected.

Hence, a similar spatial scale to that of this study would be appropriate in studying the impact on similar fish species in such experimentally manipulated studies. Natural variation in key variables can be 'controlled' for, allowing experimentally induced impacts to be assessed conclusively. Such studies have been done. Gordon, Jr. et al., (1998) assessed the abundance of benthic macrofauna and Thrush et al. (1995) also investigated the impact of scallop dredging on marine benthic communities. Sullivan et al. (2000) assessed the potential impact of scallop dredging on juvenile yellowtail flounder, *Limanda ferruginea*. Such studies are often logistically demanding and very expensive to carry out. However, the problem of controlling the influence of natural variables effectively in order to reduce 'environmental noise' and thereby achieving desirable levels of statistical power possible from such 'field experimental' studies remains a challenge. This is especially so when the magnitude of the variance of the DV(s) of interest due to environmental factors is often greater than that due to the manipulated impact.
CHAPTER 3

POPULATION STRUCTURE, HABITAT AND TROPHIC ECOLOGY OF JUVENILE AMERICAN PLAICE, HIPP OGLOSSOIDES PLATESOIDES, IN THE SOUTHWESTERN GULF OF MAINE

Abstract

Four age classes of H. platessoides were identified by length frequency analysis and scale-age determination. They were Age 0+, 1+, 2+ and 3+. The average length (TL) for the respective age classes were 5.9 cm and 10.4 cm (there were two cohorts of Age 0+ fish), 16.3, 22.3 and 28.0 cm. The abundance of the fish in each age class declined as the fish got older. Age 3+ fish were only present in the study area during the summer, possibly moving into deeper water during the winter, or possibly missed by the gear due to changes in distribution patterns associated with seasonal changes in behavior.

Based on the diet analysis of 227 gut samples with prey items present, the diet of juvenile H. platessoides was predominated by ampharetid polychaetes (Ampharetidae), other polychaetes, amphipods, bivalves and cyclopoid copepods. The Ampharetidae and amphipods occurred in all age classes, but the sizes and numbers taken varied among size classes. The cyclopoid copepods were predominant only in Age 0+ fish while the bivalves were predominant only in the Age (1 & 2)+, occurring largely in fish of Age 2+ and older. Of the bivalves, the family Nuculanidae was especially taken in large numbers.
whenever it occurred in the diet of those older fish.

Analysis of the abundance of prey taxa from the grab samples showed some evidence of fish predation pressure on the summer abundance of predominant prey taxa. However, a MANOVA test for spatial (site) and temporal (season) variation in the mean number of Ampharetidae (Polycapheta) and Amphipoda, the two taxa that were predominant in the diet of both Age 0+ and Age (1 & 2)+ H. platessoides, was not significant.

Introduction

As reviewed in the Introduction chapter, many ecological processes are organized hierarchically interconnecting more than one level of ecological organization, ranging from individual and population level to the community and ecosystem levels. In an effort to assess any ecological impact of shrimp trawling in the southwestern Gulf of Maine on selected juvenile fish species, a hierarchical approach was also taken in analyzing the data and presenting the results in this dissertation. This was done to ensure that any ecological impact not resolved at one level could be resolved at another level, and also to enable some insight to be gained of the nature of transmission of any ecological impact.

In the first chapter of this dissertation, the fish community in the study area was analyzed to determine the spatial and temporal variation in community structure. An overall profile of the fish community in the study area was described in Chapter 2. Both chapters analyzed environmental factors that were potentially responsible for shaping the patterns in community structure, as well as in affecting population aspects such as distribution and abundance in the selected species. This approach facilitates the
answering of specific questions, as outlined in the Introduction chapter, using the most appropriate data as indicated from analysis performed in the preceding chapter(s).

This chapter focuses on more population-level processes with respect to the impact of shrimp trawling. Using data for *H. platessoides*, the most abundant species captured, it was the goal of this chapter to analyze for any impact of trawling at the population level. To do this, the population structure, habitat and trophic ecology of this species was analyzed to detect if any processes relevant to its autecology and population biology were affected.

The population structure, shelf-wide habitat selection, and trophic ecology of *H. platessoides* will be analyzed to achieve this goal. The analysis of length frequency data is a very feasible technique for assessing the age composition of the juveniles recruited to the continental shelf in the study area. Tracking these patterns in age classes spatially and seasonally can offer some insight into the nursery role of the shelf habitat for this species.

Analysis of juvenile benthic fish recruiting to the continental shelf of the New York Bight has been used to provide information on the spatial and temporal variations in the use of the continental shelf by juveniles of different species (Steves *et al*., 1998). Most fish species in boreal and temperate latitudes have seasonal reproductive patterns. This has been shown to occur in many fish of the waters of the northeastern United States, including the Gulf of Maine (Bigelow and Schroeder, 1953; Colton Jr. *et al*., 1979; Sherman *et al*., 1984, 1987). This seasonality makes tracking age classes and recruitment events using length frequency analysis very convenient.

As reviewed in the Introduction chapter, scrapping, excavation and ploughing by trawl doors is the major impact caused by mobile gears such as a shrimp trawl (Messieh
et al., 1991; Jones, 1992; Brylinsky et al., 1994; Gordon et al., 1998; Pilskaln et al., 1998; Rogers et al., 1998). Vertical depths of impact into the substrate range from 4-5 cm or more (Brylinsky et al., 1994; Gordon et al., 1998; Pilskaln et al., 1998). Where trawling is frequent and extensive, this type of impact can have serious consequences on the benthic macroinvertebrate communities.

Feeding studies done in the shelf waters of the eastern United States indicated that many species of macroinvertebrates comprise important prey for flatfishes and younger stages of many benthic fish (Hacunda, 1981; Packer et al., 1994; Garrison and Link, 2000 (a)). The trophic importance of invertebrates to fish is well known. Larvae of many fish species feed on invertebrates (zooplankton). Garrison and Link (2000 (a)) reported that the young of many piscivores prey on benthic macroinvertebrates as well, and that an ontogenetic shift in the size and species selected does occur. Where predation pressure on certain species of benthic macroinvertebrates is high, the abundance and population structure of the prey can also be affected, as was shown for the brittle star, *Ophiura sarsi*, in the Gulf of Maine due to predation by *H. platessoides* (Packer et al., 1994).

*H. platessoides* was classified within the predator category of a "benthivore" throughout much of its juvenile and adult life in the Northeast United States continental shelf ecosystem (Garrison and Link, 2000 (a)). If shrimp trawling is impacting the benthic communities in the study area, it would affect the trophic ecology of such species as a result of disturbance to the bottom and prey communities. Since it is also the abundant benthic fish species in the study area, it is a good candidate to study whether shrimp trawling in the southwestern Gulf of Maine has any impacts on its autecology by focusing on its trophic ecology. The results from this analysis will be integrated with its
spatial and temporal patterns in population structure and habitat selection, to describe aspects of any overall ecological impact(s) resulting from shrimp trawling. If that cannot be conclusively resolved, some description of potential ecological impacts will be given.

Materials and Methods

Population Structure

Population structure of *H. platessoides* in the study area was analyzed from the length frequency data obtained as described in Chapter 1. Using MINITAB, the length frequency data were analyzed graphically to determine spatial and seasonal patterns in the composition of age classes (modes in frequency distribution) within the study area.

The ages of fish represented by each mode were determined from scale samples obtained at the time of fish processing (Chapter 1). Aging involved wet-mounting scale samples onto glass slides with drops of water and cover slips. Then using Penttila and Dery (1988), the scale's morphological features important to age determination were identified. These features were: 1) the anterior and posterior margins of the scale; 2) regions coinciding with slow growth (i.e. rings were closer together); 3) those coinciding with fast growth (i.e. rings were farther apart); and 4) the nucleus and axis of growth on the scale. A zone of slow growth represented growth in the winter, while that of fast growth represented summer growth (Penttila and Dery, 1988). According to the literature (Bigelow and Schroeder, 1953; Colton *et al.*, 1979; Sherman *et al.*, 1984, 1987; Scott and Scott, 1988), this species is a spring spawner in this region including Gulf of Maine. This information was used in conjunction with scale age to determine the relative ages for each cohort. This was done by counting the number of consecutive pairs of regions of
slow and fast growth, starting from the nucleus (or primordium) to the margin of the scale, along the axis of size increment in the scale. Each consecutive pair of regions of slow and fast growth was interpreted to represent winter and summer growth respectively over a one-year period, representing an annulus.

**Diet Analysis**

Gut samples were dissected at the time of fish processing (Chapter 1). In trawls where a large number of juvenile fish of a particular age class was present in the catch, which usually was the case with Age 0+ and Age 1 fish from Site 2 in the summer, a subsample was selected using a computer (M/ML4i?)-generated set of random numbers. The gut of *H. platessoides* is long, coiled and without a distinct stomach so only the anterior third of the gut was sampled. Where the gut was distended, which often occurred when fish fed on few large-sized individuals of a single taxa (e.g. shrimps by Age 2+ fish) making such sampling without bias on the separation point difficult, the entire gut was retained. The samples were preserved in 10 % buffered formalin in glass vials until further processing.

**Sample Size (n) Determination.** Before dietary analysis was done the sample size (n) needed for dietary analysis was determined using the largest catch of *H. platessoides*, collected in July 1998 at Site 2. Using graphs of cumulative numbers of new prey taxa recorded with increasing number of guts processed for fish of Age 0+, and Ages 1+ and 2+ combined, suitable sample sizes (n) needed for each age class were determined. That is, by plotting a curve of cumulative increase in prey taxa (Family level in this study) represented in each gut sample for each age class as a function of the number of guts processed, an asymptote in the curve is reached, representing the minimum sample size.
required. This is a technique stated in a recent review (Cortés, 1997) to be a desirable approach to precisely describing the diet of a predator. Based largely on sufficient samples of guts, only guts of the Age 0+ to Age 2+ cohorts were selected for diet analysis. Age 3+ fish were excluded from the dietary analysis because there were fewer fish in the samples, and they were caught only in the summer. Furthermore, most of the fish in this age class were sub-adults.

Diet Analysis. To analyze the gut samples, excess preservative in each sample vial was carefully siphoned off with a pipette, and the samples were washed into a petri dish. The gut was then teased open carefully with dissecting equipment. Contents were then separated into major prey categories under a dissecting microscope, counted and weighed. Weights were measured using a digital balance (OHAUS® TS series), accurate to 0.001g when possible. The data was recorded onto data sheets and stored as EXCEL spreadsheets for further graphical and statistical analyses.

Data Analysis. The data was analyzed using two principal methods: 1) a percentage (%) occurrence method; and 2) a numerical method. Briefly, the numerical method involves counts of each prey taxa present in each gut in a sample. Where prey items were not intact, eyes, heads, and other parts were used to account for whole individuals in the count. The gravimetric method used weights of each prey category in a sample, while the percentage (%) occurrence method involved calculating the proportion of all the guts in the sample that had a particular prey category present. These methods were described in Hyslop (1980). A mean number per gut within the age classes was also calculated for each prey taxa. This calculation was needed to assess the importance of any prey taxa whenever there is disparity between the results of the two methods since...
numerical data tend to be biased by the size of the taxa taken in the diet. Additionally, it provided a way to get an idea of prey availability and foraging intensity (Hynes, 1950), enabling conclusions to be made of habitat use when fish density data are taken into account.

The weight data was not used in the final analysis due to a number of problems. Most items were usually in an advanced stage of digestion, and also common prey items such as polychaetes and bivalves were ingested together with their tubes and shells, respectively, rendering any interpretations of the original weight of the item at the time of ingestion inaccurate. Furthermore, it was also impossible to measure the weight (0.000 g) of taxa of small size, such as amphipods and cyclopoid copepods, present in the diet. Their small size made weight measurements impossible, especially when few individuals were present in the gut contents.

**Prev Abundance**

In conjunction with diet analysis, an analysis of the abundance of macroinvertebrates (numbers per square meter) in grab samples was also undertaken to analyze spatial and seasonal dietary variation with respect to prey availability in the environment.

**Field Sampling.** Samples were obtained with a *Shipek* grab sampler consisting of a sampling bucket 20.5 cm x 20.0 cm in size that samples an area of 410 cm² (0.041 m²). This size is standard for this type of sampler (Holme and McIntyre, 1971). The bucket is half-cylindrical in shape with the above dimensions being its length and diameter, respectively. When full, it was calculated to sample a volume of 3,220.13 cm³.

The grab sampler was launched on winch over the stern of the research vessel
until a short distance off the bottom (1-2 m) then released to free-fall. This method allowed the closing mechanism of the grab to be triggered upon landing on the bottom and also enabled the grab to dig-in, especially when sampling in locations with more compact and coarse substrates. When hauled, the bucket was carefully removed the right side up and the sample scooped into a labeled plastic bag. All samples collected in this manner were transported to the laboratory and frozen immediately until ready for processing.

**Laboratory Processing.** The grab samples were processed by initially thawing the samples. Two replicates, each consisting of an amount sufficient to give a dry weight sample of 25 g, was sampled with a spoon from each sample for sediment particle size and organic content analysis (see Chapters 1 and 2). These were then stored in labeled zip-lock plastic bags. Total volume of material in each sample was measured with a graduated plastic beaker (1L capacity). Manageable amounts, usually in two batches per grab, were transferred into a plastic bucket and made into diluted slurry with water and sifted through a series of sieves of decreasing size: 4 mm, 2 mm, 0.5 mm and 0.25 mm over a bucket. The residue was examined by eye. Larger visible specimens we picked with fine forceps and transferred into labeled jars with 10 % buffered formalin. The remainder of the material was back-washed into a white background enamel tray. Excess water was carefully decanted off before pouring the rest of the material into the specimen jars for storage.

**Identification and Quantification.** When ready for macroinvertebrate identification and quantification, the preservative in each jar was carefully siphoned out with a pipette, and the contents were then washed into a petri dish. Using a dissecting microscope, an
illustrative key prepared by Grizzle (upubl.) and other published invertebrate taxonomy
texts for this region (Gosner, 1971, 1978; Weiss, 1995) all specimens were identified,
mainly to Family level. They were counted and weighed to 0.001g using the OHAUS®
digital balance. The data were then entered as EXCEL spreadsheets and processed for
subsequent data analysis.

**Sample Size (n) Determination.** Three grabs per site per sampling date were
sampled, with a fourth back-up sample also taken when grabs with volumes less than 500
cm³ were obtained (Table A2, Appendix). Therefore, it was necessary to determine the
degree of representativeness of this choice of sample size with respect to the natural
spatial variability in the abundance of the macrobenthic community within the study area.
A one-time additional fifteen grab samples were collected at each site in the summer
(July) of 2000 for the purpose of this analysis. However, since the grab data analyzed for
the two sites showed little spatial variation in the study area, only samples for Site 2 were
arbitrarily chosen for analysis. The same graphical technique used to determine sample
size in gut analysis described above was used; this time the x-axis was number of grabs
processed instead of guts. Due to the constraints stated above resulting in fewer grab
samples than desirable collected per site on each sampling date, this analysis was done to
estimate the magnitude of any under-sampling, and to determine if this problem affected
the interpretations of the trophic ecology of *H. platessoides*.

**Data Analysis.** The abundance of prey was analyzed by tabulating summaries of
prey taxa and comparing their trend in predominance in the diet of Age 0+ and Age (1 &
2)+ fish, both spatially and temporally. Graphical analyses were also done after major
patterns in the data were determined, in order to visualize the trends clearly.
Subsequently, the abundance of predominant macroinvertebrate taxa present in grab samples collected at the two study sites was also analyzed using the same graphical technique. A time-series plot of the two data sets was then done, to assess if any trends existed between taxa present in the grab samples and those that occurred in the diet of juvenile *H. platessoides*. Due to the suitability of sample sizes of guts obtained for Age 0+ fish at Site 2, this data was used to illustrate the temporal trends.

A MANOVA was done on the mean abundance of the Ampharetidae (Polychaeta) and Amphipoda in the grab samples, the two predominant taxa that occurred in both Age 0+ and Age (1 & 2)+ *H. platessoides*, to test for any site and seasonal differences.

**Sediment Organic Content**

Besides determining the distribution of organic content horizontally at the two sites from grab samples as described in Chapter 2, vertical distribution of this variable was also measured from core samples taken with a gravity corer in 1998. The corer consisted of a metal pipe designed to screw on a "rudder" portion at the top end, and a streamlined "mouth" piece at the lower end of the unit. This facilitated both steering the unit vertically when launched, and also the smooth cutting by the "mouth" piece into the sediment. Depth of the core was adjusted by adding and subtracting 20 kg-donut-shaped lead weights to the upper end to the unit. Because my study area was largely mud, and a total depth of 30 cm was to be sampled, two weights were sufficient. The length of the pipe was designed to accommodate a 2.44 m of acrylic tube, 6.7 cm in internal diameter, fitted to capture vertical sediment cores. Tubes were held firmly in place with heavy-duty tape, and the "mouth" end of each set of tubes was fitted with a metal catcher that prevented sediment from slipping out of the tubes during hauling or shipment. After
sampling, the acrylic tube was removed vertically and all six cores (three per site) were kept vertical throughout transportation to the laboratory.

In the laboratory the sediment cores were extruded onto a long tray made from a PVC pipe cut in half along the long axis. This provided a suitable receptacle to keep the core intact while sections of desired lengths were measured and cut off. Six sections were cut at the following intervals from the top to the bottom of each core: 0-1 cm, 1-4 cm, 4-7 cm, 7-10 cm, 10-20 cm and 20-30 cm. These depths corresponded with depths normally occupied by different groups of benthic infauna (Levinton, 1995; Lalli and Parsons, 1997). Each section was stored in a labeled zip-lock plastic bag and then frozen until ready for further processing. The organic content in each section of each core was analyzed using the method described in Chapter 2, only for the cores collected in 1998 at the two study sites. The data were graphically analyzed to show the vertical profiles of the organic content with respect to sediment depth.

**Results**

**Population Structure and Habitat**

The population structure analysis resulted in producing three length-frequency histograms. Figures 3.1 (a &b) used winter and summer data, respectively for fish caught in Site 2, while Figure 3.1 (c) used pooled data for both seasons for both 1998 and 1999 at this site. This analysis was only based on data for Site 2 since higher numbers of *H. platessoides* were caught at this site. Furthermore, given the spatial scale of the study, fish from Site 1 would be from of the same population and therefore expected to have the same size composition characteristics.
Figure 3.1. Size frequency histograms of American plaice, *H. platessoides*. The panels are from top to bottom: (a) pooled (1998 and 1999) winter samples; (b) summer (1998) samples; (c) pooled (1998 and 1999) samples for both seasons. All samples were from Site 2 since sufficient numbers were caught at this site.
It was apparent that three cohorts were present in the study area in winter. A few larger fish may indicate that an additional older cohort was also present. The first cohort (Age 0+) encompassed fish 6 cm TL modal length, and the second (Age 1+) was fish 15 cm TL in modal length (Figure 3.1 (a)). The older cohort also present was probably Age 2+ with the modal length between 19-22 cm TL. These modal lengths also approximated the average lengths of the two age classes confirmed with scale aging (Table 3.1).

Results of the same analysis done on summer samples revealed that besides the three age classes identified above, a fourth cohort was also present in the study area during the summer (Figure 3.1 (b)). Based on scale age (Table 3.1) and their modal length of about 28 cm TL, those were Age 3+ fish. Since the Age 0+ cohort would by then have lived past its first spring post-spawning into the second summer, it was now larger with a modal length of 11 cm TL, and are now Age 1+ fish. The Age 1+ cohort of the winter had by then attained about a 2-3 cm increase in TL, now with a modal length about 18 cm TL. This cohort would be approaching Age 2+. The third cohort with a modal length of 24 cm TL and an average length of 22.3 cm TL (Table 3.1) would constitute Age 2+ or older fish, depending on whether their lengths fall at the lower or upper tail of the length frequency distribution with respect to the mode. Age 3+ was the newly included cohort in the samples with a modal length of about 28 cm TL was the cohort (Figure 3.1 (b)), this length also concurred with the average length (Table 3.1).

The analysis was repeated for a larger sample comprising data pooled for both seasons and years from Site 2 to confirm this pattern in population structure. Again the four cohorts (Age 0+, 1+, 2+ and 3+, respectively) were revealed. Their modal lengths were 10, 16, 24 and about 28 cm TL, respectively (Figure 3.1 (c)); the modal lengths
Table 3.1. Age class determination from scale samples of juvenile plaice, *H. platessoides*. All numerical entries are averages (± S.D.) based on the number of samples (n) analyzed to determine the age class (last column).

<table>
<thead>
<tr>
<th>n</th>
<th>Sampling Date</th>
<th>TL (cm)</th>
<th>Number of complete growth bands</th>
<th>Total annuli</th>
<th>Age class</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Slow (winter)</td>
<td>Fast (summer)</td>
<td></td>
</tr>
<tr>
<td>10</td>
<td>2/23/99 (winter)</td>
<td>5.9 (0.7651)</td>
<td>0.0 (0.0000)</td>
<td>0.0 (0.0000)</td>
<td>0.0 (0.0000)</td>
</tr>
<tr>
<td>8</td>
<td>7/24/98 (summer)</td>
<td>10.4 (0.6034)</td>
<td>1.0 (0.0000)</td>
<td>1.0 (0.0000)</td>
<td>0.0 (0.0000)</td>
</tr>
<tr>
<td>7</td>
<td>7/24/98 (summer)</td>
<td>16.3 (4.0519)</td>
<td>1.4 (0.5345)</td>
<td>1.0 (0.0000)</td>
<td>1.0 (0.0000)</td>
</tr>
<tr>
<td>8</td>
<td>7/24/98 (summer)</td>
<td>22.3 (2.8324)</td>
<td>2.4 (0.5164)</td>
<td>2.0 (0.0000)</td>
<td>2.0 (0.0000)</td>
</tr>
<tr>
<td>2</td>
<td>7/29/99 (summer)</td>
<td>28.0 (1.2707)</td>
<td>3.2 (0.4082)</td>
<td>3.0 (0.0000)</td>
<td>3.0 (0.0000)</td>
</tr>
</tbody>
</table>
being comparable with the average lengths (Table 3.1). The Age 0+ samples taken in winter included the smallest fish (TL of 6 cm) usually caught during the study.

It is also apparent from this analysis that there is some degree of seasonal habitat segregation among the cohorts. The younger age cohorts (Age 0+, Age 1+ and Age 2+) were resident in the study area (60-70 m depth on average) throughout the year while the older fish (Age 3+) are present only during the summer (Figures 3.1 (a-c)).

**Diet Analysis**

**Sample Size Determination.** The curve of cumulative increase in prey taxa represented in the diet with the number of guts analyzed approached a clear asymptote after about 21 guts were processed for Age 0+ fish (Figure 3.2). Age 1+ and 2+ combined appeared to have a narrower range of prey taxa selected in their diet with the curve reaching stabilization at 16 guts (Figure 3.2). Based on this graphical technique, 21 and 16 guts were the sample sizes (n) needed for dietary analysis for Age 0+ and Ages 1+ and 2+ fish, respectively. This criterion in selecting sample sizes for the two age classes was rarely met for Site 1 samples due to fewer plaice being caught at this site during the study (Table 3.2). However, the sample sizes analyzed for Site 2 were in close compliance with this guide on selecting an appropriate number of guts to analyze (Table 3.2). Overall, a total of 227 guts with food items present were analyzed, and a very small percentage of empty guts were encountered in the samples (Tables 3.2, 3.3 and 3.4).

**Trophic Ecology.** The trophic ecology of *H. platessoides* in the two selected age categories, Age 0+ and Age (1 & 2)+, is summarized in Tables 3.3 and 3.4, respectively. Three major taxa appeared to be consistently predominant in the summer diet of Age 0+ fish at both sites, both in % occurrence and % numbers (Table 3.3). These were
Figure 3.2. Cumulative increase in the number of new prey taxa included in the diet of juvenile *H. platessoides* with an increase in the number of guts processed for Age 0+ and Age (1&2)+ fish. The curve stabilized at 21 guts (n = 21) for Age 0+ fish and at 16 guts (n = 16) for Age (1 & 2)+ fish.
Table 3.2. Summary of the total number of guts processed (N), the proportions of guts with food present (% FP) and proportion of empty guts (% E) by year, site and age class. A total of 227 guts were processed with diet analysis done only on guts with food present, as summarized in Tables 3.3 and 3.4.

<table>
<thead>
<tr>
<th>Site</th>
<th>Age class</th>
<th>1998</th>
<th></th>
<th>1999</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Winter</td>
<td>Summer</td>
<td>Winter</td>
<td>Summer</td>
</tr>
<tr>
<td></td>
<td>N</td>
<td>% FP</td>
<td>% E</td>
<td>N</td>
<td>% FP</td>
</tr>
<tr>
<td>1</td>
<td>0+</td>
<td>0</td>
<td>0.0</td>
<td>0</td>
<td>100.0</td>
</tr>
<tr>
<td></td>
<td>(1 &amp; 2)+</td>
<td>0</td>
<td>0.0</td>
<td>0</td>
<td>100.0</td>
</tr>
<tr>
<td></td>
<td>Total</td>
<td>0</td>
<td>30</td>
<td>18</td>
<td>100.0</td>
</tr>
<tr>
<td>2</td>
<td>0+</td>
<td>12</td>
<td>100.0</td>
<td>0.0</td>
<td>47</td>
</tr>
<tr>
<td></td>
<td>(1 &amp; 2)+</td>
<td>13</td>
<td>100.0</td>
<td>0.0</td>
<td>31</td>
</tr>
<tr>
<td></td>
<td>Total</td>
<td>25</td>
<td>78</td>
<td>108</td>
<td></td>
</tr>
</tbody>
</table>
Table 3.3. Summary of diet of Age 0+ *H. platessoides* by site and season (n = number of guts analyzed) for the two years sampled. % O = % Occurrence; % N = % by number; Av.N (s.d) = mean number/gut and standard deviation. * Order: Cyclopoida.

<table>
<thead>
<tr>
<th>Prey Taxa</th>
<th>Site 1 Winter (n = 0)</th>
<th>Site 1 Summer (n = 18)</th>
<th>Site 2 Winter (n = 12)</th>
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<td>Winter (n = 11)</td>
<td>Summer (n = 24)</td>
</tr>
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<td>% N</td>
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<td>% O</td>
</tr>
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</tr>
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<td>Amphipoda</td>
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<td>Ophiuroidea</td>
<td>0.0</td>
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<td>0.0 (0.00)</td>
<td>7.1</td>
</tr>
<tr>
<td>Bivalvia</td>
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<td></td>
</tr>
<tr>
<td>Nuculanidae</td>
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<td>0.0 (0.00)</td>
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<tr>
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<td>0.0</td>
<td>0.0 (0.00)</td>
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Table 3.4. Summary of diet of Age (1 & 2)+ *H. platessoides* by site and season for the two years sampled. % O = % Occurrence; % N = % by number; Av.N (s.d) = mean number/gut and standard deviation. * Order: Cyclopoida.

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<th>% O</th>
<th>% N</th>
<th>Av.N (s.d)</th>
<th>Summer (n = 12)</th>
<th>% O</th>
<th>% N</th>
<th>Av.N (s.d)</th>
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<tr>
<td>Polychaeta</td>
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<td></td>
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<td></td>
<td></td>
<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Ampharetidae</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>91.7</td>
<td>80.1</td>
<td>9.8 (8.44)</td>
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<td>69.6</td>
<td>6.2 (4.39)</td>
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<td>0.4 (0.90)</td>
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<td>0.5 (0.66)</td>
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<td>-</td>
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<td>0.0</td>
<td>0.0 (0.00)</td>
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<td>-</td>
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<td>0.1 (0.29)</td>
<td></td>
<td>0.0</td>
<td>0.0</td>
<td>0.0 (0.00)</td>
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*Order: Cyclopoida.*
Table 3.4. Continued.

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<td>% N</td>
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<td>0.0</td>
</tr>
<tr>
<td>Cumacea</td>
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<td>0.0</td>
</tr>
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<td>0.0</td>
</tr>
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<td>0.0</td>
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<td>Other</td>
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</tbody>
</table>
polychaetes of the family Ampharetidae, amphipods and cyclopoid copepods (Order: Cyclopoida), an order that has benthic-dwelling species (Wells, 1988; Lalli and Parsons, 1997). Of the three, amphipods were not as common in the winter (Table 3.3 and Figure 3.3). Also, as will be seen below, the copepods were not represented as a predominant taxon in the grab samples.

In the Age (1 & 2)+ age category, besides the Ampharetidae and the amphipods, other polychaetes and bivalves were predominant in the diet of this age category. Apparently, the copepods were not an important part of the diet of fish ≥ Age 1+ fish, with the Bivalvia being included in the diet of these older fish (Table 3.4). Of this taxon, the family Nuculanidae (*Yoldia* sp.) were usually taken in larger numbers by the older age classes (≥ Age 2+) of fish whenever they occurred in the diet. The amphipods again were not common in the diet in winter for this age class too. As in Age 0+ fish, there were no apparent spatial (between-site) differences in the dietary patterns of this age class (Table 3.4 and Figure 3.4).

**Prev Abundance**

Results of macroinvertebrate analysis showed that most of the taxa that predominated the diet of plaice (Tables 3.3 and 3.4; Figures 3.3 and 3.4) were also predominant taxa in the grab samples (Figure 3.5; Table A2, Appendix). At both sites the density of taxa that were major constituents of the diet of juvenile *H. platessoides* were much lower than the ones that were not. The Ampharetidae (Polychaeta), a taxon that was important to both age categories of juvenile plaice, and the Bivalvia, a taxon only important to older fish together with the Ampharetidae, showed this trend (Figure 3.5). An even clearer illustration of predator influences on prey abundance was shown by the
Figure 3.3. Diet composition by predominant prey taxa (Table 3.3) of Age 0+ *H. platessoides*. The data for 1998 and 1999 were pooled by season within sites. Error bars are standard deviations.
Figure 3.4. Diet composition by predominant prey taxa (Table 3.4) of Age (1 & 2)+ *H. platessoides*. The data for 1998 and 1999 were pooled by season within sites. Error bars are standard deviations.
Figure 3.5. Abundance of predominant macroinvertebrate taxa in grab samples collected at the two study sites during the winter and summer of 1998 and 1999. The data are means of replicate grabs (usually three) per site per sampling date, and the error bars are standard error of the means.
comparison of the mean abundance of ampharetid polychaetes and amphipods in the study area, and the mean abundance of the two taxa in the diet of Age 0+ fish at Site 2 (Figure 3.6). As the abundance of these taxa increased in the diet, their abundance in the environment declined (Figure 3.6). However, the results for both spatial (site) and temporal (season) variation in the mean density of both taxa tested with MANOVA were not significant for both site ($\lambda = 0.7103; F_{[2,4]} = 0.816; p = 0.505$) and season ($\lambda = 0.7262; F_{[2,4]} = 0.755; p = 0.527$), respectively (Table 3.5).

Figure 3.7 indicated that most abundant taxa were adequately included by the choice of sample sizes chosen, though when taking rare taxa into account, 10 grab samples would be the minimum number of grabs samples required.

**Sediment Organic Content**

Results of the analyses of the vertical distribution of organic content showed that the sediment organic content in winter showed higher levels from the surface to 4-7 cm before it began to decline (Figure 3.8 (a)). A similar trend was also present in the summer (Figure 3.8 (b)). The vertical changes seemed to be more pronounced at Site 1 than Site 2 (Figure 3.8 (a & b)). Horizontally, there was generally about 1.5 % more organic content at Site 1 than at Site 2 where levels were about 5 %. Besides this spatial difference, the summer appeared to have slightly higher levels than that in the winter (Figure 3.9).

**Discussion**

This chapter focused on the population, habitat and trophic ecology of the most predominant species of fish in the study area, *H. platessoides*. As revealed in the correlation done in Chapters 1 and 2, the abundance of this species did not show any
Figure 3.6. Comparison of the trends in the abundance of two predominant prey taxa in the diet (guts) of Age 0+ fish at Site 2 with that of grab samples (grabs) from the site over the study period.
Table 3.5. MANOVA results of significance test of spatial (site) and temporal (season) variation in mean density (number/m²) of Ampharetidae (Polychaeta) and Amphipoda from grab samples. These two taxa were among the predominant prey taxa that occurred in both Age 0+ and Age (1 & 2)+ H. platessoides.

<table>
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<th>Factor by species</th>
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<th>Multivariate*</th>
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<td></td>
<td>F [df]</td>
<td>p</td>
</tr>
<tr>
<td>Site</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ampharetidae</td>
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<tr>
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</tr>
<tr>
<td>Both</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Season</td>
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<td></td>
</tr>
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<td>Ampharetidae</td>
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<td>0.320</td>
</tr>
<tr>
<td>Both</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

* The multivariate test statistic used was Wilk's Lambda (λ).
Figure 3.7. Cumulative increase in the number of new macroinvertebrate taxa included in the grabs with an increase in the number of grab samples processed. The lowest level of taxonomic resolution is to the family level. An asymptotic sample size of 10 grab samples is the minimum number of grabs needed to account for most taxa, including rare taxa too. Samples from Site 2 taken in July, 2000 were used for this figure.
Figure 3.8. Profile of mean percentage by sediment dry weight of organic content with sediment depth at the two study sites during the winter and summer of 1998. Error bars are standard deviations based on three cores per site per sampling date.
Figure 3.9. Mean organic content by percent dry weight of sediment for sediment samples obtained from grab samples between the two sites for winter and summer pooled for both 1998 and 1999. Error bars are standard error of means.
correlation with any composites of the environmental variables measured in the field that were ordered by principal component analysis. A possible explanation offered in Chapter 2 was that these environmental variables were not heterogeneous enough over the spatial scale (4 km x 4 km) of this study, thereby resulting in having no significant influence on the occurrence of $H. platessoides$ between the two sites. However, there was a temporal trend, but this was not statistically significant.

The data used for this chapter were age composition, distribution and diet of $H. platessoides$, as well as the abundance of its prey species. These variables can directly indicate small-scale habitat use and have a greater potential of revealing any impact of shrimp trawling. Not only was this approach more direct, it involved variables that were likely to be sensitive to perturbations over spatial scales relevant to the microhabitat of this species. The magnitude of any impact relevant at the microhabitat level has a greater chance of impacting aspects of the ecology of $H. platessoides$. If the ecological variables such as the abundance of prey organisms of juvenile plaice are sensitive to this magnitude of impact, a detectable response in the feeding ecology of $H. platessoides$ could occur. Consequently, changes in spatial and temporal dietary patterns may become apparent.

**Population Structure and Habitat**

Recruitment is a very important process regulating the dynamics of fish populations by adding new cohorts. In boreal and temperate latitudes where recruitment events occur over distinct time periods as a result of seasonal reproductive patterns, different cohorts are identifiable by length frequency analysis (Pitcher and Hart, 1982). The use of this technique assumes that: 1) fish length within each age class is normally distributed; 2) fish spawn over a relatively short spawning period; and 3) growth within
an age class is uniform and rapid (Sullivan et al., 1990; Deveries and Frie, 1996). The applied value of length frequency or cohort analysis in fisheries science is in fish stock assessment (Hilborn and Walters, 1992). Virtual population analysis (VPA) is one stock assessment technique that specifically requires knowledge of age-structure in a population to trace back the population of each age class in preceding years, using catch data containing length and age information (Hilborn and Walters, 1992; Wallace et al., 1994). From this estimate of the abundance of recruits, their subsequent age class strengths, any mortality losses, and the size of a harvested stock can be determined (Lapointe and Peterman, 1991; Lapointe, et al., 1992; Wallace et al., 1994). Age-class information is important in VPA because not all age classes contribute equally to recruitment (Marteinsdottir and Thorarinsson, 1998), and mortality and fecundity are both age and size dependent in most fish species (Pitcher and Hart, 1982).

Length-frequency analysis can also be useful in ecological studies, to trace recruitment events and to assess the productivity of aquatic systems (Allen, 1971). This technique is often used when the determination of Production: Biomass ratios (P/B) are needed to calculate secondary (heterotrophic) production in species with multiple age classes in a population. It is commonly used in invertebrate studies in aquatic systems (Warwick, 1980; Brey, 1990; Benke, 1996).

Length frequency analysis was used in this study mainly to identify age cohorts in *H. platessoides*, a spring spawner in the Northeast United States (Bigelow and Schroeder, 1953; Colton, Jr. et al., 1979; Sherman et al., 1984, 1987). Since the focus of the study was to investigate the impact of shrimp trawling specifically on immature fish, it was important to identify the age classes in the samples. Tracing the spatial and temporal
distribution patterns of these classes is important to interpreting trawling impact, especially from the perspective of its impact on microhabitat and prey resources. Diet and microhabitat use by fish are known to vary ontogenetically (Gerking, 1994). Overlooking ontogenetic differences could mask any important impacts affecting juvenile fish if the magnitude of impact varies among different age classes. In an already statistically "noisy" system such natural variations could result in a statistically significant size-related interaction term, rendering any interpretation of ecological impacts due to habitat perturbation from trawling impossible.

The results of the analysis done on H. platessoides indicated that there were four age classes, Age 0+, 1+, 2+ and 3+, of this species present in the study area (Figures 3.1 (a-c); Table 3.1). The first age class (Age 0+) had a modal size of 6.0 cm TL during their first winter (Figure 3.1 (a)). Based on published records (Bigelow and Schroeder, 1953) this would be fish that settled towards the end of the previous summer (early August). By the following summer their modal length was 11 cm TL (Figure 3.1 (b)), an increase by 5 cm TL in five months, since my winter and summer samples were taken in mid February and mid July, respectively. A similar magnitude of change in size was also evident for the next age class, Age 1+ fish, whose modal length in the winter was 15 cm TL (Figures 3.1 (a & b)). Five months later this age class was now Age 2+ fish with a modal length of 18 cm TL. The next age class, Age 3+, appeared for the first time in the summer. Its modal length was about 28 cm TL. The first three age classes (Figure 3 (a-c)) were confirmed by scale aging as fish approaching their first, second and third years of growth, respectively (Table 3.1). As indicated by the presence of maturing gonads in some specimens of Age 3+ fish, this fourth age class comprised sub-adult fish. Age and growth
on this species in the Gulf of Maine have reported 3.8 years as the age at which 50% of
the fish of this age class acquired maturity, with corresponding length (size) at which this
was achieved to be 29.7 (Sullivan, 1982; Walsh, 1994). These figures are consistent with
the data from this study.

Length frequency analysis was done in this study to enable the identification of
length classes representing juveniles. Age and length of fish have a curvilinear
relationship, especially in older fish. Factors such as nutrition, temperature and
competition can affect the growth rate among fish of a given age. This means that a fish
of a given length may not be the same absolute age as a conspecific of the same size.
Hence, there is always an overlap at the tails among adjacent length frequency
distribution curves representing successive cohorts. This technique may be useful if the
assumptions listed at the beginning of this section are met. Therefore, if the objective of
an investigator is to know accurate ages, age determination using scales or otoliths is
recommended.

In winter, the Age 3+ fish were absent in the study area (Figure 3.1 (a)), but were
present during the summer (Figures 3.1 (b)). This temporal pattern in distribution could
be due to a movement into deeper water in winter by this larger age class. Short seasonal
(winter) movement into deeper water has been reported to occur in larger H. platessoides
in the Gulf of Maine (Bigelow and Shroeder, 1953). Seasonal bathymetric movements
were also reported in the Rex and Dover soles on Heceta Bank off Oregon, being more
abundant in deeper water stations during winter than summer (Pearcy, 1978).

Another behavioral response that could result in the observed seasonal
segregation spatially of the fish could be their dispersion patterns. On the Grand Banks of
Newfoundland, Canada, *H. platessoides* were found to be more aggregated in their distribution in winter than in other seasons (Morgan and Brodie, 1991). If this behavior is also present in fish in the area sampled in this study, the sampling method would undersample during periods of such behavior. That is, if the aggregations were so dispersed spatially, then towing a small trawl (16-foot otter trawl in this case) for a short duration (10 minutes) would have a lower probability of sampling through any such aggregations resulting in the under-sampling of these older fish.

Fewer fish of the three first age classes too were caught in winter. For most fish populations the older a cohort gets, the fewer the numbers because of mortality (Pitcher and Hart, 1982; Sogard, 1997). Hence, when there is aggregation of older age classes with fewer numbers, their exposure to the gear will also decline compared to any aggregation in younger cohorts behaving similarly. Consequently, they can be totally missed by tows of short duration using smaller nets as was done in this study. This could be another explanation of the absence of the older fish in the study area.

These ontogenetic differences in behavioral responses with respect to depth and season can be explained physiologically. Seasonal and ontogenetic effects on thermal preferenda exist in many fish species (McCauley and Huggins, 1979). Seawater temperature is one variable that is important to the physiology of fish, and it also changes seasonally and bathymetrically.

Shrimp fishing season is from winter to early spring in the area of the study (Haynes and Wigley, 1969; Dobbs, 2000). If the first explanation of the absence of Age 3+ fish is correct, then age classes 0+ to 2+ are residents in the study area, making it a potential nursery area for young *H. platessoides*. Disturbances to the benthos from shrimp
trawling would surely have trophic implications for these younger fish when prey resources are also impacted by this activity. Consequently, the potential nursery role of the study area to young fish may be affected. The nature of such an impact will become apparent in the discussion on their trophic ecology.

**Trophic Ecology**

The impact on food and feeding is one of the concerns of mobile gear-related habitat perturbations. In this section the trophic ecology of *H. platessoides* will be discussed to provide a basis for some assessment of any ecological impact of shrimp trawling on this species. As reviewed in the Introduction Chapter of the dissertation, the contact of trawl doors with the bottom as the net is towed tends to scour out the top 4-10 cm of the sediment. The analysis of the vertical distribution of organic content in the sediment at the study sites of this study showed that levels were highest from the surface down to the 4-7 cm depth interval (Figure 3.8).

It was observed in this study that the Ampharetidae (Polychaeta), one the most important prey taxa for juvenile fish, have tubes that extend vertically into the sediment. Tube lengths of 4-5 cm were estimated from another tube-dwelling polychaete family, the Maldanidae, that build longer tubes to accommodate their generally longer bodies compared to the Ampharetidae. The Shipek grab sampler used to sample the benthic macroinvertebrates in the study is designed to penetrate to 10-11 cm maximum in soft sediments. The data collected with this device (Figure 3.5) showed that most of the predominant prey taxa of *H. platessoides* (Tables 3.3 and 3.4; Figures 3.3 and 3.4) do occur within this depth range. Obviously, the 4-10 cm scouring depths possible from otter trawl doors (Brylinsky *et al.*, 1994; Kaiser *et al.*, 1996, Gordon *et al.*, 1998; Pilskaln *et al.*
al., 1998; DeAlteris et al., 1999) has ecological-impact implications for these prey taxa. Furthermore, it also has trophic implications for juvenile *H. platessoides*.

Figure 3.6, based on the diet of Age 0+ fish, indicated that there was a strong trend in the occurrence of the ampharetid polychaetes and amphipods in the diet with respect to availability in the habitat. It showed the basic characteristics of a "predator-prey curve" often seen in ecological textbooks where an increase in predation rate is tracked by a decline in the abundance of the prey, but usually separated by a delay period that is dependent on the recruitment interval of the prey. A classical example in marine ecosystems was the tracking of plankton production by herring populations in the North Atlantic that led Cushing to propose the "match-mismatch" hypothesis (Cushing, 1990). A delay period was not visible in this case due to a lack of temporal resolution.

The summer is expected to be a period of increased abundance and production at temperate and boreal latitudes. It does not seem to be the case here with macroinvertebrate abundance; in all four of the important prey taxa their winter densities were higher than that of the summer at both sites (Figures 3.5). In contrast, the sediment organic content analyzed from grab samples showed some increase during this season (Figure 3.9), indicating some input of organic matter to the benthic community.

How could this lack of a match be explained? As discussed in Chapter 1, there was a greater abundance of juvenile fish in the study area in the summer than in the winter. The population structure analysis above showed also that sub-adult fish were also present during this season. Considering this information together with the opposite temporal trends in the abundance of prey taxa in the habitat and the mean numbers occurring in the diet of Age 0+ fish apparent in Figures 3.6, fish predation might be a...
possible explanation of this trend. However, a specific study focusing on the temporal
dynamics of both benthic microinvertebrate populations and fish at higher temporal
resolutions (say sampled monthly over ≥ 1 year) would enable a better explanation of this
pattern.

The results from the MANOVA on the density of Ampharetidae and Amphipoda showed no significant spatial and temporal differences on the abundance of both taxa (Table 3.5; Figure 3.5). Even if predation was causing this trend (Figure 3.6), the large-scale influence of dispersal and recruitment of marine invertebrates to open continental shelf habitats may mask the effect of any small-scale localized impact from predation pressure, because of its greater magnitude of influence both spatially and temporally. However, the pattern does have potential use in generating hypothesis for studies designed to specifically test the possible effect of predation on benthic macroinvertebrate communities on the continental shelf. Such studies have been done in shallow water coastal/estuarine habitats where the presence and absence of fish was experimentally manipulated using meshed-cage enclosures without affecting prey recruitment. A study that investigated suitable release sites for reared juvenile winter flounder, *P. americanus*, in a stock-enhancement research program in New Hampshire (Fairchild, 1998) is one example of such manipulative field studies. Sogard (1992) did a similar study in New Jersey with juvenile *P. americanus* and juveniles of two other species to investigate food availability and other factors important to enhancing the nursery role of estuarine habitats to juvenile fish. Unfortunately, such manipulative experiments may be difficult to do on the continental shelf.

The continental shelf habitat within the southwestern Gulf of Maine appears,
based on the predominance of four juvenile age classes (Table 3.1; Figure 3.1) and the
diet analysis (Tables 3.3 and 3.4; Figures 3.3 and 3.4), to be an important nursery area for
*H. platessoides*. A previous study on discards of commercial shrimp trawling (Howell
and Langan, 1992) also showed a high proportion of Age 1+ and Age 2+ fish in a study
that encompassed a larger area within the southwestern Gulf of Maine. The prey
abundance data analyzed from grab samples (Figure 3.5) showed that the soft-bottomed
muddy substrate prevalent in the study area also provided a suitable habitat for taxa of
macroinvertebrates important to the diet of juvenile plaice. Sediment grain sizes suitable
for burying and feeding are known to determine juvenile flatfish distribution (Jager *et al.*, 1993; Gibson, 1994). Hence, the study area within the southwestern Gulf of Maine
continental shelf is an important nursery ground for *H. platessoides*.

The prey available (Tables 3.3 and 3.4) consisted of taxa of a broad range of sizes
allowing different age classes of fish to coexist without too much competition for food. It
is evident from the diet analysis (Tables 3.3 and 3.4; Figures 3.3 and 3.4) that the
Ampahretidae (Polychaeta) are an important prey for all age classes of fish, but from
personal observation during the analysis, the different age classes feed on different sizes
of the prey. In addition, Age 0+ fish complement this prey with cyclopoid copepods,
while the older age classes do the same with bivalves (Figure 3.3 and 3.4). A shift in the
size of bivalves was also observed during the analysis. Age 2+ fish and the sub-adults
(Age 3+) tended to target a particular family of bivalves, the Nuculanidae (*Yoldia* sp.). It
is not clear from the data whether this is due to their larger size or because of their
abundance in the study area. There were some specimens of Age 3+ fish (diet not
analyzed due to fewer fish of this size class) caught in the summer seen with distended
guts packed with this bivalve. This indicated that the older fish fed intensively on them when patches of this bivalve were encountered. A similar intensive feeding behavior was indicated from examining guts distended with Ampharetidae. In this case, it seemed juvenile *H. platessoides* fed on these patches by indiscriminately "bulldozing" and ingesting large quantities together with any other fauna, such as amphipods, occurring within the patch. The results of an experimental study (Hill et al., 2000) on the foraging behavior of juveniles *Pleuronectes platessa* (Pleuronectidae), using the bivalve, *Donax vittatus*, as the prey indicated that this species uses some localized cue associated with the prey to locate the prey within a patch. A characteristic "arch-lunge-bite" sequence of behavior in which the anterior part of the body was arched and then flexed rapidly to move the mouth around the prey was then used to attack the prey.

Diet preference is ontogenetically plastic and is often influenced by prey availability, size of prey relative to fish size, and foraging efficiency (Gerking, 1994; Jennings et al., 1997). The taxa shown to be predominant in the diet of *H. platessoides* in this study seemed to be influenced by the bottom-type of the study area, which in turn influenced the composition of the benthic macroinvertebrate community. A study that included trophic aspects of *H. platessoides* in the western Gulf of Maine, reported brittle stars (Ophiuroidea) to be the predominant prey (Packer, et al., 1994). The study site was in deeper water (148-156 m) with a silt-clay substrate and based on fish largely caught by commercial trawlers. The size of fish ranged between about 23 cm TL (= Age 2+ as analyzed in this study) and 68 cm TL with the modal length being about 35 cm TL. The data in that study was based largely on adult fish sampled in deeper water. Although the sediment type was similar to that in the location of this study, the pattern may be
influenced by size and age of fish, and possible bathymetric differences in the composition of prey species. Langton (1982) reported that *H. platessoides* fed on a variety of benthic animals, but echinoderms predominated the diet of larger (> 45 cm) fish. Smaller-sized ophiuroids did occur in the diet, especially in the summer in Age 1+ & 2+ fish, in this study (Table 3.4), but were not one of the predominant prey taxa. However, bivalves were found to be included more in the diet of these larger age classes (Figure 3.4). The Family Nuculanidae was the common bivalve observed in the diet of Age 3+ fish, usually present in the study area only in the summer. An analysis of the dietary guild structure of the fish community of the Northeast United States continental shelf ecosystem (Garrison and Link, 2000(a)) classified *H. platessoides* within the predator category "benthivore". The diet of fish within this size range given in Garrison and Link (2000 (a)), deduced as Age 0+ and Age 1+ from the age analysis done in this study (Table 3.1; Figure 3.1), were grouped into a guild that also included three other flounder species (yellowtail, winter and witch flounder). On the other hand, the larger *H. platessoides* had a diet classified as similar to that of the haddock, *Melanogrammus aeglefinus*, and large-sized ocean pout, *Macrozoarces americanus*. Of a total of 40 species analyzed (Garrison and Link, 2000 (a)), there were similar size-related differences in the diet of other species as well, supporting the notion of ontogenetic plasticity in diet preferences within a broader predator category.

Despite this plasticity, for any fish species, their preferred range of prey taxa is restricted at a given age. It can be noted from Figure 3.2 and Figure 3.7 that the range of taxa selected by juvenile *H. platessoides* is about 50 % of the available taxa. However, taking into account their preferences for certain taxa based on factors such as abundance,
size, and foraging efficiency, this percentage would be further reduced. As demonstrated in the analysis presented above, the taxa that predominated in the diet were among the predominant taxa in the environment as well.

The results from this and the two previous chapters should now set the stage for the last chapter. This final chapter will consider: 1) issues relevant to the effective design of ecological impact studies; 2) the appropriate spatial and temporal scales of study with respect to the natural variability in variables important to an impact study; and 3) to provide a synthesis of the results of this study within the context of ecological impacts of shrimp trawling. Some recommendations relevant to fisheries management will also be offered.
CHAPTER 4

ECOLOGICAL IMPACT OF SHRIMP TRAWLING ON JUVENILE BENTHIC FISH: SYNTHESIS AND RECOMMENDATIONS

Abstract

Ecological impact studies in marine ecosystems are very challenging to do due to time and financial constraints, confounding influences from non-targeted environmental variables on results, and logistic and practical constraints. When possible to do, impact studies can be of two types; a “field assessment” or a “field experiment” impact study.

The distribution and abundance of juvenile benthic fish are influenced by temperature, salinity, bathymetry and bottom type. These variables have a large-scale spatial and temporal influence. Over small spatial scales such as that used in this study (4 km X 4 km) they are less heterogeneous. Consequently, there was no significant correlation in the abundance of juvenile fish with these environmental variables.

Results on the analysis of the possible impact from sediment scouring related to shrimp trawling on trophic aspects of the different age classes of juvenile H. platessoides were insightful. Conclusive interpretations were not possible without data on the intensity of scouring and prey patch size and distribution data.

Four recommendations from the results of this study are proposed for consideration in fisheries management. These include: 1) the need to combine both population and community level data to interpret impact from mobile fishing gear; 2) the need to put a greater emphasis than has been on “field experimental” studies; 3) the
selection of sensitive variables in "field assessment" studies; and 4) the need to quantify the intensity of impact from trawls and other mobile gear on benthic habitats.

Introduction

Studying ecological impacts of any form of perturbation, either natural or anthropogenic, in marine ecosystems can be challenging. The complex interconnections and interactions among different biological and environmental processes at different ecological levels, demands a clear insight into the theoretical functioning of such complex systems in order to design suitable studies. With such insight careful decisions on: 1) variables to measure; 2) metrics to measure them with; 3) spatial and temporal scales at which to measure them; and 4) suitable methods of data analysis can then be made to answer important questions forming desired research objectives. Unlike chemical and physical systems, studies of biological systems are more dynamic spatially and temporally, and also are influenced by differences due to the inherent biological variations among taxa.

Despite this complexity, approaching such studies using the power of the scientific method by systematically following some of the processes can provide some understanding of the links in the complex interconnections. If this is not possible any accomplishments of the research would at least provide insights into the design of additional research. When this is achieved, the major advantage of community and ecosystem-level research is that a holistic understanding of perturbed systems can be acquired, and better management decisions can be made (Fogarty and Murawski, 1998;
Murawski et al., 2000). This has been the premise of this research project, and hopefully this final chapter of the dissertation will illustrate this point.

As reviewed in the Introduction of the dissertation, ecological impacts principally transmit through the food web and are detectable by studying processes at the community and ecosystem level. To study the impacts due to trawling in this context is not easy. The complex nature of ecological processes characteristic of these higher levels of ecological organization, and environmental ‘noise’ due to natural among-site variations in habitat attributes, can confound and mask the existence of any real impact. Therefore, questions of interest, and variables to measure, must be translated into appropriate and sensitive sampling designs. Careful considerations must be given to relevant spatial and temporal scales at which to study the impacts (Langton et al., 1994; Langton and Auster, 1999). All possible precautions must be taken to control for any confounding influences of natural environmental variations in the system being studied (Underwood, 1993; 1996; 1997; Osenberg et al., 1996). Physical disturbance on the benthic environment from trawling is a process that occurs over spatial scales ranging between 10-100 km² and over a temporal scale of months (Langton et al. (1994). High-resolution data on the distribution of fishing effort using such gear is presently lacking (Langton and Auster, 1999). Trawling intensity within 30' latitude by 30' longitude boxes on the Middle Atlantic Bight showed that the shelf area south of Nantucket, Nantucket Shoals, was heavily trawled in 1985, with an area 3 times greater than the actual area trawled (Churchill, 1989).

This study investigated the ecological impacts of shrimp trawling on juvenile fish with a detailed analysis done with respect to the ecology of H. platessoides. To assess the
magnitude of this type of disturbance, the sampling design in this study has taken the
above range of spatial and temporal scale into account, in order to increase the chances of
detecting any impact of this method of fishing on the ecology of juvenile fish. An
overview of important considerations in designing ecological impact studies will be
discussed in this chapter, together with the principal results of this study, and their
ecological implications on the juveniles of the selected fish species. Recommendations
based on these results for potential application in environmental and fisheries
management will also be proposed.

Ecological Impact Studies

Ecological impact assessment is a growing field of applied ecology. Ecological
impacts are known to many by its more restricted terminology as "environmental
pollution". This term connotes the occurrence of a pollutant in the environment that
results in extreme impacts such as species loss or poor water quality. Besides an increase
in management and political attention to this subject, there has also been a lot of effort to
develop powerful sampling design techniques to be able to detect such impacts in order to
provide appropriate scientific advice on management and political decisions (Stewart-
Oaten, 1996). Green (1979) is a popular text on this subject, and another publication
(Schmitt and Osenberg, 1996) that had attempted to integrate many of the important
developments on the subject into a single source of reference was published recently.
Both of these references are focused on examples in aquatic systems, perhaps because the
world's freshwater ecosystems and oceans have always been a source of natural products,
and a dump for waste products from domestic and industrial sources, resulting in many
impacts caused to these ecosystems. They discuss most of the important sampling-design issues in conducting ecological impact assessment studies. Two other related texts were also published recently. These are focused more on statistical aspects of the design of field ecological experiments and appropriate techniques of data analyses (Scheiner and Gurevich, 1993; Underwood, 1997).

The use of the phrase "ecological impact" implies a change in baseline levels of one or more variables among abiotic and biotic variables of interest. Abiotic variables are usually physico-chemical variables while biotic ones are usually some living entity in the environment; a plant, an animal, or a microorganism taxon. Changes in processes such as growth, reproduction, distribution and abundance of such entities become the focus of measurement in order to determine if an impact by a perturbation has occurred. In any quantitative ecological studies the choice of variables, the metric to measure them with and the spatial and temporal scale of measurement (Schneider, 1994) are very important decisions to make in order to ensure that an appropriate magnitude of change can be resolved by the method of measurement. It is recommended that in an impact study the impact must be defined (Underwood, 1992, 1993, 1996). For example, when abundance is the metric of measurement it should be defined as: "some difference in the change of mean abundance, or time-course of mean abundance in the defined location from before to after the disturbance, compared with such changes in the control locations" (Underwood, 1993).

Fishing is a means of harvesting natural products. The direct removal of fish biomass (Beverton, 1994; Knauss, 1994; Waters and Maguire, 1996; Botsford et al., 1997; Fogarty and Murawski, 1998; Pauly et al., 1998; Hutchings, 2000) and the...
destructive consequences of certain types of fishing methods (Caddy, 1973; Langton and Robinson, 1990; Messieh et al., 1991; Bergman and Hup, 1992; Brylinsky et al., 1994; Kaiser and Spencer, 1996) both result in ecological impacts of marine ecosystems (Jennings and Kaiser, 1998).

**Types of Impact Studies**

Ecological impact studies can be of two types. A "field assessment" study is one where a site with a known perturbation is studied relative to another where the specific perturbation is absent. On the other hand, a "field experimental" study is one in which impact and control sites are experimentally manipulated by the researcher (s) with the appropriate replication of treatments, and the effect of the experimentally-induced impact on the dependent variables of interest are tested (Osenberg and Schmitt, 1996).

It was the aim of this study to analyze any ecological impact of shrimp trawling on the juveniles of selected fish within the southwestern Gulf of Maine. Given the spatial characteristics and the pre-existing nature of this impact, a "field assessment" study was the appropriate choice for the type of impact study to undertake. An area of known shrimp fishing on the inshore continental shelf of the southwestern Gulf of Maine was compared with a smaller area nearby without the impact. As shown in the previous chapters, both abiotic and biotic variables were measured. A range of data analysis techniques including correlation, classification, graphical analysis, ordination, and analysis of variance based on a multivariate data set was used to analyze any impacts.

**Sampling Design of this Study**

The replication of treatments, shrimp trawling in this case, is a recommended statistical requirement in experimental design (Osenberg et al., 1996; Scheiner and
Gurevich, 1993; Underwood, 1993, 1996, 1997) needed to detect an impact. It is an impossible goal to achieve in most "field assessment" impact studies such as this study where an impacted site is compared with a control (non-impacted) site. In this situation the researchers have no control over where the impact occurred and therefore cannot replicate this treatment. Furthermore, the effect size or the magnitude of the impact is often fixed relative to the history of the impact. Besides sufficient sample sizes, one should also replicate the control treatment by selecting sites at locations that are relatively comparable spatially in terms of the constituent environmental conditions to increase statistical power. Using appropriate statistical techniques any impact can be analyzed (Glasby, 1997; Schmitt and Osenberg, 1996; Underwood, 1992, 1993, 1996).

It was both costly and practically impossible to optimize the sampling design by replicating sites within treatments ('trawl' and 'no trawl') to achieve the statistical rigor outlined above in this study. The minimum requirement of replicating at least the control site was not possible because this treatment was spatially confined to start with; there were no other comparable areas nearby. The control site was within a small stretch of the continental shelf interspersed with patches of coarse topography, and was located adjacent to the shrimp trawling ground.

Replicating the impact treatment was not possible, both from cost and practical perspectives. It would have involved selecting a site further south of the study area, separated by about 4-5 km apart. That would have doubled the ship-time cost, rendering the project quite expensive to do. When the number of samples to process is taken into account, a time-related cost becomes apparent. There was also a practical problem. A problem commonly faced by this kind of studies is that an anthropogenic, or natural,
perturbation is not replicable spatially and temporally by the same process. Even if it was, it may not have occurred within a spatially comparable location given the usual heterogeneity in environmental variables both spatially and temporally (Underwood, 1992, 1993; Raimondi and Reed, 1996). If impacts were anthropogenic, it would be unethical to replicate impacts just to meet a statistical requirements of conducting a suitable impact assessment study (Underwood, 1992).

Impact studies are often undertaken without prior knowledge of the variables adversely impacted, but usually with some theoretical expectation of ones that could potentially be affected. If optimum situations prevailed for the design of a statistically sound "field assessment" impact study, and adequate funds were available, these sorts of studies are possible to do. They are best done on a large-scale, preferably involving multidisciplinary teams of researchers engaged on separate aspects (e.g. benthos, fish, physical oceanography) of the large project. This would enable the effective collection of multivariate data sets, permitting the determination of the variables that are most sensitive to a given level of impact using rigorous statistical analysis. This is not possible to accomplish at the scale of a dissertation project such as this one.

Different types of mobile gear are in use and dredges and trawls are the principal types used. The nature and magnitude of their impact on benthic communities will depend on a number of factors. They include: 1) their design, 2) size, 3) weight, 4) tow speed, and 5) the oceanographic environment (depth, currents, and bottom type) at the location within which the gear is used. Given also the complex interconnections between biotic and abiotic process in marine ecosystems, it is important to focus on a specific gear type and where possible, within a specific fishery. This would enable an effective
accomplishment of research objectives, and also permit clear management guidelines to be developed and implemented.

In this study the main impact of interest was the scouring and scraping action of shrimp trawls on the bottom in the study area. The existence of the shrimp fishery in this area (Haynes and Wigley, 1969; Howell and Langan, 1992; Ross and Hokenson, 1997) provided the opportunity to focus spatially in order to allow the interpretations of the results within an applied context easier. Using information from David Goethel, an experienced fisherman from Hampton, who is very familiar with the fishing grounds within the southwestern Gulf of Maine, Site 2 was selected as the impacted site (see Chapter 1 for study site descriptions). Besides Goethel's information, trawling activity was confirmed in the vicinity of Site 2 during a number of sampling cruises (July 1998, November 1998 and July 2000). Boats often moved in a loop over Site 2 towards Site 1 and back. A plan to obtain fishing logbook data on trawling activity in the study area was abandoned after a discussion on obtaining such data with Roland Barnaby, a University of New Hampshire Extension Educator, proved to be difficult to get. Another attempt to detect trawl tracts on the bottom in the study area using side-scan sonar in March 2001, in collaboration with Klein Associates, a New Hampshire-based company specializing in side-scan sonar technology, was prevented by bad weather. These difficulties hindered the quantification of trawling intensity. Hence, shrimp trawling was assigned as a categorical variable; Site 1 classified as the "no trawl" site and Site 2 as the "trawl" site in all the analyses.

On a positive note, these difficulties elucidate the common problems often encountered in ecosystems-oriented studies concerning applied problems. It should not
discourage us from doing such studies, but direct us to come up with better strategies to plan and execute them more effectively. Efforts must be made to engage multidisciplinary teams to both pool expertise and resources to collect suitable multivariate data sets to find solutions to these applied problems more effectively. Current developments in computer technology and the readily available multivariate statistical packages should allow us to do this effectively.

**Summary of Results and Impact of Shrimp Trawling**

The principal objective of this study was to study the ecology of the juveniles of selected benthic fish species in relation to shrimp trawling in the southwestern Gulf of Maine, targeting the northern shrimp, *Pandalus borealis*. The overall hypothesis tested in the study was that shrimp trawling gear causes physical disturbances by scraping and scouring the benthic habitat, impacting habitat and trophic attributes important to resident juvenile fish populations, consequently causing further declines in their abundance besides that due to commercial bycatch and natural mortality. To test this hypothesis, juveniles of the four selected fish species: 1) American plaice (*H. platessoides*); 2) winter flounder (*P. americanus*); 3) silver hake (*M. bilinearis*); and 4) Atlantic cod (*G. morhua*), were sampled at the two sites, as well as a suite of environmental variables (both abiotic and biotic) perceived to be important components of the habitat of these fish. A wide range of statistical and graphical techniques were used to classify, correlate and assess any differences in fish abundance with respect to these variables between the two sites. This section will summarize the major results of this study and provide some
interpretation with respect to any implications of the ecological impact of shrimp trawling.

**Fish Community**

The overall fish community was analyzed in Chapter 1 to determine its community structure and to see how this varied spatially and temporally. The results of cluster analysis confirmed that there was a difference in the composition of predominant species constituting the community at each study site. Site 1 was dominated by the longhorn sculpin, *M. octodecemspinosus*, while Site 2 was dominated by American plaice, *H. platessoides*. The red hake, *U. chuss*, was equally dominant at both sites.

Results of community structure analysis based on diversity indices indicated that there was a greater seasonal variation, mainly due to temporal changes in the influx of species in the study area.

A correlation between the two predominant fish species, *M. octodecemspinosus* and *H. platessoides* and a number of environmental variables: 1) depth; 2) temperature; 3) salinity; 4) % gravel (all percentages were appropriately transformed); 5) % sand; 6) % mud; 7) abundance of the sand shrimp, *C. septemspinosa* (bycatch); and 8) abundance of starfish (bycatch) was done. These environmental variables, ordered into composites (principal components (PCs)) with PCA, revealed that the abundance of only *M. octodecemspinosus* was predictable by PC2. This PC was a composite variable accounting for temperature and salinity effects, while PC1, a composite accounting for depth and the nature of the bottom sediment had no predictive value on the dependent variables. Both of these variables, especially temperature, were found to vary greatly by season, further explaining the seasonal fluctuation in community structure.
These results indicate a broad scale seasonal pattern in the fish community of the southwestern Gulf of Maine largely influenced by physical variables. The difference in the predominant species between the two sites seemed to be influenced by some differences in bathymetry and sediment particle-size composition. Fish are adapted to small-scale habitat variations, and would be resilient to any effect of the scrapping and scouring effect of shrimp trawling. This means that the more profound responses to seasonal variation in physical variables such as temperature and salinity would influence abundance and distribution more than any impact of shrimp trawling.

**Juvenile Fish Abundance and Environmental Variables**

The abundance of the juveniles of the four study species was highest for *H. platessoides*, followed by *M. bilinearis*. Few specimens of *P. americanus* were caught in the study area overall, usually during the summer. Only one Atlantic cod, *G. morhua* (20.7 cm TL), was caught at Site 1 in the summer of 1998 and the reasons for its rare occurrence are not clear from the data. Consequently, only the abundance of juvenile *H. platessoides* and *M. bilinearis* were used in a canonical correlation analysis with selected environmental variables (Chapter 1).

A canonical correlation was done, this time correlating the abundance of *H. platessoides* and *M. bilinearis*. Of the environmental variables from the above list, starfish was replaced with % sediment organic content, and the ordination of these variables to identify composite variables was repeated with PCA. Again the first and second PCs ordered the variables into similar composites as that in Chapter 1, but there was no predictive value of the model that correlated the dependent variables with those...
PCs. A MANOVA to test for spatial (site) and temporal (seasonal) differences in abundance was also not significant.

These results were interpreted as a lack of a significant effect of the environmental variables on the abundance of juvenile *H. platessoides* and *M. bilinearis*. A possible explanation was that given the spatial scale (4 km X 4 km) of this study, most the variables did not differ very much over this spatial scale except a marked seasonal variation in temperature and salinity. It was concluded from this result that the spatial scale selected in the study was adequate in ensuring spatial comparability between the control and the impact site. Hence, such a spatial scale may be a suitable scale of choice if "field experimental" studies on juveniles of similar species of fish are conducted in inshore continental shelf habitats. Sullivan *et al.* (2000) did such a study on the New York Bight continental shelf to test the impact of scallop dredging on newly settled yellowtail flounder, *Limanda ferruginea*.

**Trophic Ecology Among Size Classes of Juvenile *H. platessoides***

The trophic ecology of juvenile fish with respect to shrimp trawling was another aspect proposed for study in this project. Based on the overall dominance of *H. platessoides* in the study area, it was the species selected to do this analysis.

The results of length frequency analysis and scale aging showed four age classes of *H. platessoides*, Age 0+, 1+, 2+ and 3+, were occurred in the study area. Age 3+ fish were caught only in the summer, or were also present in the winter. However, a change in their distributional behavior in winter may have resulted in none being caught in this season. The other size classes were present in the study area over both seasons suggesting that they are resident in the study area throughout the year. Hence, it was concluded that
the inshore continental shelf of the southwestern Gulf of Maine is a suitable nursery ground for *H. platessoides*.

Of all the results, the result from diet analysis was notable from the perspective of any ecological impact of shrimp trawling. All age classes of juvenile *H. platessoides* fed largely on prey taxa that were also abundant in the benthos. Despite an ontogenetic shift in the relative preferences of cyclopoid copepods and bivalves between young of the year (Age 0+) and older (≥ Age 1+) fish respectively, Ampharetidae (Polychaeta) and Amphipoda occurred in the diet of all juvenile age classes, varying only in the size and mean number taken per fish.

The Ampharetidae are tube-dwelling polychaetes. Members of this family and another tube-dweller, Maldanidae, construct vertical tubes that span 4-5 cm deep into the sediment, a depth range also confirmed from the vertical sediment cores to contain the highest levels of organic content. Considering the scouring depths of otter trawl doors to be between 4.5-10 cm (Brylinsky et al., 1994; Kaiser et al., 1996, Gordon et al., 1998; Pilskaln et al., 1998; DeAlteris et al., 1999), there is a potential adverse impact from shrimp trawling on the trophic ecology of *H. platessoides*. If the amount of prey biomass removed by scouring is substantial, feeding and growth are likely to be affected. Computations based the available data on: 1) the density of a predominant prey taxa such as Ampharetidae estimated in the grab samples; 2) their mean abundance in the diet; and 3) fish density, can provide some insight into how much potential food is lost due to trawling. However, the impact of trawling in this respect to juvenile *H. platessoides* was not directly testable in the study because no quantitative estimates of the level of scouring by trawl doors was done.
Furthermore, like the distribution of many marine organisms, the distribution of prey taxa is expected to be patchy. Also, the size of such patches may vary both spatially and temporally. Hence, data on: 1) patch size and distribution; 2) fish density at the patch during feeding; and 3) feeding periodicity in juvenile *H. platessoides*, is also necessary besides data on the intensity of scouring. Otherwise, it may not be very useful to interpret any impact of sediment scouring by trawl doors on the trophic ecology of juvenile *H. platessoides* even if the scouring intensity was known.

**Recommendations for Fisheries Management**

Having undertaken the study in the manner described, and providing a synthesis of the relevant issues and results above, a number of recommendations for application of any of the results from this study in fisheries management, or for consideration in designing similar impact studies will be given here.

**1. Ecological Level of Focus**

It was the objective of this study to investigate the impacts of shrimp trawling at the ecosystem and community level. It may sound ambitious but it is the approach that should be taken to put the data and interpretations into a clearer applied ecological perspective. Separation of ecological phenomena into levels should always be understood as artificial and that many ecological processes span different levels. What is perhaps important is that for any ecological process that is investigated, an effort must be made to focus on an appropriate scale of investigation. This will provide integrity to the data and also improve any ecological understanding based on it.
I have considered both the population and community level of organization in this study as an approach to obtain and analyze the data, facilitating interpretations within an ecosystem context. The use of diversity indices, as used in Chapter 1, is a common approach to study structure of communities (Magurran, 1988; Morin, 1999). It is a very convenient way to get a snapshot of the dynamics of the community as whole. Where impacts result in changes in the community structure due to shifts in the relative abundance of constituent species, diversity indices is a suitable method to detect these changes. If dominant species, as identified for the fish community in the southwestern Gulf of Maine, are involved such changes in the diversity indices can be dramatic.

Furthermore, this can help to identify indicator species in impact monitoring programs. The predominance of the sculpin, *Myoxocephalus octodecemspinosus*, in Site 1 in the study area seems to be associated with the occurrence of coarse sediments at this site. The occurrence of starfish and other epibenthic macrofauna (Table 1.3) seems to support this explanation. If mobile gear such as dredges that have a more dramatic impact on the sediments are used in such habitat patches, this species can be greatly affected and be a good indicator species for the intensity of impact from such gear in similar habitats.

The analysis of the population structure, distribution, abundance and trophic ecology of *H. platessoides* as done in this study is an example of using population-level data to infer potential ecological ramifications of the impact from trawl gear. This autecological approach helps to understand a big problem in a small way, with a potential to provide useful insights for decision making or planning more specific studies.

It is my recommendation that gear impact studies should collect as much autecological data as possible, and analyzing these multivariate data sets utilizing
powerful multivariate statistical techniques can enable effective ecological interpretations relevant at the ecosystem level to be made. This will provide a suitable framework for transferring the results of population-level studies to fisheries management and other similar applications.

2. A Need for Field Experimental Studies

The results from Chapters 1 and 2 indicate that the influence of environmental variables such as temperature, salinity and bottom type occur over large spatial and temporal scales. Therefore, they are relatively homogeneous over spatial scales within the range (4 km X 4 km) selected for this study. The magnitude of their influence on the distribution and abundance of juvenile fish is bound to be greater than any impact from shrimp trawling, and since fish are among marine taxa that are quite resilient to small-scale changes in habitat attributes, "field assessment studies" may not always detect an impact.

This demands that studies on the impact of fishing gear need to also consider opportunities for "field experimental" studies. Small-scale changes occurring over meters and having effects that last over days to months (Brylinsky et al., 1993; Langton et al., 1994; Gordon Jr. et al., 1998; DeAlteris et al., 1999) can be resolved more accurately relative to the impact by this approach. This has been done by a number of studies looking at benthic macroinvertebrates (Brylinsky et al., 1993; Thrush et al., 1995; Kaiser and Spencer, 1996; Gordon Jr. et al., 1998; Kaiser, 1998), though a recent study have looked at flatfish recruitment using this approach (Sullivan et al., 2000). Given the inherent statistical power of such studies with adequate replication of treatments, they
would provide more interpretable results because any "noise" in the data would be minimized.

An extensive body of literature on this subject exists that cover both statistical issues and issues related experimental design. It is beyond the scope of this chapter to discuss this subject in detail. For those interested in this subject, the work of Tony Underwood and his team out of the University of Sydney, Australia (Underwood, 1992, 1993, 1996; Glasby, 1997), a recent book by Schmitt and Osenberg (1996) and an earlier text by Green (1979) are good places to start from.

Studies focusing on benthic macroinvertebrates were often able to detect impacts from the experimental treatment (trawling, dredging or scouring) (Brylinsky et al., 1993; Thrush et al., 1995; Kaiser and Spencer, 1996; Gordon Jr. et al., 1998). Only one study has, from my knowledge of the literature, looked at fish using this experimental approach. This may be due to the fact that relatively mobile organisms such as fish may not be suitable. They may not be resident within the experimental plots throughout the study in such manipulative studies. However, restricting such studies to small-scale habitat patches within inshore coastal environments and estuaries may possible do.

3. Selection of Sensitive Variables in "Field Assessment" Studies

"Field assessment" studies are still the more common type of impact study. Such studies investigating the impact of fishing gear have often detected impacts on the more sedentary benthic macroinvertebrates (Bergman and Hup, 1992; Jones, 1992; Collie et al., 1997; Engel and Kvitek, 1998). Impacted areas are often dominated by species that are persistent and tolerant to such perturbations (Engel and Kvitek, 1998).
Given the greater likelihood of the influence of environmental factors on the abundance of juvenile fish over large spatial scales, abundance is not a suitable variable to use in such studies, especially if the impact is not catastrophic enough (e.g. an oil spill) to severely affect it. As discussed in the previous chapters, the distribution and abundance of juvenile *H. platessoides* was influenced more by variables such as temperature and bottom type, though a lack of correlation discussed in Chapter 2 is interpreted to be due to the small spatial scale of the study.

My suggestion would be to use a variable that integrates any ecological impacts on the species of interest over time. Changes in growth rate would be such a variable since impacts such as scouring can cause a decline in prey resources that could affect this variable directly through poor nutrition or indirectly through competition. Studying dimensions of otolith growth bands associated with major seasons of growth (winter and summer) with respect to age of fish could indicate differences related to the impact of interest (e.g. scouring by shrimp trawling), by matching the specific bands that coincide with the period of impact. Temperature effects on growth, salinity and nutrition are among factors known to affect otolith chemistry and depositional patterns (Radtke and Shafer, 1992; Tzeng *et al.*, 1997).

Juvenile *H. platessoides* are very abundant on the continental shelf over mud-sand habitats and specimens can be easily obtained in the field in such areas as shown in this study. Being a cold water species, the northeastern United States is at the southern boundary of their range (Bigelow and Schroeder, 1953; Scott and Scott, 1988). Studying treatment-related changes in size increment in such species using standard lengths and other metrics suitable for morphometric analysis can be used as well to establish growth
differences associated with impact. Confining such analysis to specific age classes, which are easy to establish from techniques such as length frequency analysis and scale aging (relatively easier to process large number of samples than using otoliths) as done in this study, can increase the reliability of such an approach.

4. Quantifying Trawling Intensity

Habitat destruction from trawling and other mobile gear is a major issue in fisheries management in the United States. It is the basis of the 1996 Magnuson-Stevens Fishery Conservation and Management Act (Benaka, 1999; Rosenberg et al., 2000).

"Field assessment" impact studies of trawling gear must also aim to obtain quantitative data on the level of gear impact, say the intensity of scouring by trawls, in order to obtain the necessary data to adequately address the ecological implications of this type of impact. Few studies have done this. DeAlteris et al. (1999) is one example that studied the physical impact of mobile fishing gear in Narragansett Bay, Rhode Island in relation to natural oceanographic processes. They calculated the intensity of gear impact in the study area to be less than 1 % using side-scan sonar detection of trawl tracks.

Such calculations are very useful in estimating the magnitude of any ecologically relevant impact. In this study it was shown that the important prey taxa of juvenile _H. platessoides_ are within the top 4-5 cm of the sediment, a depth range normally penetrated by trawl doors (Brylinsky _et al._, 1994; Kaiser _et al._, 1996, Gordon _et al._, 1998; Pilskaln _et al._, 1998; DeAlteris _et al._, 1999). If it had been possible to collect side-scan sonar data on trawling intensity in this study, the amount of sediment displaced, as well as the constituent prey resources lost would have been possible to calculate with the data
available. A better perspective of the impact of scouring from shrimp trawling on different age classes of *H. platesoides* would then have been possible.

Auster and Langton (1999) have published a map of the distribution of trawl and scallop dredge tows on the Georges Bank and Gulf of Maine from 14,908 tows using the NOAA sea sampling database from 1989-April 1994. Conducting scouring intensity and prey loss studies within 'hot spots' depicted by such maps, in relation to size and distribution of prey patch sizes of species such as tube-dwelling polychaetes as described above, would be very useful to do.

In conclusion, the above recommendations mean that there is a lot yet to be done before habitat management and ecosystem approaches become an effective fisheries management tool, especially if predictive capabilities can be developed (Auster, 1998; Auster and Langton, 1999; Langton and Auster, 1999) using the results of carefully-designed localized studies.
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Table A1. Fish catch data per trawl for each sampling date at each site.

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TOTALS           |             |      | 27    | 242   | 84    | 128   | 61    | 238   | 1     | 1     | 0     | 0     | 0     | 0     |

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Table A2. Summary of abundance of dominant benthic macroinvertebrate taxa sampled with the Shipek grab sampler at the two sites on each sampling date. The total volume (cm$^3$) of sediment processed measured for each sampling date is the sum of volumes for the respective number of grabs (n) sampled. The total area (cm$^2$) is the calculated area represented by the volume of each grab and summed for all grabs from each site by sampling date. Abundance was scaled to density per square meter for each grab, and the averages based on all grabs for each sampling date calculated in this manner are tabulated here.

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