The Decline of Fisheries Resources in New England

Evaluating the Impact of Overfishing, Contamination, and Habitat Degradation

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Published by the MIT Sea Grant College Program
292 Main Street, E38-300
Cambridge, Massachusetts 02139
http://web.mit.edu/seagrant

Acknowledgment: Sponsor, Massachusetts Bays National Estuary Program
Publication of this volume is supported by the National Oceanic and Atmospheric Association
contract no.: NA86RGO074.

Design and production by Gayle Sherman

Cover photo credits:
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MIT Sea Grant College Program Publication No. 05-5

We dedicate this book to our colleague, John Moring,
who was a tireless advocate for the health of our
fisheries in the marine environment.

Acknowledgments

We appreciate the encouragement and the financial support provided by the Massachusetts Bays Program (MBP), part of the National Estuary Program of the United States Environmental Protection Agency. This work was sponsored by a MBP grant, and the original concept came from their Technical Advisory Committee. In particular, we thank Jan P. Smith, Director of the MBP, for helping us to develop the vision for this project and for supporting us as the project unfolded. We also sincerely thank the many reviewers who volunteered their time and energy to critique each of the chapters of the book at various stages and for their valuable suggestions and advice. These include Peter Auster, Mark Chandler, Michael Dadswell, Ellie Dorsey, Michael Fogarty, David Franz, Madeleine Hall-Arber, Gareth Harding, Boyd Kynard, Sandra Macfarlane, Jack Pearce, Jack Schwartz, Jan Smith, Peter Wells, Christine Werme, Robert Whitlatch, and several anonymous reviewers. We especially thank our authors for each of their contributions, and, in addition, for their patience as we moved the process of publishing this book along.

Finally, the publication of this work at MIT Sea Grant would not have been possible without the dedication and persistence of Gayle Sherman, who transformed all the raw manuscripts into a professional publication.

Contents

Preface

First Preface (2001)

This book emerged from one of several issue papers sponsored by the Technical Advisory Committee of the Massachusetts Bays Program, one of the National Estuary Programs funded by the Environmental Protection Agency. When we started this project, the crisis in New England groundfish populations was not nearly as much in the public consciousness as it is now. There was a fair amount of conjecture within the fishing industry, due in part to uncertainty that exists in stock assessment, concern about pollution and habitat impacts, and anecdotal descriptions of improved fish catch that seemed at odds with agency predictions as to the root cause of the problem. We thought it useful to put together a volume that examined the scientific evidence for effects of three major factors overfishing, pollution, and habitat degradation on northeast finfish, lobsters, and shellfish populations. This approach omits the socio-economic drivers which influence fisheries management decisions and the industry's perception of the ecosystem. We acknowledge the importance of these factors in the broader scope of the issue, however without a sound scientific knowledge, achieving sustainability in fisheries will falter.

During the intervening years since we began the project, the public became increasingly aware of the seriousness of the crisis through numerous newspaper articles, scientific publications, public forums, and political activities. The closure of large parts of Georges Bank to groundfishing, and the recent passage by the New England Fishery Management Council of measures to further reduce fishing mortality focused much attention on the effect that overfishing may have on marine ecosystems and the fishing communities that depend on those ecosystems. The cause of fish population declines, the rate and extent of recovery, and approach to management are still debated. The usefulness of examining the declines in certain species of commercially important fish, lobsters, and shellfish by taking multiple factors into consideration is compelling because it represents a holistic approach to managing fisheries and addresses management needs into the future.

Each of the authors was asked to review the latest scientific information on either overfishing, pollution, habitat degradation, or the relationship among all three variables, on fish, lobsters, or shellfish, particularly as it relates to the Massachusetts Bays region. Authors were encouraged to think broadly and to give some sense of the relative impacts on fisheries of the factors they examined in detail compared to those attributable to other causes. They were also asked to identify data gaps and to suggest needed research. Management implications were then addressed in the final chapter.

Robert Buchsbaum
Judith Pederson
William E. Robinson
January 2001

Second Preface (2004)

Since we first assembled the draft contributions of our authors into this volume and penned the above Preface in early 2001, high profile lawsuits have changed implementation of fisheries management. Significant changes in fisheries stocks and fisheries management have influenced our overall approach to using and managing our marine environment. Some trends in fish population recovery appear favorable, whereas others are not quite so optimistic.

While not universal, a number of improvements have been documented in various fishery stocks. Probably the most notable change is in the northeast groundfishery, where several species populations have increased as a result of the drastic measures implemented to reduce fishing pressure (e.g. closure areas, severe restrictions in the number of days a fisherman can fish). The National Oceanic and Atmospheric Administration's (NOAA's) annual reports to Congress on the

"Status of the Fisheries of the U.S." have indicated an overall improvement in groundfish stocks, as evidenced by the stock assessment data collected in 2000, 2001, 2002, and 2003. When we started this work, over 70% of New England groundfish were classified by the National Marine Fisheries Service (NMFS) as overfished (NOAA, 1995). Status of the Fisheries Resources off the Northeastern United States for 1994. NOAA Tech. Memorandum NMFS-NE-108. National Marine Fisheries Service, Woods Hole, MA. 140 pp.). The percentage now is about 30% (NMFS, 2001. Report to Congress. Status of Fisheries of the United States. NMFS, NOAA, Silver Spring, MD. 127 p.). An exceptionally good cod spawn in the Gulf of Maine in 1998, combined with restricted fishing effort, has led to the recent surge in the numbers of legal size fish that have been caught (although cod quotas have sometimes prevented the landing and sale of these fish). Nevertheless, groundfish stocks, while increasing as a result of less overfishing, are still nowhere near their historic high numbers and some would argue they are not sustainable. In NOAA's most recent status report (16 June 2004), the Northeast groundfishery was still considered the national fishery in the most trouble. Twelve stocks were still listed as "overfished" (i.e. overall biomass below a set level), and eight stocks were still subjected to "overfishing" (i.e. too many fish taken).

A remarkable recovery has occurred in North Atlantic Swordfish stocks. While the fishery is still at approximately 65% of maximum sustainable yield, stocks have rebounded in an incredibly short time, due primarily to the restrictive international quotas and minimum sizes imposed by the International Commission for the Conservation of Atlantic Tunas in 1997, the imposition of Individual Transferable Quotas (ITQs) by Spain in 1999, and the closure of critical nursery areas to longline fishing by the U.S. in 2000 (NOAA, 2002. http://www.publicaffairs.noaa.gov/releases2002/oct02/noaa02131.html; Garza-Gil et al., 2003. Mar. Policy 27:31-37).

Debate continues over Maine's native salmon populations, even while their numbers remain perilously low. In June 2004, the draft Federal plan was released by the Atlantic Salmon Commission to save the endangered Atlantic salmon in eight Maine rivers. But the loss of habitat, uncertainty in the primary cause of mortality, and small wild populations may limit a successful comeback to a thriving fishery.

The story with lobsters is mixed. Lobster populations have drastically fallen in southern New England, with a concern that the same trend may be about to happen throughout the region. An article in the December 2004 Commercial Fisheries News reported that Maine's lobster landings have remained "relatively high" (yet down from 2002), although landings have fallen precipitously in other states. A recent Cape Cod Times editorial (28 June 2003) reported that 91% of the lobsters taken from Cape Cod Bay north to the New Hampshire border were barely of legal size, a trend that highlights growth overfishing and does not bode well for sustaining this fishery. Nevertheless, questions continue to be raised as to whether the lobster decline is due to pollution, habitat degradation, or disease, rather than simply to overfishing. The declines in the southern part of New England are further complicated by two different types of diseases, one of which may be related to rising temperatures. Some attribute the lack of collapse in Maine waters to community management of the fishery, but recognize that increases in landings are due to other factors (Deitz, 2003. Science 302:1907-1912).

Fluctuations and trends in bivalve stocks continue to be difficult to assess. Recent advances in high resolution video surveying of scallop populations (Stokesbury, 2002. Trans. Am. Fish. Soc. 131: 1081-1092) have led to a proposal for rolling openings of different areas of offshore scallop grounds. It remains to be seen whether the rolling opening approach will lead to a sustainable fishery as fishermen put pressure on management to open any productive area left unfished. Outside influences may further complicate analysis of success. The recent presence of an aggressive sea squirt, *Didemnum* sp. that covers 80% of approximately 75 km^2 of prime scallop beds on Georges Bank adds another dimension to habitat degradation and is one for which we have little experience or knowledge.

Shellfish are also at risk. An outbreak of disease (i.e. a parasitic infection Quahog Parasite Unknown or QPX) in southern New England quahogs has recently decimated quahog populations in Wellfleet Massachusetts, as it did in Provincetown Massachusetts in 1995. It is not known whether pollution, habitat degradation or even climate change has led to physiological stress that weakens animals enough to allow parasitic infection to take hold.

Fisheries management debates in the northeast have remained as contentious as ever, and possibly more so! The passage of the federal Sustainable Fisheries Act (SFA) in 1996, with its mandated call for revised definitions of what “overfished” means and its attention to essential fish habitat, led to a great deal of debate (and little initial action) as to how to incorporate these mandates into the region’s groundfisheries management plans. The perceived slow federal action led the Conservation Law Foundation, National Audubon Society and the National Defense Council to file a lawsuit in May 2000 against the Commerce Department and its agencies, NOAA and the National Marine Fisheries Service, for not doing enough to prevent overfishing of Northeast groundfish stocks or to reduce bycatch mortality when it approved Framework Adjustment 33 to the Northeast Multispecies Fishery Management Plan. The suit alleged that the Adjustment did not comply with the SFA because it based its recommendations on the Amendment 7 building plan rather than the more stringent Amendment 9 plan, and that even Amendment 9 failed to reduce bycatch. On 28 December 2001, District Court Judge Gladys Kessler ruled in favor of the three conservation groups, and asked both the plaintiffs and defendants to propose remedies for her consideration. While each side opposed the other’s plan, the two groups forged a negotiated settlement and sent it to the court on 22 April 2002. Four days later, Judge Kessler rejected the compromise plan, and instead handed down a stunning “remedial order” that included area closures and drastic reductions in allowable days at sea. What followed was a tremendous outcry by fishermen, their political representatives and several of the parties to the case, all of whom asked the Judge to reconsider her decision. On 23 May 2002, Judge Kessler granted the motions to reconsider, vacated her 26 April order, and replaced it with the compromise settlement that had been worked out by the parties in the case. This action did not end the courts involvement in Northeast groundfisheries management, however. Just after Amendment 13 to the Northeast Multispecies Fishery Management Plan went into effect on 1 May 2004, suits were filed in U.S. District Court by both fisherman and conservation groups. The Trawlers Survival Fund, a fisherman’s group from Fairhaven, alleged that NMFS made illegal changes to the final rule implementing Amendment 13 that would be detrimental to fishermen. On the other side, the Conservation Law Foundation and the National Resources Defense Fund filed briefs alleging that Amendment 13 will not stop overfishing, and Oceana filed two lawsuits alleging that Amendment 13 ignores essential fish habitat.

All of these contentious legal battles and drastic reductions in both the areas where fishermen are allowed to fish and in the number of days that they are allowed to fish occurred against the backdrop of the recovering groundfish stock. Throughout this time period, fishermen repeatedly questioned why additional harsh measures have to be implemented when the overall trends are improving. They argue that mandating a particular stock recovery in a five year time span is both unrealistic and economically disastrous to the fishing industry. To make matters even worse, in September 2002 NMFS acknowledged that otter trawl lines on the R/V Albatross IV that had inadvertently been mismatched in length since February 2000 *may* have led to mistakes in assessing groundfish populations off of New England. While they later provided evidence to show that this mismatch did not lead to any significant changes in the stock assessments, fishermen nevertheless maintained that the stock numbers were underestimated and therefore unreliable for use in management decisions. The perspective of the New England fishermen and management is in contrast to the Northwest fisheries where scientific advice and stringent measures are more readily accepted. What seems clear today, is that overfishing has been reduced, yet a number of stocks are still considered overfished. In addition, stock abundances, while increasing, have yet to reach their historically high numbers. There are three issues that are hurdles for broad support of stringent management options. First, uncertainty is a strong component of stock assessment, which confounds projections of stock biomass. Secondly, the past approach to managing single stocks has failed to sustain some stocks, but currently there are no acceptable models for managers to implement. NOAA and other agencies are focusing on ecosystem management but this science is in its infancy and without higher levels of certainty, fisheries management approaches will not be readily accepted by the industry.

The original premises that we based this book on – that overfishing, habitat degradation and contamination each contribute to the health of our fish and shellfish populations (albeit to different degrees in each fisheries stock); that each of these three impacts has been studied independently by disciplinary scientists, in isolation from each other; that we need to consider each of these three impacts together, in an interdisciplinary holistic approach, in order to understand the total stress on commercially important fish stocks; that we need to place more reliance on a precautionary approach, adaptive management efforts and ecosystem-based management in order to manage our fisheries populations in a sustainable manner – have been reemphasized many times over the intervening years. Many of the specific points that we made in 2001 have independently been raised since then:

- In a recent paper by Giulio Pontecorvo (2003. Marine Policy 27: 69-73), the "insularity of scientific disciplines" was identified as a significant impediment to fishery management.

- There is an increasing awareness that current fisheries practices *worldwide* are not sustainable (e.g. Myers and Worm, 2003. Nature 423: 280-283; Pauly et al., 2002. Nature 418: 689-695). Approximately 30% of worldwide fish stocks are depleted, overfished or slowly recovering and 44% are currently being fished at or near their sustainable yields (National Research Council. 1999. Sustaining Marine Fisheries. National Academy Press, Washington D.C.).

- Ecosystem-based management (EBM) and the use of Adaptive Management have been endorsed by the United Nations Food and Agriculture Organization (FAO), the European Union, and the National Research Council (NRC). The National Marine Fisheries Service (NMFS)'s 5-year Strategic Plan for Fisheries Research (December 2001) placed ecosystem considerations as a priority in its "new generation" stock assessments.

- The Ocean Studies Board of the National Research Council of the National Academy of Sciences released their report Effects of Trawling and Dredging on Seafloor Habitat in May 2002, which concluded that negative effects seafloor habitat were happening in some areas, and that sufficient data were available to at least conduct preliminary assessments of trawling/dredging in other areas.

- Some scientists have advocated against basing fisheries management on Maximum Sustainable Yield (MSY). Richard Zabel and colleagues, for example, have suggested that we now address what he has termed "Ecologically Sustainable Yield" (2003. Am. Scient. 91: 150-157). This concept recognizes that single species cannot adequately be managed in isolation, but must be managed as an ecosystem.

- The importance of climate change on long-term fisheries trends has now been recognized. Range shifts of New England marine fish in response to ongoing warmer seawater temperatures had already been documented (Murawski, 1993. Trans Amer. Fisheries Soc. 122: 657-658), and this trend will continue, perhaps in a more accelerated rate, in the future for species like cod (Scavia et al., 2002. Estuaries 25: 149-164). The failure of the Canadian cod stocks to rebound after their collapse in the late 1990s, even though fishing pressure, has been eliminated, may be of a changing climate. Scavia et al. also predict changes in phytoplankton-zooplankton dynamics that could alter the food sources for fish. Based on mesocosm experiments and examination of short term temperature variations, the US Global Climate Change Research Program predicts that global warming will be detrimental to populations of winter flounder in southern New England (New England Regional Assessment Group, 2001. Preparing for a

Changing Climate: the Potential Consequences of Climate Variability and Change. New England Regional Overview. U.S. Global Change Research Program, Univ. of New Hampshire, 96 pp.) Although its impact has yet to be specifically documented, the spread of lobster shell disease and the quahog parasitic infection QPX northward may be linked to global climate change.

- In January 2005, EPA will release its National Coastal Conditions Report, in which it designates the Northeast as "one of nations dirtiest regions."

- Finally, both of the two major, recent and long-awaited, reports on the state of our oceans and marine environment, the Pew Ocean Commissions America's Living Oceans. Charting a Course of Sea Change (May 2003) and the U.S. Commission on Ocean Policy's An Ocean Blueprint for the 21st Century (Sept 2004) highlighted overfishing, contamination and habitat alterations (direct impacts such as by trawling and loss of nursery areas, and indirect impacts due to eutrophication, invasive species, to name a few) as all being important contributors to our marine resource declines, and called for an ecosystem-based management approach. Both reports can be summed up in the words of the Chairman of the U.S. Commission on Ocean Policy:

> *The oceans and the coasts are in trouble, and we need to change the way we manage them.*
>
> *- James D. Watkins, 2004*

We are pleased that our original premises for this book have now been more broadly accepted by the scientific community. When we initiated this project, many voiced skepticism that overfishing, habitat degradation and contamination could each have a role to play in the health of our many fishery stocks, and, even if they did, whether we could make useful comparisons of the impact of overfishing, habitat degradation and contamination on fisheries stocks. We believe that the information that our authors have summarized and reviewed clearly demonstrates that each of these three impacts can be significant, although the data are not yet available in most cases to estimate the degree to which each of the three operates.

We thank the Massachusetts Bays Program for providing the initial funding for this project. We also thank our authors for sharing our vision, and for their patience as we worked to bring this book to publication. We also thank the many reviewers who read the chapters and whose thoughtful insights strengthened this work. (They are acknowledged in each chapter.) Given the perceived restricted audience that this volume would likely attract, publishers proved illusive. We sincerely thank MIT Sea Grant for taking up our cause and publishing this work. We are all the more grateful in that MIT Sea Grant agreed to publishing it "on the web", making its distribution and hopefully its impact much more widespread and easily accessible. Finally, we would like to thank you, our readers, who will have the ultimate vote on whether our work proves useful in advancing the ongoing debate on fisheries management.

Robert Buchsbaum
Judith Pederson
William E. Robinson
December 2004

Chapter I

Contamination, Habitat Degradation, Overfishing - An “Either-Or” Debate?

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> *Fish are good for the heart, but such knowledge will soon be of little use if we cut the heart out of the ocean.*
> *-Derrick Z. Jackson, Columnist*
> *Boston Globe, 14 Jan 1998*

Introduction

A great number of nearshore and offshore fishery stocks have deteriorated throughout the Northeast over the past 30 years. The most visible example of this decline was the precipitous drop in populations of groundfish (benthic-feeding fish such as Atlantic cod (*Gadus mohrua*), yellowtail flounder (*Pleuronectes ferrugineus*) and haddock (*Melanogrammus aeglefinus*)) (NOAA, 1998; NEFSC, 2000). These stocks were severely overfished by foreign fishing fleets in the 1960s and early 1970s, and then partially recovered in the mid-1970s, coincidentally with the implementation of the Magnuson Fishery Conservation and Management Act (Magnuson Act) of 1976. The overall decline of most of these groundfish species eventually resumed, and has continued up to the present time (Figure 1.1; NOAA, 1998; NEFSC, 2000). With recent stringent management measures, some stocks appear to be showing signs of recovery, but they are still nowhere near their former levels of abundance (NEFSC, 2000). This decline notwithstanding, groundfishing has remained an important contributor to the overall northwest Atlantic (Northeast) fishery (Figure 1.2), accounting for an average of 121,000 metric tons of landings and approximately $179,000,000 in ex-vessel value for the years 1993-97 (NOAA, 1998). When considering finfish landings alone for this period of time, the groundfishery provided approximately 24% of the total finfish catch in the Northeast, yet accounted for 55% of the ex-vessel value for the total landed finfish (NOAA 1998).

The recent crisis in the groundfishery, which has been steadily unfolding since the enactment of the Magnuson Act in 1976, was highlighted in Massachusetts with the publication of the Assessment at Mid-Decade (MA DMF, 1985). Warnings have been issued repeatedly ever since (OGTF, 1990; NEFMC, 1991, 1994, 1996, 1999, 2000; Doeringer and Terkla, 1995; Murawski et al., 2000). Stocks had plummeted to such a point by 1991 that the Conservation Law Foundation and the Massachusetts Audubon Society filed a lawsuit against the New England Fishery Management

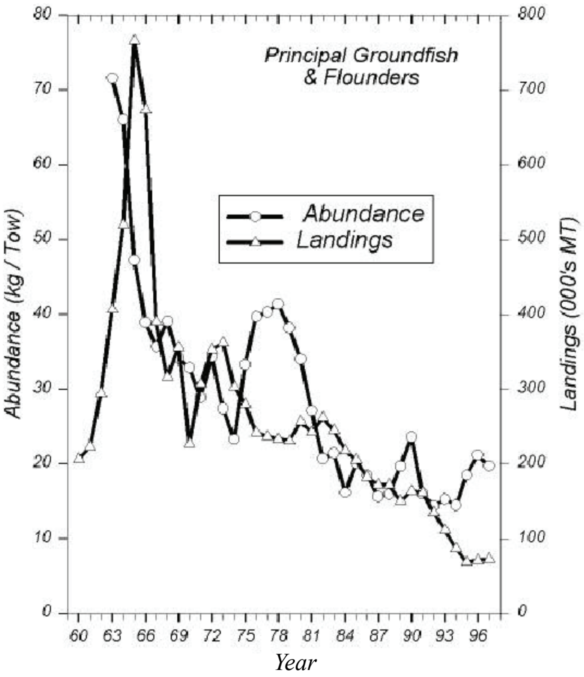

Figure 1.1. Trends in abundance and commercial landings (U.S. and foreign fleets) of principal groundfish from the northeastern United States. (Figure from NOAA, 1998).

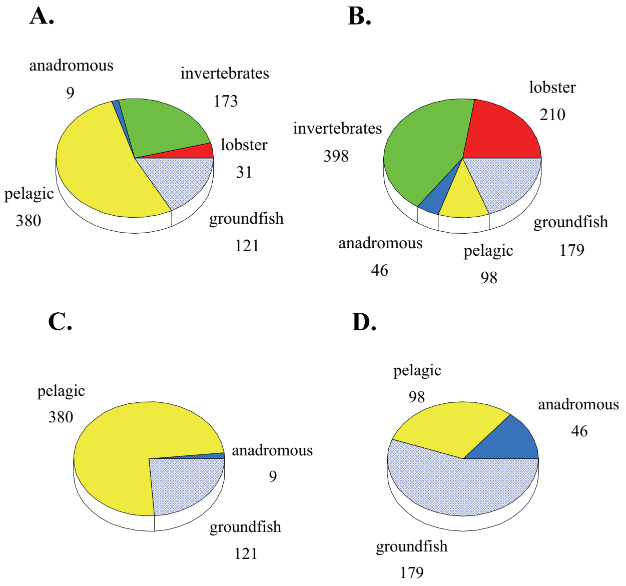

Figure 1.2. Mean Northeast fishery landings (thousand metric tons; A, C) and ex-vessel value (millions of dollars; B, D) for the years 1993 - 97. A and B include finfish, shellfish and other invertebrates; C and D include only finfish. All data from NOAA (1998).

Council (NEFMC) for failure to prevent overfishing. This resulted in the promulgation of Amendment V to the Northeast Multispecies Fishery Management Plan and an emergency closure of parts of Georges Bank in 1994 (NEFMC, 1994). Since stocks did not rebound as anticipated, additional amendments and framework adjustments were issued, closing large areas to fishing and severely curtailing the number of allowable days-at-sea (NEFMC, 1996; 2000). The reauthorization of the Magnuson Act, the Sustainable Fisheries Act (SFA) of 1996, imposed new requirements, including: (1) regular reporting of the status of individual fish stocks, (2) revised overfishing definitions, and (3) recovery plans for overfished stocks that included delineating and conserving essential fish habitat. These restrictive measures are having an inordinate impact on the economic and social well-being of our fishing communities.

While groundfisheries have received the most public attention, other commercially important fish and shellfish species have also dwindled in numbers. Bluefish (*Pomatomus salatrix*) stock biomass has shown a downturn over the past nineteen years following a peak in total east coast landings in 1981 (NOAA, 1998). Stock biomass fell far below densities needed to maintain maximum sustainable yield (MSY) (e.g., only 21% of MSY in 1997; NOAA, 1998). The 1999 Atlantic bluefin tuna (*Thunnus thynnus*) spawning stock biomass was approximately 16% of the 1975 level, and only 31% of the level needed to achieve MSY (NMFS, 1999). There are indications that soft-shell clam (*Mya arenaria*) and quahog (*Mercenaria mercenaria*) populations have declined in some locales (Alber, 1987; Matthiessen, 1992; MacKenzie and McLaughlin, 2000) and several anadromous fish (alewife, *Alosa pseudoharengus*; blueback herring *Alosa aestivalis*; Atlantic salmon, *Salmo salar*; and American shad, *Alosa sapidissima*) are exhibiting all-time low population numbers (NOAA, 1998). The situation became so serious for the wild stock of Atlantic salmon that the Gulf of Maine population was proposed for protection by the Endangered Species Act (Fed. Reg., 1999). Finally, there is fear that the lobster (*Homarus americanus*) spawning biomass may also be declining even though lobsters have been harvested in record numbers for almost a decade (NOAA, 1998; McBride and Hoopes, 2000). Fisheries managers are concerned that if environmental conditions become unfavorable, the lobster population will not be able to sustain current catch efforts.

Not all species of fish and shellfish are exhibiting an unequivocal decline in numbers. Sea scallops (*Placopecten magellanicus*), while considered by National Marine Fisheries Service to be overexploited, exhibit "boom and bust" years, dictated by inconsistent and unpredictable cycles of recruitment and fishing pressure (NOAA, 1998; Taylor, 1998). Striped bass (*Morone saxatilis*) suffered from overfishing and poor recruitment, but have rebounded following Northeast cooperative fishing bans initiated in 1982 (NOAA, 1998). Species, such as mackerel (*Scomber scombrus*) and Atlantic herring (*Clupea harengus*), that are not sought after intensively by the fishing industry, have increased in numbers over the past decade (NOAA, 1998).

The Northeast fisheries are not unique. While the unrestricted exploitation of Georges Bank fish stocks by foreign fishing vessels in the 1960s and 1970s resulted in the passage of the 1976 Magnuson Act, U.S. vessels quickly entered the fishery to resume the same trend of overexploitation. This regional decline in fish populations is similar to what is occurring in many other regions of the United States and throughout the world (FAO, 1997; Pauly et al., 1998; Christen, 1999). The reductions, however, are particularly evident and well-documented, here in the Northeast.

PUBLIC PERCEPTIONS

Appearances can be deceiving.
- (Anon. proverb)

What is the major anthropogenic cause of the stock declines in the Northeast fisheries—overfishing, introduction of contaminants, or habitat degradation? The public is regularly barraged with alarming news reports on the collapse of our fishing industry, coastal and marine habitat degradation, bacterial contamination of our inshore shellfish species, and toxic chemicals in the marine environment. Can declines in fish stocks be attributed largely to one of these factors or is it their interactions that are of the greatest consequence? In this work each of the human-induced impacts—overfishing, contamination, and habitat alteration—is examined and discussed as to their relative importance to selected fish and shellfish stocks.

OVERFISHING

The cod has symbolized fisheries in Massachusetts and New England since the time of the Basque fishermen who started bringing New England salted cod back to Europe as early as the 1400s. Initial reports of explorers and European colonists describe an unlimited abundance of cod and other fish in New England waters. Impressive catches were recorded throughout the 19th century:

> *The bankers, particularly if the fishing was good, would have to row back to the schooner with a doryful of up to 1800 pounds five or six times to finish with a single trawl, hollering "Dory!" as they bumped up alongside, bringing skipper and cook running to the rail.*
>
> *- Joseph E. Garland, 1983*

The first indication to the public that something was seriously awry with New England fisheries was probably in the 1980s when fish prices rose and newspapers began to report the plight of fishermen no longer able to make a living catching fish.

What has put us in the situation where two-thirds of the Northeast commercial fish stocks are now designated as overexploited and 59% of these stocks are categorized as having "low abudance" (Table 1.1; NOAA, 1998; NEFSC, 2000)? Over the past 100 years, the fishing industry has undergone major changes—from fishing in wooden boats under sail, to steam engines, to diesel-powered steel-hull trawlers, and even to factory ships that process seafood at sea. Improved methods of locating and catching large concentrations of fish have increased the efficiency of fish harvesting (although catch-per-unit-effort has now decreased due to the declining biomass of targeted species). For years, fisheries scientists carrying out stock assessments have warned that unregulated fishing would eventually lead to stock declines:

> *Excessive fishing has led to significantly reduced resource abundance, smaller and less fish and shellfish being landed in our ports, and economic hardship for the state's fishing industry.*
>
> *- MA DMF, 1985*

There is a general consensus among fisheries managers that the effects caused by overfishing have far outweighed the adverse impacts caused by contaminants and habitat loss, at least with respect to recent groundfish declines (Werme and Breteler, 1983; Cohen and Langton, 1992; Serchuk et al., 1994; Myers et al., 1995). Simply stated, their position is that overfishing has reduced our stocks of groundfish to levels that cannot support sustainable yields at current landings. A number of fishermen agree that overfishing is a cause of groundfish stock depletion, but also cite habitat degradation, pollution and natural weather events as important factors (Dorsey and Pederson, 1998; Pederson and Hall-Arber, 1999). Sea scallops and lobsters are also listed by NMFS as overexploited, but nearshore shellfish and anadromous fish are not as easily characterized. As discussed more fully in the following chapters, the real or perceived importance of overfishing depends to a large extent on the species, its various subpopulations (if any), the subregion in question, and the availability of data.

Table 1.1. Stock abundance and level of exploitation for 51 Northeast finfish and invertebrate fisheries. Percentages refer to the total number of stocks in each category. Data taken from NOAA, 1998. SNE = Southern New England; GB = Georges Bank; LIS = Long Island Sound; GM = Gulf of Maine; S. = Southern; N. = Northern.

Level of Abundance	**Underexploited** (10% of all stocks)	**Fully Exploited** (24% of all stocks)	**Overexploited** (66% of all stocks)
High (10%)	Atlantic herring Atlantic mackerel (4%)	Striped bass (2%)	Am lobster - GM Spiny dogfish (4%)
Medium (31%)	Atlantic surfclam Butterfish N. Red hake (6%)	Longfin inshore squid Ocean quahog Northern shortfin squid Skates N. Windowpane flounder (10%)	Summer flounder Am. lobster - GB+S, SNE-LIS N. Silver hake Yellowtail flounder-Cape Cod Am. Plaice Winter flounder - GB Northern shrimp (15%)
Low (59%)	(0%)	Haddock-GB Yellowtail flounder-SNE Redfish Pollock American Shad Yellowtail flounder - GB (12%)	Scup, Black Sea Bass, Sea scallop Cod-GB, GM Witch, Cusk, Tilefish, Wolffish, Goosefish, Bluefish White hake, Ocean pout Yellowtail flounder - Mid-Atlantic River herring Haddock - GM S. Silver hake, S. Red hake Atl. Sturgeon, Shortnose Sturgeon S. Windowpane, Atl. Salmon Winter flounder - SNE-MA, GM (47%)

CONTAMINATION

Another factor that is often identified as a culprit responsible for declining fish populations is the impact of chemical contaminants—those chemicals that enter the environment from land-based, human activities. Are these chemicals having an impact on fish populations? The answer is far from simple.

Numerous studies have documented the presence of a wide range of chemical toxicants (Dow and Braasch, 1996; Jones et al., 1997; NOAA/NS&T, 2000), including those listed as "priority pollutants" by the U.S. Environmental Protection Agency (EPA, 1991), in waters, sediments and the tissues of our fishery resources. Nevertheless, few of these priority pollutants, and almost none of the approximately 65,000 to 75,000 substances in commercial use today, have been routinely monitored in commercial species. Based on studies of contaminants in marine sediments and marine mussels (Buchholtz ten Brink et al., 1996; Jones et al., 1997), a number of urban harbors in the Northeast (Boston, Salem and New Bedford Harbors in Massachusetts; Hudson-Raritan Bay in New York/New Jersey; western Long Island Sound in Connecticut; Portsmouth NH Naval Shipyard) have been identified as contaminant "hot spots". In general, organisms more distant from these hot spots (e.g. offshore organisms) contain lower levels of contaminants than animals more immediately exposed (McDowell, 1996).

Our concern over chemical contaminants often focuses on human health. Reports of elevated concentrations of contaminants in commercial fish or shellfish almost always raise questions of human health impacts. For example, Murchelano and Wolke (1985) described tumors in winter flounder (*Pseudopleuronectes americanus*) collected in 1984-85 from areas in Boston Harbor MA, including an area near a sewage outfall on Deer Island Flats. While raising awareness of contaminant effects on individual organisms, their study invoked concerns by the news media of a possible link to human cancer. Similarly, the U.S. Environmental Protection Agency's Quincy Bay, MA study (EPA, 1988; Cooper et al., 1991) brought attention to high levels of polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) in lobsters, soft-shell clams, and winter flounder. This raised fears of human neurological and carcinogenic impacts, and resulted in the issuance of a public health advisory that is still in effect today (MA DPH 1988). In addition, the widespread distribution of mercury in the northeast from mid-western coal-fired power plants and incinerators has resulted in the bioaccumulation of methyl-mercury in the aquatic food webs (Pilgrim et al., 2000). Human health advisories have been issued, owing to mercury's neurotoxic effects, particularly in developing children. While most of this concern has focused on mercury bioaccumulation in freshwater fish caught by recreational anglers, a variety of estuarine fish and shellfish that are commonly used for human consumption have also been found to contain elevated methylmercury levels (Burger et al., 1998; Kawaguchi et al., 1999; Gilmour and Riedel, 2000).

Although the most widely publicized contamination concerns address human health issues, a number of studies have directly examined the effects of contaminants on the health of marine organisms. While the linkage between contaminants and the health of marine populations is difficult to establish conclusively, pollution has been implicated as a factor in several fishery declines (Goodyear, 1985; Barnthouse et al., 1990) and even in large-scale ecological perturbations (Sarokin and Schulkin 1992). Many more studies that integrate chemical concentrations in water, sediment and tissues with adverse physiological effects, and that ultimately document the effects of contaminants on populations, are clearly needed. As is evident in the chapters reviewing effects of contaminants on finfish and shellfish, the investigation of contaminant impacts on populations is one of the most pressing needs for future research, particularly for stressed populations.

HABITAT DEGRADATION

Degradation of marine and estuarine habitats is also perceived as harmful to marine resources. This degradation may take many forms, over and above the effects caused by human related chemical contamination. Habitat degradation may be caused by physical changes, such as increased suspended sediment loading, overshadowing from new piers and wharves, filling coastal wetlands, and trawling and dragging for fish and shellfish. Although often ignored as a habitat alteration, increased nutrients from wastewater, fertilizers, and atmospheric inputs also degrade habitats, especially those nearshore. The resulting accelerated eutrophication may cause unwanted algal blooms, low dissolved oxygen and altered community composition. Natural environmental phenomena (e.g. weather, climate) must also be considered in conjunction with these habitat changes (Werme and Breteler, 1983; Serchuk et al., 1994; Hofmann and Powell, 1998).

Regardless of whether a habitat alteration is due to natural or anthropogenic causes, it can have a long-term impact on community structure. For example, the extensive die-off of eelgrass beds (*Zostera marina*) caused by a disease which struck the East Coast during the 1930s markedly changed the vacated habitat and the structure of the benthic food chain, and lead to sharp declines in bay scallop populations (Orth and Moore, 1983; Short et al., 1986; Buchsbaum, 1992). Some of these changes are still evident today. Otter trawling and scallop dredging also appear to be causing dramatic shifts in benthic community structure due to physical disruption of the bottom, reduction in habitat complexity and direct interference with trophic transfer (Langton et al., 1996; Langton and Auster, 1999). At the far extreme, habitat may be completely lost as a result of coastal development, harbor dredging and offshore mining operations. Nearshore finfish and lobster "nursery grounds" are particularly susceptible to all these types of habitat loss.

Understanding these effects and separating physical changes from overfishing is a challenge for scientists, managers and the fishing industry. Some changes, such as building dams, armoring of the shoreline and coastal development have impacted shellfish and anadromous fish, but these impacts are not well documented. The challenge for managers in regard to habitat is similar to that for chemical contamination—how localized are the impacts and how have they effected populations? In the chapter on habitats, the authors review studies that relate habitat alteration, particularly those related to human activities, to effects on populations.

THE "EITHER-OR" DEBATE

> *The first step in science is the step from qualitative impression to quantitative measurement. The occasional difficulty of this task does not lessen its importance.*
>
> *- M.A. Gates, 1978*

Given that there are at least three disparate types of impacts on fisheries stocks, the impact of each of these factors is sometimes viewed as an "either - or" question— "Is a particular fishery decline due to overfishing, or to contamination, or to habitat degradation?" Scientists are partly to blame for perpetrating this situation, since they have generally avoided investigating these factors simultaneously. Instead, separate groups of independently-trained scientists tend to focus on each factor in isolation: fisheries scientists typically deal with fish population assessments and gear issues; aquatic toxicologists with contamination issues; and benthic and estuarine ecologists with changes in habitat and community structure. As a result, a type of scientific polarization has unintentionally arisen.

A number of researchers have now acknowledged the interrelationship between overfishing, pollution and habitat changes (Barnthouse et al., 1987, 1989, 1990; Buchsbaum et al., 1991; Sindermann, 1994; Dow and Braasch, 1996). While they have advocated an integrative approach, much of the species-specific information on overfishing, contamination and habitat degradation has yet to be translated into comparable measures and totally integrated. For example, Barnthouse et al. (1987) attempted to express the effects of contamination in the same units as for overfishing, although they did not further expand their study by adding in habitat degradation in the same type of common currency.

Novel, integrative techniques need to be developed and applied, if the effects of overfishing, contamination, and habitat degradation are to be compared. In the past, fisheries managers have relied on age-frequency distributions collected during annual stock assessment cruises for assessment of spawning biomass and year class recruitment. Models based on classical population studies were then adapted for fisheries managers (see NRC, 1998a, b for descriptions and assessment of these models). These used age-frequency and age-fecundity distribution patterns and formed the basis for fisheries managers to conclude that overfishing was the main factor affecting fishery stock declines (NOAA, 1998; NRC 1998a, b, 1999). Recently, however, fisheries scientists have developed a number of innovative models to guide fisheries management decisions as they manage fishing effort (Sissenwine and Shepherd, 1987; Fogarty et al., 1992; Myers et al., 1995). These models point out that factors such as environmental change and habitat loss may prevent the recovery of some stocks, such as spring-spawning Icelandic herring (*Clupea harengus*)and Pacific salmon (*Oncorhynchus* spp.; Myers et al., 1995).

One type of model, a recruitment overfishing model, allows managers to calculate maximum fishing mortality rates which would still allow sustainable harvests (Sissenwine and Shepherd, 1987; Fogarty et al., 1992). It has been adapted in the Northeast Multispecies Fishery Management Plan (NEFMC, 1994). This model, which includes estimates of fishing mortality and natural mortality rates, could be modified to include contaminant-induced and habitat-associated mortality.

Aside from the capture and removal of fish and shellfish, other harvesting-related factors must also be considered when assessing fisheries stocks. Juvenile bycatch (pre-recruit individuals of commercially important species) suffer significant mortalities during all seasons of the year (Robinson et al., 1993; NOAA, 1998). Mortality is particularly severe when deck conditions are most extreme, such as in mid-summer or mid-winter. This bycatch mortality has not yet been incorporated into fisheries management models.

Interactions between fish species have also received little attention. Declines in the population of a fished species can have repercussions throughout the community structure. On Georges Bank, for example, the declines in cod, haddock and flatfish were apparently accompanied by an increase in cartilaginous fish numbers. Whereas skates (*Raja* spp.) and spiny dogfish (*Squalus acanthias*) accounted for only 25% of the NMFS survey trawl catch in 1963, their proportions increased to nearly 75% in the early 1990s, but has since declined (NOAA, 1998). Total biomass of the system had remained relatively high, but the biomass of the more commercially-valuable resource species had fallen. Although it now appears that there is no causal connection between the decline in commercially important species and the increase in cartilaginous fish biomass (Murawski, Chapter 2), the community structure has nevertheless shifted dramatically. An ecosystem approach to fisheries management clearly needs to be adapted to ensure sustainability (Parsons, 1992; NRC 1999).

This is not to say that all of the information is currently at hand to resolve the question of whether overfishing, contamination, or habitat degradation is the most important factor affecting fisheries declines. Data gaps are inherent in all scientific pursuits. Habitat data and contaminant effects on populations are particularly scant. There is a real need to identify these gaps and to recommend studies to fill them.

THE NEED FOR A HOLISTIC APPROACH

> *The whole is equal to the sum of its parts.*
> *- Euclid, 365-300 B.C.*

> *The whole is greater than the sum of its parts.*
> *- Gestalt Theory, M. Wertheimer, 1924*

Expressing the issues of overfishing, contamination, and habitat degradation in "either-or" terms has hindered dialogue among fisheries managers, aquatic toxicologists, and benthic ecologists. The question must be rephrased if we are to address it systematically. Rather than examine fish populations in terms of exclusive categories, the more important question is "To what degree do overfishing, contamination and habitat degradation each adversely affect our fisheries resources?" We need to develop approaches that allow us to study the interactions among these three, not just examine their effects individually.

Each of these factors is a form of stress on a population. Populations stressed by one factor are generally more susceptible to additional stresses caused by other factors. Goodyear (1985), for example, demonstrated that contaminant exposure can be more deleterious to fish populations that are also subjected to fishing-induced stress. Barnthouse et al. (1990) described a significant interaction between contaminant-induced mortality and fishing mortality in Gulf of Mexico menhaden (*Brevoortia patronus*) and Chesapeake Bay striped bass (*Morone saxatilis*) populations. Contaminants had little impact on the populations when population levels were high (unexploited populations) but had measurable impacts when populations were low. While this interaction was rather modest, it demonstrated that overfishing and contaminant effects cannot be viewed as independent issues.

We must not overlook the importance of natural environmental variability (both biotic and abiotic factors) and its impact on population and community structure. Often, decadal or multi-decadal cycles of environmental change must be considered in order to understand population shifts (Hofmann and Powell, 1998). Longer-term effects, such as climate change and sea level rise, must also be factored in. While environmental variability is certainly of significance, the importance of fluctuating environmental factors in comparison to anthropogenic influences (overfishing, contamination and habitat destruction) is being debated by the industry, managers, and environmental groups. Nevertheless, holistic models must contain provisions for entering changing environmental variables. In the absence of data on cyclic processes, these models must, at the very least, include stochastic functions.

Consideration of essential fish habitat has recently been incorporated into fisheries management plans (Stevenson 1994; NEFMC, 1999). This should result in a more holistic approach to fisheries management by, in effect, including the interactions among overfishing, contamination, habitat loss and natural factors in the decision-making process. The Atlantic States Marine Fishery Commission (ASMFC 1992) conducted an extensive analysis of habitat requirements in its Winter Flounder Fishery Management Plan, but it was not until the reauthorization of the Magnuson Act (i.e. the Sustainable Fisheries Act of 1996) that all managed species were required to be examined for habitat needs. The ASMFC went so far as to conclude that coastal habitat restoration could result in greater long-term benefits to the fishery than simply reducing fishing mortality.

OUR PART IN THE DEBATE

Our intention with this volume is to examine existing data on the effects of overfishing, contaminants, and habitat degradation on various fish stocks in the Northeast. This assessment will, however, have implications for fishery stock management in areas far beyond those of the Northeast. Our authors offer their perspective on the degree to which overfishing, habitat degradation and contaminants are affecting Northeast stocks, but are limited by lack of studies correlating each factor with population impacts—the common currency. One outcome has been to identify research areas where new data are needed in order to improve our estimates of the importance of these factors, and to integrate them into a more holistic view.

Four types of fisheries will be highlighted in the chapters that follow: groundfisheries, anadromous fisheries, inshore bivalve shellfisheries and the lobster fishery. Each of these fisheries has its own unique blend of the various anthropogenic factors that have impacted their fish populations. These types of fisheries were chosen because of the availability of at least some data on overfishing, contamination, and habitat degradation for each of them. Moring's treatment of anadromous fish (Chapter 3) has come the closest to integrating all three issues within one chapter. The mass of species-specific data on one or more of these three issues has precluded this approach for the other three fisheries. Instead, chapters will address pollution, habitat and overfishing issues separately. Thurberg and Gould (Chapter 4) will highlight the effects of contaminants on groundfish, whereas McDowell (Chapter 7) will cover pollution impacts on shellfish. Murawski (Chapter 2) will address stock assessment and overfishing issues for the Northeast groundfishery, while Brousseau (Chapter 6) will examine inshore bivalve populations, and Steneck (Chapter 8) the lobster fishery. Deegan and Buchsbaum (Chapter 5) will discuss the importance of habitat loss and degradation.

The main goal of this work is to unite the scientific energies of fisheries managers, aquatic toxicologists, and marine ecologists in order to reach a consensus as to the severity of overfishing, contamination, and habitat degradation on our fisheries stocks. The data and background presented in Chapters 2 through 8 provide the basis for the summarization and evaluation of the relative importance of overfishing, habitat degradation and pollution for each of these fisheries, as presented by Buchsbaum (Chapter 9). As further discussed in the final chapter (Pederson and Robinson), this holistic approach may challenge fisheries managers to modify the way in which they manage each of the Northeast fisheries. It is our hope that the conclusions presented here can then be conveyed to the general public, eliminating at least some of the confusion that currently enshrouds these issues.

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Chapter II

The New England Groundfish Resource: A History of Population Change in Relation to Harvesting

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INTRODUCTION

The fishing industry of New England has, for over 400 years, been identified both economically and culturally with groundfishing (German, 1987; Hennemuth and Rockwell, 1987). A mixture of bottom-dwelling fishes including cod, haddock, redfish and flounders and allied bottom-dwelling species constitute the groundfish resource (Table 2-1). Many of the groundfish resources off New England are now recovering from record low stock sizes and landings observed in the early 1990s (Clark, 1998). Management measures have only recently resulted in demonstrable reductions in groundfish mortality rates to levels low enough to allow stock rebuilding (Northern Demersal Working Group, 2000). These recent reductions in fishing pressure required unprecedented regulation of a fishery that historically was allowed to operate virtually unconstrained (Anthony, 1990, 1993; Fogarty and Murawski, 1998).

Important historical themes in these fisheries were: (1) almost continuous change since the turn of the 20th century, owing to ecological, political and market trends, (2) gear sectors in competition for grounds, labor and fish, (3) an eastward movement of the fleet to the Scotian Shelf and Newfoundland, followed by a westward contraction owing to changing markets and regulations, (4) “writing-off” of collapsed stocks, as effort expanded, and fisheries diversified, (5) failure to effectively deal with conservation problems in a timely fashion and to implement recommendations of scientific investigations, leading to (6) missed opportunities to establish sustainable fisheries or avoid major declines in production. This paper examines the exploitation history for important groundfish resources off New England, and the role of fishing and other factors influencing stock abundance and recruitment. Research and management challenges in achieving stable and productive fisheries for these stocks are discussed.

HISTORICAL PERSPECTIVE

DEVELOPMENT OF THE MODERN FISHERY

In the late 19th and early 20th centuries large fleets of vessels sailed from Gloucester, Boston and other New England ports to fish local and distant offshore banks as far away as the Grand Banks off Newfoundland and Labrador (German, 1987; Kurlansky, 1997; Murawski et al., 1997). Catches of salt cod supported nearly 400 schooners in each of these main ports, and a multitude of shore-side businesses including salt mining, ice harvesting in fresh-water ponds, and a boat building industry that made the shipyards on the Essex River, north of Boston, among the busiest and best known in the world. The fish landed in New England and the Maritimes eventually supported the infamous ‘triangle trade’ with Caribbean and west African countries and colonies (salt cod, molasses, and slaves; Kurlansky, 1997).

Table 2.1. Species and stocks comprising the New England groundfish resource.

Common Name	Scientific Name	Management Stocks	Included in Multi-species FMP?
Atlantic cod	*Gadus morhua*	Georges Bank South Gulf of Maine	yes yes
Haddock	*Melanogrammus aeglefinus*	Georges Bank Gulf of Maine	yes yes
Acadian Redfish	*Sebastes fasciatus*	Gulf of Maine	yes
Pollock	*Pollachus virens*	Gulf of Maine	yes
White Hake	*Urophysis tenuis*	Gulf of Maine	yes
Red Hake	*Urophysis chuss*	Gulf of Maine/N. Georges Bank S. Georges Bank/Middle Atlantic	yes yes
Silver Hake	*Merluccius bilinearis*	Gulf of Maine/N. Georges Bank S. Georges Bank/Middle Atlantic	yes yes
Ocean Pout	*Macrozoarces americanus*	Gulf of Maine/S. New England	yes
Atlantic Halibut	*Hippoglossus hippoglossus*	Gulf of Maine/Georges Bank	yes
Winter Flounder	*Pleuronectes americanus*	Georges Bank Gulf of Maine S. New England	yes yes yes
Witch Flounder	*Glyptocephalus cynoglossus*	Gulf of Maine	yes
Yellowtail Flounder	*Limanda ferrugineus*	Georges Bank S. New England Cape Cod Middle Atlantic	yes yes yes yes
American Plaice	*Hippoglossoides platessoides*	Gulf of Maine	yes
Windowpane Flounder	*Scopthalmus aquosus*	Gulf of Maine/N. Georges Bank S. Georges Bank/S. New England	yes yes
Cusk	*Brosme brosme*	Gulf of Maine	no
Wolffish	*Anarchichas lupus*	Gulf of Maine	no
Spiny Dogfish	*Squalus acanthias*	NE USA and Canada	no
Skates	*seven species*	Gulf of Maine/Middle Atlantic	no
Goosefish	*Lophius americanus*	Gulf of Maine/N. Georges Bank S. Georges Bank/Middle Atlantic	no no

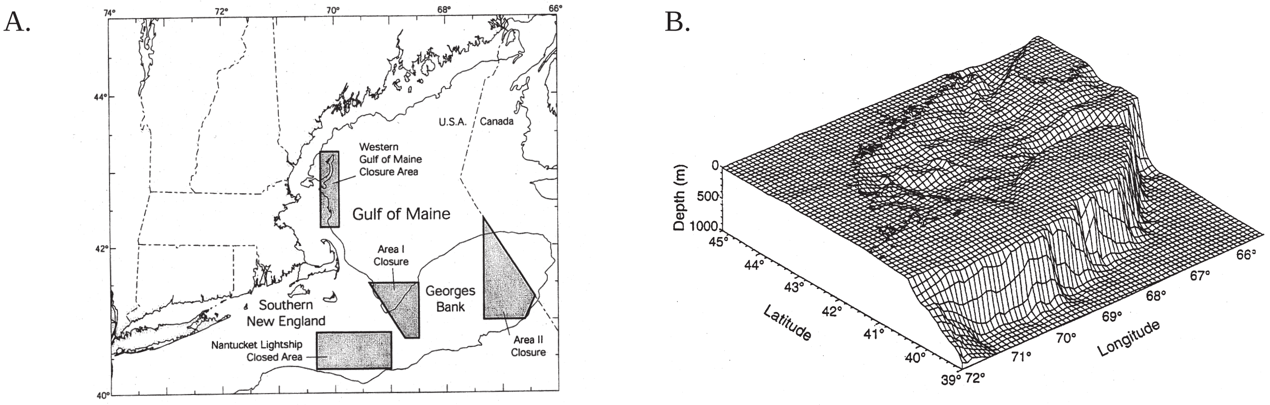

Figure 2.1. (A) Areas closed to fishing in 1994 and the U.S./Canada boundary decided by the World Court. (B) New England shore areas and fishing grounds. Year-round fishery closures for the protection of groundfish are shaded.

The New England groundfish industry changed significantly around the turn of the century. During this period, there were major shifts in how fish were caught, handled, processed, distributed and sold (Herrington, 1932; Hennemuth and Rockwell, 1987; Serchuk and Wigley, 1992). Once dominated by artisanal, community-based fishermen, the resource was subjected to increasing levels of industrialization, first by company-based fleets of long-liners and gillnetters. At the turn of the century, steam-powered trawl vessels were specially-built to harvest flounders and haddock along the smooth bottomed areas along the continental shelf south and east of New England (Fig. 2.1 and 2.2; Anonymous, 1906; Alexander et al., 1915). The introduction of the steam-powered trawler based on designs from England (Anonymous, 1906) fundamentally changed how groundfish were caught, and rapidly replaced the schooner fleets. While it was apparent that some stocks were depleted by the schooner fleets (e.g., halibut on Georges Bank suffered significant declines in productivity by the 1850s), overfishing of various resources and other issues of fisheries management became more problematic with the introduction of trawling (Alexander et al., 1915; Herrington, 1932).

DOMESTIC OVERFISHING

By 1930 there were clear signs that the fleet had grown too large in relation to the capacity of the stocks to sustain growth in landings (Herrington, 1932). Haddock landings peaked in 1929, but declined rapidly thereafter, as stocks were less abundant on Georges Bank (Fig. 2.3). This prompted the development of a modern study of the population dynamics of haddock, headed by Dr. William Herrington (Herrington, 1932, 1947).

> *"It is only in the last few years when the fishing fleet has suffered from a marked scarcity of haddock that the folly of (the) belief in the inexhaustibility of nature has become potent." (Herrington, 1932).*

A major focus of the program was to document harvesting practices and to determine appropriate mitigation measures. The research by Herrington

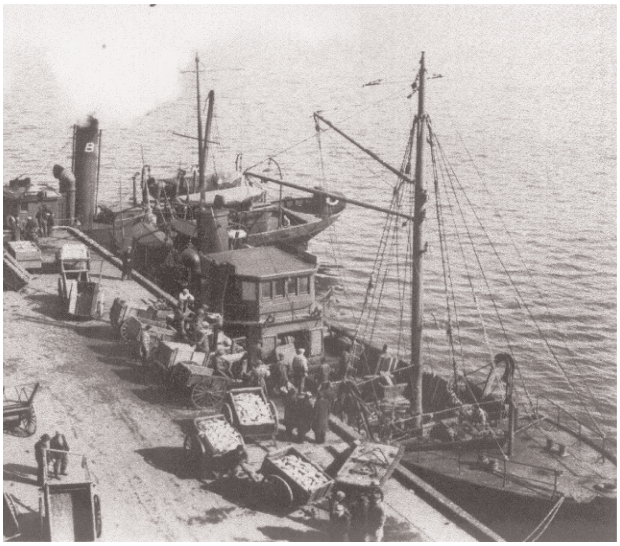

Figure 2.2. Otter trawl fishing vessels at Boston Fish Pier, ca. 1931. The vessel at the end of the pier is the Spray. Built in 1905 it was the first steam trawler introduced into the U.S. groundfish fleet.

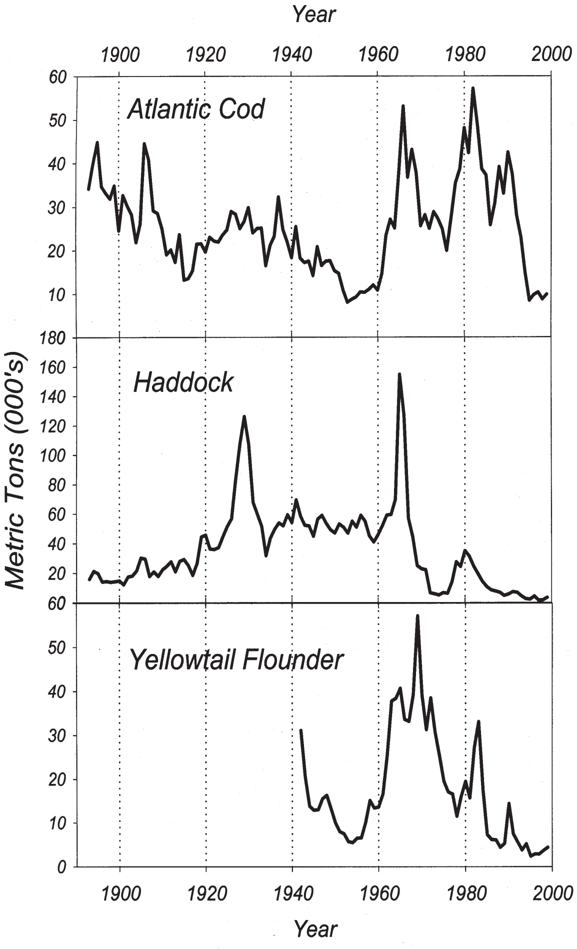

Figure 2.3. Total landings of Georges Bank cod and haddock and landings of yellowtail founder from all New England waters, 1893-1999.

(1932) and Graham (1952) confirmed the earlier work of Alexander et al. (1915) demonstrating the large discard of juvenile haddock, and great potential waste of the resource. At this time catches of very small fish were common, with a large fraction of fish being under 17 inches in length (43 cm). It was readily apparent to researchers and some members of the industry that the yield potential of fish was not being realized since they were being caught at a relatively young age, before achieving most of their growth potential (e.g. “growth overfishing”). More troubling was the tremendous numbers of discarded “baby” haddock, that were below commercially useful size (Herrington, 1932; Graham, 1952). Comments by one Clarence Birdseye confirmed industry leader’s concerns for wasteful practices (see comments at the end of Herrington, 1932). Scientific investigations using sea sampling showed just how destructive the trawl technology was. In 1930 the fishery landed 37 million haddock at Boston, with another 70-90 million juvenile haddock discarded dead at sea (Herrington, 1932). The very small mesh size used in the nets was judged the cause, and yet mesh size regulations to protect haddock were not implemented until the U.S. had the authority to do so under the auspices of the International Commission for the Northwest Atlantic Fisheries (ICNAF), beginning in 1953 (Graham, 1952). Interestingly, a similar study published in 1915 (Alexander et al., 1915) also used sea sampling to document the high rates of haddock discard by the otter trawl fishery.

Prior to WW-II the fleet was large in size, but profitability was low (Dewar, 1983). The war years were again prosperous for the industry as production was boosted, and protein demands and rationing necessitated higher fish consumption. The fleet was reduced at this time, as many of the largest trawlers were requisitioned for war duty as mine sweepers. The return of these vessels from war, along with reduced demand resulted again in low profitability to the fleets (Dewar, 1983). Development of new markets such as selling ocean perch in the midwest as a substitute for Great Lakes yellow perch sustained the offshore fleet. Many government subsidy programs were launched after the war (Dewar, 1983).

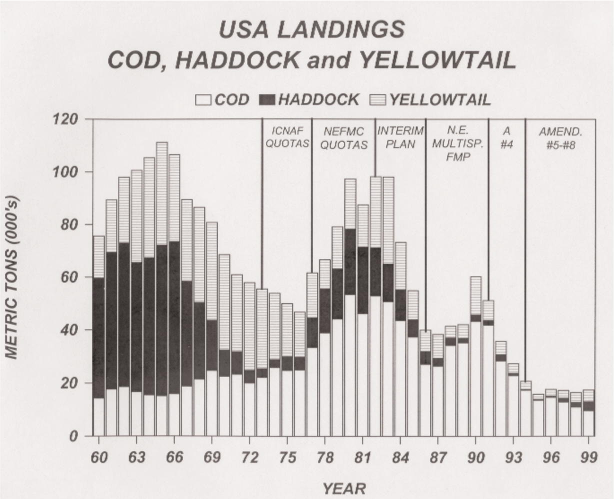

Figure 2.4. Total USA landings of Georges Bank cod, haddock and yellowtail founder, 1960-1999.

Distant-Water Fleets

Scouting vessels for the Soviet fleets first ventured into New England waters in 1961 (Hennemuth and Rockwell, 1987). Their target was Atlantic herring and their fishery took about 63,000 metric tons that year. In subsequent years, the fishery for herring expanded, and other species including silver and red hake, and haddock were targeted (Brown et al., 1976; Mayo et al., 1992; Fogarty and Murawski, 1998). From 1960 to 1966 total groundfish landings increased from about 200,000 metric tons, to about 760,000 metric tons. Landings of haddock reached an all-time record of 154,000 in 1965, and declined rapidly thereafter (Fig. 2.3). Between 1964-1967 total groundfish landings were comprised primarily of silver hake, haddock, red hake, flounders and cod. Herring landings peaked in 1968 at 439,000 metric tons, and declined rapidly with the collapse of the Georges Bank herring stock (Anthony and Waring, 1980). The intensive mackerel fishery occurred in the early 1970s, with landings peaking in 1972 at 387,000 metric tons (Anderson and Paciorkowski, 1980).

Effort exerted by the distant-water fleets thus shifted from one abundant target stock to the next, in a typical pattern of sequential resource depletion (Hennemuth and Rockwell, 1987). Restrictive management actions, enacted beginning in the early 1970s severely limited catches, and distant water fleet effort declined accordingly (Hennemuth and Rockwell, 1987; Mayo et al., 1992). Total standardized fishing effort had increased four-fold on

Figure 2.5. NOAA R/V Albatross IV, launched in 1962, and responsible for most standardized bottom trawl survey cruises between 1963 and 2000.

Georges Bank between 1960 and 1972 (Mayo et al., 1992). Under these high effort levels, fishing mortality rates increased to unprecedented levels, and landings and stock sizes declined (Fig. 2.4).

The Bureau of Commercial Fisheries instituted an innovative program to gain fishery independent data on fish abundance off the Northeast USA (Grosslein, 1969; Smith, 2000) beginning in 1962. Although standardized research vessel surveys had begun in the late 1940s, most fishery research conducted on the northeast shelf consisted of single- species studies of fisheries involving data primarily derived from commercial fishing operations (Smith, 2000). Commercial fisheries data have obvious biases due to the concentration of fisheries in known areas of high density, and non-commercial components of the ecosystem could not be effectively monitored using data solely from fisheries. A new program was developed using statistically rigorous stratified random sampling designs. Beginning with the delivery of the NOAA Albatross IV (Fig. 2.5) in 1962, the Northeast Fisheries Science Center initiated what has become the longest continuously operating survey of its scope in the world (Grosslein, 1969; Azarovitz et al., 1997). The survey has proved to be an invaluable tool to monitor specific stocks and species assemblages independent of biases and data quality considerations inherent in fishery-dependent data (Brown et al., 1976; Clark, 1981; Clark and Brown, 1977).

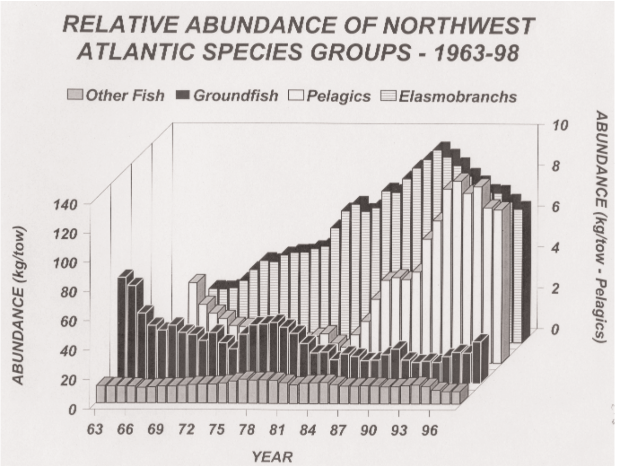

Figure 2.6. Relative abundance of four finfish species groups from fishery-independent surveys (stratified mean catch/tow in kg from NMFS bottom trawl surveys), 1963-99.

Abundance, as measured by the Northeast Fisheries Science Center research vessel surveys, declined rapidly as various components of the demersal and pelagic systems were pulse-fished by the distant-water fleets (Fig. 2.6; Brown et al., 1976; Clark, 1981). The Georges Bank haddock resource collapsed under the pressure from distant water fisheries, failing to produce anything but poor year classes between 1964 and 1974 (Northern Demersal Working Group 2000). Other stock collapses included silver and red hakes, Atlantic mackerel and the Georges Bank herring stock (Fogarty and Murawski, 1998).

Beginning in 1973, quota-based management was instituted under the auspices of the International Commission for the Northwest Atlantic Fisheries (ICNAF; Hennemuth and Rockwell, 1987; Fig. 2.4). Quotas for each species were allocated by country, with the sum of each species equal to the total recommended removals. Additionally, ‘second-tier’ quotas, less than the sum of a country’s species allocations, were intended to mitigate the effects of non-targeted bycatch, so that species quotas would not be exceeded. The quota system under ICNAF effectively ended directed distant-water fisheries on New England groundfish resources, as these resources were determined to have little capacity to support fisheries beyond the levels that would be taken by the United States and Canada. Quotas were progressively lowered on mackerel, herring, squids and other species, as these resources declined as well.

200-MILE LIMIT

The clamor for the U.S. to assert control over waters out to 200 miles was great. The U.S. Congress enacted the Magnuson Fishery Conservation and Management Act of 1976, taking control of the exclusive economic zone (EEZ), and setting up a system of regulation of the domestic industry. Fueled by great expectations, the U.S. fishing industry expanded rapidly (Fordham, 1996). The fleet, once dominated by wooden side-trawlers, was replaced virtually overnight by steel stern-trawlers which were equipped with modern technology for locating, catching and handling fish. Quota-based regulations, a hold-over from the last days of international restrictions, were an anathema to the growth of the revitalized U.S. groundfish fleet. Catch quotas were abandoned, in favor of ineffective measures to control the size of meshes in the nets, and the minimum length of fish landed (Fig. 2.4; Anthony, 1993).

> *"No one knew exactly how many newcomers had arrived during the last four months of 1977, but according to one report, new boats entered the fishery at the astounding rate of about one every four days." (Dewar, 1983).*

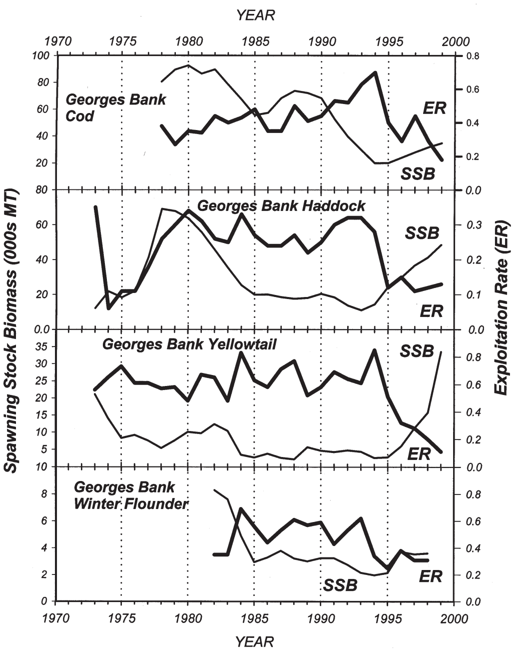

Figure 2.7. Changes in spawning stock biomass (SSB, 000s of metric tons) and exploitation rate (ER) for four New England groundfish stocks, 1973-1999.

Relatively strong year classes of cod, haddock and some other groundfish stocks produced in 1975 and later resulted in improved resource conditions, and increased groundfish abundance and effort in the late 1970s and early 1980s (Fig. 2.4, 2.6 and 2.7). Between 1976 and 1984 USA otter trawl fishing effort doubled (Mayo et al., 1992; Anthony, 1993; Fogarty and Murawski, 1998), with many new fishing vessels entering the fishery (Dewar, 1983). These newer vessels were more technologically advanced, safer, and more capable of fishing in foul weather than the vessels they replaced. As a result, their fishing power was substantially higher than those in the fleet prior to 1976.

Groundfish abundance again peaked in the early 1980s, primarily as a result of improving resource conditions for cod and haddock (Fig. 2.6 and 2.7). However, in the face of rapidly expanding fishing effort, abundance of the groundfish complex declined precipitously (Fig. 2.6 and 2.7). In the Gulf of Maine, high relative catch rates were supported in the late 1970s and early 1980s by redfish, haddock, cod (Serchuk et al., 1994) and mixed flounders. From 1970 onward the cod resource became the mainstay of the New England groundfish catch, as haddock and then yellowtail flounder resources collapsed (Fig 2.4). The Canadians had extended their territorial jurisdiction 200 miles seaward, excluding U.S. vessels which had fished off the Scotian Shelf and the southern Grand Banks for generations. With the return of the redfish fleet from the Scotian Shelf, as Canada extended its jurisdiction, the residual Gulf of Maine redfish resource was quickly depleted (Clark, 1998). Overlapping territorial claims in the Georges Bank region between the U.S. and Canada resulted in high-level diplomatic negotiations. In 1979 a draft treaty on reciprocal fishing rights was agreed to at the ministerial level. The treaty recognized historical fisheries by the U.S. off Canada, and vice-versa. However, with the change in administrations in 1980, and opposition from some segments of the U.S. fishing industry, the draft treaty was not ratified by the U.S. government. Ultimately, the boundary between the U.S. and Canada was settled in the World Court (Fig. 2.1). Americans were barred from fishing areas off Canada, and areas in the northern part of Georges Bank, where so much of the haddock landings of the 1920s-1950s had been taken.

On Georges Bank, stock sizes of haddock, cod and yellowtail flounder, which had improved in the late 1970s-early 1980s again declined as spawning biomasses and recruitment diminished (Fig. 2.7). The strong 1987 year class of Southern New England yellowtail flounder, was quickly fished out (Northern Demersal Working Group, 2000).

As a result of the failure of indirect controls to prevent overfishing (NEFSC, 1987), environmental groups sued the Department of Commerce in 1991 (Fordham, 1996). The court settlement of the law suit required the New England Fishery Management Council to develop a fishery management plan to end overfishing and rebuild depleted stocks, the result of which was Amendment #5 to the Northeast Multispecies (groundfish) FMP, implemented in 1994. This plan required a reduction in groundfish effort by 50% over 5-7 years, increased mesh sizes, expanded closed areas, a moratorium on new effort in most fleet sectors, and mandatory reporting (logbooks).

Amendment #5 was implemented in May 1994. However, in June of that year, new fishery stock assessments indicated that the resource condition had deteriorated to the point that scientists warned:

> *"Failure to take strong management actions now to preserve the limited spawning biomass for Georges Bank cod may have severe and potentially long-lasting consequences for both the stock and fishery."* *-NEFSC, 1994*

In response, scientists recommended "...substantial, immediate reductions in groundfish fishing mortality on Georges Bank", and that "...fishing mortality for cod and yellowtail flounder be reduced to as low a level as possible, approaching zero" (NEFSC, 1994). In response to the poorer prognosis for groundfish stocks, particularly on Georges Bank, the Secretary of Commerce instituted a series of measures under his emergency authority. Chief among the measures instituted was the closure in December of about 17,000 km^2 to groundfish fishing on Georges Bank and in southern New England (Fig. 2.1). The areas have remained closed to groundfishing since then. Other measures, including the closure of additional areas in the Gulf of Maine (Fig. 2.1), trip limits on some species and increases in mesh size were also instituted as part of later plan amendments.

RECENT EVENTS

Owing to the decrease in fishing effort (days fished), primarily by offshore trawlers, and the implementation of other measures including closed areas, exploitation rates for some stocks have decreased substantially in recent years (Fig. 2.7). In particular, exploitation rates of Georges Bank yellowtail flounder, haddock, and to a lesser extent cod, declined to less than 20% per year, from levels, in the case of yellowtail flounder, of up to 80% (Fig. 2.7; Northern Demersal Working Group, 2000). The reduction in fishing mortality has had a significant impact on the age distribution of the stocks, especially for haddock and yellowtail flounder, where older ages/larger sizes are more abundant than in recent years when exploitation rates were excessive (Northern Demersal Working Group, 2000). Increased survivorship of older age groups is thought to be important in groundfish stocks, owing to improved hatching rates and larval survival due to maternal spawning experience and size effects (Trippel et al., 1997; Murawski et al., 1999). In the case of some New England groundfishes, spawning had become increasingly reliant on first-time spawners in years prior to 1995 (Wigley, 1999).

Improved survival of older age groups is the primary reason for modest increases in spawning stock biomass (SSB) for Georges Bank cod (Northern Demersal Working Group, 2000); recruitment of the Georges Bank cod stock remains poor. In contrast, improved recruitment combined with higher adult survivorship has increased yellowtail flounder SSB to the highest level observed in the analytical stock assessment time series (e.g., since 1973; Fig. 2.7). Haddock recruitment remains well below the historic (1931-1999) average, but the 1998 year class is the largest since 1978, and is projected to continue expansion of SSB when recruited to the spawning population (e.g., 2001).

Although landings of Georges Bank groundfish stocks have remained stable since 1995, the species composition of landings reflects modest increase in

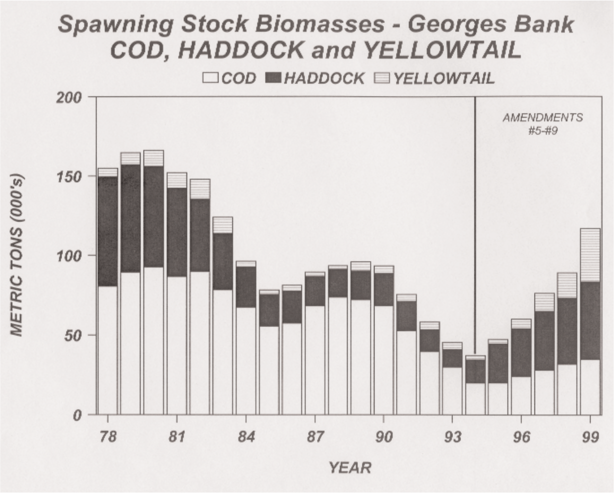

Figure 2.8. Spawning stock biomasses (thousands of metric tons) for Georges Bank cod, haddock and yellowtail flounder, 1978-1999 (Northern Demersal Working Group 2000).

diversity among cod, haddock and yellowtail flounder (Fig. 2.4). Spawning biomass increases among the species better reflect the increasing diversity (Northern Demersal Working Group, 2000; Fig. 2.8), as landings have been seriously constrained. Aggregate spawning biomass for the three Georges Bank stocks is now higher than any time since 1983, and will increase in the next few years (Northern Demersal Working Group, 2000).

Other groundfish stocks (Table 2.1), have showed variable trends in exploitation rate and stock biomass (Clark, 1998; Northern Demersal Working Group, 2000). In general, stocks on Georges Bank have lower exploitation rates and are further along in rebuilding to long-term biomass targets than are stocks in the Gulf of Maine region. In particular, Gulf of Maine cod and white hake resources, concentrated in the Gulf of Maine, have biomass <40% of the biomass that would produce maximum sustainable yield (B_{MSY}), and are exploited at rates above rebuilding targets.

As traditional target stocks declined, the New England groundfish fishery re-targeted to exploit other available resources, including goosefish (monkfish), spiny dogfish, skates, white hake, northern shrimp and other stocks. Two important species to which effort was refocused were goosefish (monkfish) and spiny dogfish. Landings of both species increased substantially in the early 1990s, reflecting increased directed fishing. In both cases, however, the level of harvest resulted in non-sustainable harvest rates, and populations (particularly of mature animals) declined (Clark, 1998; Rago et al., 1999). Fishery management plans for both stocks were developed and are now implemented to reduce fishing mortality and rebuild the stocks to B_{MSY}.

The dramatic decline in groundfish abundance in the late 1980s was accompanied by a variety of changes in other fish components of the ecosystem (Fig. 2.6). In particular, there were rapid and significant increases in principal pelagics (Atlantic herring and Atlantic mackerel), as well as the small elasmobranchs (spiny dogfish and skates). The abundance of mackerel and herring declined to very low levels in the late 1970s, but has since rebounded to historic proportions (Clark, 1998). Fishing mortality of herring and mackerel remains very low, as compared with sustainable harvest rates, and those observed when the distant-water fleets targeted them. Most of the increase in elasmobranch abundance was due to the increase in dogfish, particularly since 1980 (Rago et al., 1999). The abundance of mature dogfish has declined substantially in recent years, and the dearth of mature females in the population has produced very poor recruitment. Managers have severely curtailed directed fishing for dogfish in order to be able to eventually restore the biomass of mature dogfish to that necessary to produce MSY.

Skate resources on the northeast shelf are comprised of seven species (NEFSC, 2000). The large-bodied species (winter skate, barndoor skate and thorny skate) have showed significant signs of overfishing, due to their vulnerability to harvest and (in the case of winter and thorny skate) directed overfishing. Barndoor skate abundance declined significantly prior to 1970, and has only recently made a modest increase in the past several years (NEFSC, 2000). Winter skate abundance on Georges Bank peaked in the mid-1980s and declined substantially thereafter. Thorny skate abundance has declined throughout the past 30 years. The small-bodied skates (smooth, rosette, and little), have shown stable or increasing trends in recent years (NEFSC, 2000).

As a group, groundfish resources have undergone episodes of overfishing, in the usual scenario of discovery, build-up of directed harvest, overfishing and stock collapse. In several cases (haddock, yellowtail flounder) there were a number of such episodes during the 19th and 20th centuries, while in others (redfish, Atlantic halibut, barndoor skate), the population dynamics of extreme K-selected species has precluded such cyclic response. For those stocks where fishing has been reduced to low levels following a stock decline, in virtually all cases, a substantial recovery of biomass and recruitment has ensued. Atlantic herring on Georges Bank were virtually extirpated in the mid-1970s, but now appear to be at historic high levels. Likewise, mackerel recovered from overfishing to unprecedented high stock abundances. Numerous groundfish stocks including redfish, haddock, yellowtail flounder, witch flounder and others have increased substantially following relaxation of exploitation. There are a few exceptions, however. Red and silver hake populations in the Middle Atlantic Bight have failed to recover following significant overexploitation by the distant water fleets (Clark, 1998). Gulf of Maine stocks of the same species have fared much better, deepening the mystery of the lack of recovery of these populations. Several theories as to the lack of recovery of the two southern hake stocks include habitat destruction by demersal fishing gear, changes in the trophic composition of the system increasing predation pressure on juveniles, continued overfishing of the stock (and in particular catch and bycatch of juveniles) and changes in oceanographic conditions necessary for effective reproduction. These two stocks apart, the dominant factor controlling the population abundance of northeast groundfish stocks has been fishing.

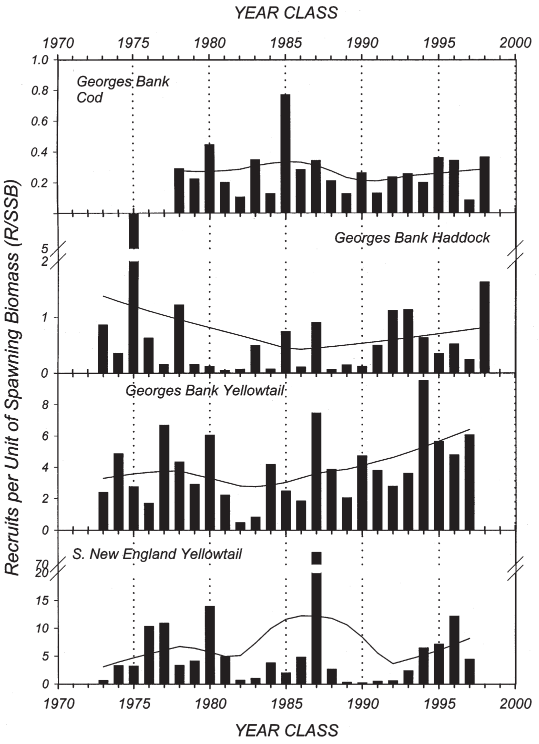

Figure 2.9. Changes in recruitment survival (measures as recruits at age 1 per kg of spawning stock biomass) for four New England groundfish stocks, 1973-1998. Horizontal lines are LOWESS smooths, assuming a tension value of 0.5.

The Roles of Overfishing and Environmental Variation

Herrington (1932) and Graham (1952) clearly demonstrated growth overfishing (excessive harvest preventing maximum yield from a given number of young fish over their life span) and incredible waste of the Georges Bank haddock resource, owing to the very young age at selection and high discards by the fishery. Growth overfishing was a serious problem for most of the period before 1994, even with increases in minimum mesh size to 5-1/2 in. for the directed groundfish fishery, owing to the mis-match with minimum legal fish sizes (NEFSC, 1987). Only in the last several years does it appear that the combinations of harvest rates (exploitation rates below ~25%), and selection characteristics of the gear extant in the fisheries result in near-optimum yield per recruit for some stocks (Northern Demersal Working Group, 2000).

There is ample evidence for significant recruitment overfishing of many of New England's fishery resources throughout most of the 1980s and early 1990s (Sinclair and Murawski, 1997). Recruitment overfishing occurs when the reproductive capacity of the stock is decreased by fishing, so that the level of recruitment, on average, is substantially lower than when the spawning stock is larger—more mommies, more babies. Haddock are a good example of a stock substantially recruitment overfished during various times during the 20th century (Overholtz et al., 1986; Overholtz et al., 1999). Although there is a great deal of variation in haddock stock and recruitment data, it is nonetheless apparent that at stock sizes below about 80,000 metric tons, the probability that year classes >25 million fish will be produced is greatly diminished. By keeping the stock below the 80,000 metric ton level (e.g. since the late 1980s), the population has had insufficient reproductive capability (numbers of eggs spawned) to generate year classes in excess of 25 million fish, except in years of unusually high survival of age 1 fish (e.g. 1975; Fig. 2.9). That the 1978 year class was good, and the product of the high spawning biomass of the 1975 year class (at age 3) and not unusually high survival rate (Fig. 2.9) is an important demonstration of the role of spawning biomass in determining year class strength. During the 1930s through early 1960s, the Georges Bank haddock produced year classes in excess of 25 million fish regularly, and very good year classes (>50 million fish) in about half of the years (Overholtz et al., 1999). The 1998 year class appears to be in excess of 25 million fish, and, if conserved, should increase the SSB to over 80,000 metric tons in 2002 or 2003 (Northern Demersal Working Group, 2000). Analysis of the historical record indicates that recruitment prospects for Georges Bank haddock should continue to improve.

Most northeast groundfish stocks exhibit significant, albeit noisy, stock-recruitment relationships (Brodziak et al., 2000). Given the substantially greater likelihood of good recruitment at SSB's higher than the median, there is convincing evidence that maintaining high spawning stock will produce benefits in higher and more regular recruitment and yields to the fisheries.

Natural environmental variation is a substantial contributing factor to the strength of individual groundfish year classes, and, if not properly accounted for, can exacerbate recruitment declines due to overfishing (Werner et al., 1999; Fogarty et al., 1996). The survival of young fish (expressed as a ratio of the number of age 1 produced per kilogram of SSB) exhibits important patterns of variation, as illustrated by some New England groundfish resources (Werner et al., 1999; Fig. 2.9). Recruitment survival was generally good for haddock in the early 1960s, but declined in the late 1960s and early 1970s (Werner et al. 1999). This pattern was generally similar for haddock on Georges Bank and on Browns Bank off Southwest Nova Scotia, suggesting some level of geographic coherence, perhaps due to regional-scale environmental factors. Recruitment survival improved for haddock in the late 1970s, but again declined in the 1980s. Since 1990, recruitment survival has improved for numerous groundfish stocks (Fig. 2.9).

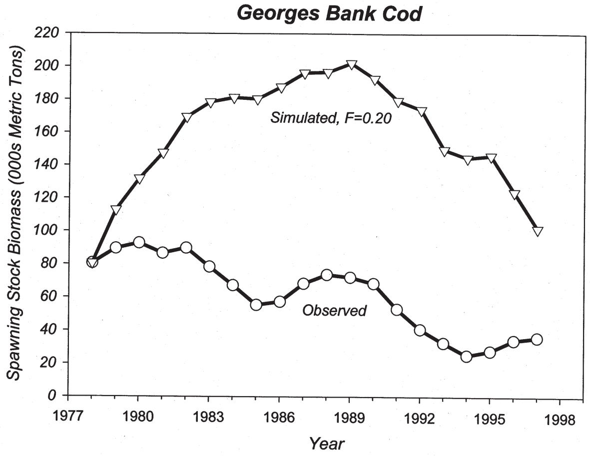

Figure 2.10. Observed and simulated spawning stock biomass of Georges Bank cod, 1978-1997. Simulated SSB assumed that the stock was fished at F=0.2 (16% exploitation rate) for years 1978-1997. Recruitment was assumed to be that observed at age 1.

The relationship between natural variation in recruitment survival and effects of fishing were evaluated in a simple simulation model (Fig. 2.10). Beginning in 1978 the calculated numbers of cod at age was subjected to various fishing mortality rate scenarios. Two simple assumptions of cod recruitment were used: (1) the annual numbers of recruits (age 1) varied as observed in the fishery stock assessment (Northern Demersal Working Group, 2000), and (2) the observed pattern of recruitment survival (R/SSB), was multiplied by the simulated SSB to derive total recruitment. The importance of fishing to the overall level of SSB and the trend is given in Fig. 2.9. In this case, the only difference between the scenarios was the fishing mortality rate (Fig. 2.4 and 2.9). When the stock was fished at a low level (fishing mortality rate=F=0.2), the SSB peaked at over 200,000 mt in 1989, nearly three times the observed level. Although SSB declined from 1989 to 1997, the absolute level of SSB at the end of the simulation was over twice as high when F was low. The overall yield obtained in the low-F case was about equal to that assuming higher Fs, but was less variable from year to year. In the last few years of the simulation, the low-F scenario produced substantially higher landings. This simulation assumed annual recruitments as calculated from the actual high-F scenario. In all likelihood, these higher simulated SSBs would have generated even higher recruitments, thereby producing still higher SSBs and landings. When the simulation was run with observed R/SSB values and simulated SSBs, the landings and stock sizes under the low-F scenario were substantially greater yet. This simple example shows the importance of maintaining proper exploitation rates during periods of both good and poor recruitment survival. When recruitment survival was good (early-mid 1980s), SSBs and likely absolute recruitment would have been much greater than that observed. In periods of poor recruitment survival, maintaining low Fs (or even reducing Fs when recruitment survival was poor) would have resulted in smaller declines in SSB, and perhaps set the stage for a quicker turn around in SSB and yields once conditions for recruitment survival improved. Thus, rather than being an alternative explanation for observed patterns of stock variation, natural fluctuations in the survivorship of young fish may have exacerbated declines occurring due to overfishing. An important empirical observation (at least for northeast groundfish) is that a conservative approach to exploitation rates, perhaps including adaptive reductions in F when survivorship is poor, will produce greater long-term benefits and more stable catches than the opposite pattern of fishing mortality ‘ratcheting up’ eventually leading to stock decline, typical of open access fisheries (Ludwig et al., 1993). The experience in the northeast groundfish fisheries is a convincing case history of recruitment overfishing and the “ratchet effect”—a case that need not be repeated (Ludwig et al., 1993; Sinclair and Murawski, 1997).

CONCLUSIONS

Groundfish resources in the offshore New England region have varied considerably in abundance and landings during the last 10 decades, primarily due to their exploitation history. Dramatic reductions in most offshore stocks occurred as a result of systemic recruitment overfishing by the distant water fleets, who sequentially depleted the wide array of species available. Subsequent to the end of distant-water fleet fishing, some stocks rebounded to very high levels, only to be overfished once again. The Atlantic herring stock on Georges Bank was virtually extirpated in the 1970s, but has returned to relatively high abundance, and is now occupying historically important spawning areas. The Atlantic mackerel stock has, as well, increased in abundance following intensive overfishing in the early 1970s.

Groundfish, however, did not fare well under domestic management following adoption of the Magnuson Fishery Conservation and Management Act (Fordham, 1996). Most stocks of groundfish declined to near record low levels of abundance by the early 1990s, precipitating intervention in the management of the resources by the federal courts. Fishing practices during much of this period reduced the inherent resilience of the populations by removing many of the older (breeding) fish and resulting in the fisheries depending almost completely on the strength of incoming year classes “recruitment fisheries”; Murawski et al., 1999). At lower exploitation rates, the population would be comprised of a greater diversity of age groups, and thus, if recruitment of the incoming cohort is low, the fishery could concentrate for a while on the accumulated stock of older animals. In the case of New England groundfish, however, high rates of exploitation obviated this option. The dependence on the recruitment of young fish resulted in great economic incentives to target animals at or near legal sizes. Retention and discard of juveniles became more problematic.

Improvements in some resources followed implementation of direct effort controls (prescribing a 50% reduction in days at sea), along with the first ever moratorium on new vessel entrants into the New England groundfish fishery, closure of large blocks of productive fishing area, and other measures (Fogarty and Murawski, 1998; Murawski et al., 2000). The role of marine protected areas, such as the Georges Bank closures, in long-term conservation of resources and ecosystems is a current subject of intense speculation and study. Closed areas on Georges Bank are important nursery areas for a variety of groundfish and other species (Murawski et al., 2000). Given the potential for habitat destruction by heavy towed gears such as otter trawls and scallop dredges, it is possible that improved recruitment may result from permanent protection of nursery areas providing high quality feeding opportunities and cover from predation. At this point, such mechanisms have not been verified through scientific investigation, although some studies have been instituted (Collie et al., 1997). Clearly, managers have found an adequate combination of measures that has allowed stock rebuilding to occur on Georges Bank (Murawski et al., 2000; Northern Demersal Working Group, 2000). Some resources such as Georges Bank yellowtail flounder are approaching target biomass levels, and could be harvested at increased rates, while others will require additional protection to achieve long term rebuilding (Applegate et al., 1998). The challenges for the next several years will be to manage the resource in a way that will allow less productive resources to meet their biomass and yield potentials while considering additional fishing opportunities for more productive stocks. Biomass goals have been established for all of the significant resource species, but these targets have been calculated from information collected from stocks that have been exploited at or above their optimum rates for all of the recorded history. It is possible that yield potentials for some stocks may be much greater than the maximum landings recorded in the fishery, owing to growth or recruitment overfishing. Thus, an adaptive approach to managing recovering resources to assure that the full productive potential of the resources is realized is an appropriate goal for future research and management. In the future, consideration of the potential ecological constraints to the simultaneous optimization of biomasses and yields of the array of resources species will become more important (Murawski 2000), but there is considerable empirical evidence that at current levels of abundance, predation and competition are not significant impediments to stock rebuilding for New England groundfish resources.

The history of exploitation of New England's groundfish resource has produced a repeated record of failure to address identified conservation problems followed by inevitable stock collapse and economic dislocation. The dominance of fishing as the primary factor in determining the abundance of resource species, and in fact, the structure of the fish component of the New England offshore ecosystem has been established. Environmental variation, working to increase or decrease recruitment survival, has exacerbated the effects of overfishing in some instances of stock collapse. However, it is this variation that has allowed collapsed stocks to rebuild from extremely low population sizes. For example, unusually high recruitment survival of haddock in 1975 and yellowtail flounder in 1987 (Fig. 2.9) resulted in significant but temporary stock rebuilding, as these year classes were rapidly fished out. In the past, these unusual events of high recruitment survival at low spawning stock sizes had been interpreted as evidence that environmental variation was the defining factor in year class strength, and that spawning stocks could be fished to very low levels without threatening the productive capacity of species. More complete consideration of the relationships between SSB and recruitment, however, has established that for most groundfish resources there is a higher probability of good recruitment at spawning biomasses above the median, and that good year classes are the product of good recruitment survival combined with sufficient spawning stock. Thus, the appropriateness of managing for high and stable spawning stocks as a necessary element of fishery management goals for the New England groundfish resource is firmly established (Brodziak et al., 2000; Overholtz et al., 1987, 1999).

The suite of management measures currently in place has been sufficient to allow recovery of some components of the resource. The challenge for managers in the future is to extend rebuilding to other, less productive components of the resource. Given the complex biological and technical interactions among various resource species, extension of single-species concepts into an overall ecosystem perspective should allow consideration of the inevitable trade-offs between species. In the long run, conservative management of fishing capacity for vessels capable of switching target species, combined with appropriate uses of marine protected areas to preserve ecosystem function and nurseries for resource species, and improved gear designs are undoubtedly the basic elements of a sustainable fishing strategy for the New England groundfish resource—a strategy that has yet to be fully realized.

ACKNOWLEDGMENTS

This paper is dedicated to several individuals whose collective vision and determined hard work

forever changed the scientific view of fisheries on the northeast shelf, and put into place data collection schemes and programs of sufficient rigor that they have stood the test of time: William Herrington, William Royce, Herbert Graham, Robert Edwards, Richard Hennemuth and Marvin Grosslein. Generations of scientists to come will forever be in their debt. I also dedicate this review to the memory of Ellie Dorsey, a tireless worker for improved conservation of the region's fishery resources.

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Chapter III

Recent Trends in Anadromous Fishes

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> *The beauty of this Kennebec fishery was that freshwater and saltwater fish kept company. Thanks to the tides running up as far as forty miles, there were two universes of fish at every farm's foot. There used to be legions of [striped] bass. [Captain George] Weymouth saw great salmon jumping out of the river where Bath now stands. And there were sturgeons longer than a man. Sometimes they came right aboard the canoes where men were spearing them by torchlight, and upset the boat!*
>
> *-Robert P. Tristram Coffin, 1937*

INTRODUCTION

Anadromous fishes are born in freshwater, subsequently move into saltwater to grow, then return to freshwater to spawn. Because these fishes are dependent on diverse environments during different portions of their life cycle, they can be especially vulnerable to a variety of environmental changes. During their early life stages, these fishes are sensitive to deleterious alterations in freshwater. Later, when they pass through estuaries and into the marine environment, coastal pollution can affect survival. At maturity, habitat alterations, pollution, and commercial harvest can have profound impacts on spawning grounds. Therefore, not only are anadromous fishes subject to environmental and harvest pressures at sea, they encounter dams, pollution, urbanization impacts, and habitat changes in freshwater.

It is impossible to conclude how much anadromous fish habitat has been lost because we are uncertain of the original historical distribution of these species. However, the construction of dams beginning in 1798 denied access for Atlantic salmon (*Salmo salar*), American shad (*Alosa sapidissima*), alewives (*A. pseudoharengus*), blueback herring (*A. aestivalis*), Atlantic sturgeon (*Acipenser oxyrhynchus*), and shortnose sturgeon (*A. brevirostrum*), in particular, to most of their original habitat. Kimball and Stolte (1978) estimated that, by 1950, less than 2% of the original freshwater habitat was still accessible to Atlantic salmon in New England. With improvements in fish passage and dam removal, that percentage has risen to over 64% of the original habitat (USFWS and NMFS, 1995).

The National Marine Fisheries Service (NMFS) has concluded that, “Atlantic anadromous stocks have been heavily influenced by nonfishing human activities in the coastal zone. Damming of rivers preventing occupation of former spawning grounds was a major factor in the decline of Atlantic salmon, sturgeons, river herrings, and shad. Environmental contamination is implicated in the declines of several species” (NMFS, 1992). As a consequence, successful restoration and rehabilitation of most species of anadromous fishes will rely on improving the freshwater and estuarine environments through reduction of pollution, improvement of fish passage, and rehabilitation or protection of nursery and spawning habitats.

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*Note: Author is deceased.

Table 3.1. Anadromous fish species of the Massachusetts Bay Region[a], their relative abundance, numbers of streams, and aspects of freshwater residence. Further life history information can be found in Bigelow and Schroeder (1963), Murawski et al. (1980), Danie et al. (1984), Mullen et al. (1986), Weiss-Glanz et al. (1986), and Jury et al. (1994).

Species	Relative abundance	Minimum number of streams	Freshwater residence
Atlantic salmon	Rare; extirpated; restoration on Merrimack River	1	Upper portions of river; two years freshwater residence by juveniles
Striped bass	Seasonally present in coastal waters and lower rivers in summer; Bay fish are from Hudson River and other areas	2 (non-spawning)	Enter Merrimack and Parker rivers from April to October; no spawning
Rainbow smelt	Seasonally common in spring; also landlocked populations in lakes	18	Enter coastal tributaries in spring; eggs deposited in lower sections of streams; juveniles abundant in estuaries
Sea-run brook trout	Rare, isolated populations	Unknown	Move between lower sections of streams and estuaries
Sea-run brown trout	Uncommon, few locations	Unknown	Exotic fish introduced to a few streams and estuaries; little information
Alewife	Very common	11	Merrimack and Charles rivers are principal runs; enter freshwater to spawn in late spring where there is access to lakes; adults return to sea; juveniles in freshwater until October
Blueback herring	Common	2	Similar to alewife, but do not migrate far upstream
American shad	Common	4	Merrimack River is principal run; enter freshwater to spawn in spring; use upper portions of watershed where not blocked; young in freshwater during summer
Shortnose sturgeon	Rare	1	Merrimack River; instream movements; spawn near some urban areas
Atlantic sturgeon	Uncommon	1	Juveniles enter Merrimack River in summer; adults and sub-adults in Bay; no river spawning, although fish once spawned up River 200 km
Pacific salmon species	Rare	Strays	Exotic; previous introductions in New Hampshire now largely gone; some remnant chinook salmon
Sea lamprey	Common	Unknown	Enter coastal streams for spawning, notably Merrimack River

[a] The Massachusetts Bay Region is here broadly defined as the area from the mouth of the Merrimack River south to Marshfield, Massachusetts (see USFWS, 1980). The minimum number of streams is the known number of streams of any size.

In this chapter, I will discuss the status of anadromous fish species in the Massachusetts Bay and the larger Gulf of Maine region and identify the causes of declines where known. These are summarized in Tables 3.1-3.2. After a historical review of the region's anadromous fisheries, I will identify the principal constraints to and opportunities for restoration and rehabilitation of these unique fish populations.

Table 3.2. Summary of the status of anadromous fish stocks of the Massachusetts Bay/North Shore region, in comparison to historical levels, and the principal factors affecting declines.

Species	Long-term trend[a]	Short-term trend	Factors involved
Atlantic salmon	Extirpated; undergoing restoration	Low returns	Dams, pollution, overfishing
Striped bass	Native population extirpated; declining numbers of migratory fish from non-Bay sources	Increasing numbers of migratory fish	Habitat destruction, dams,pollution, over fishing; now harvest reduction and stocking
Rainbow smelt	Declining	Variable, runs generally high in 1989 and 1994, depressed other years	Stream blockage, siltation, possibly pollution-decreased substrate quality, predation by aquatic birds
Sea-run brook trout	Declining	Unknown remnant populations	Urban development, habitat destruction
Sea-run brown trout	Introduced exotic species	Increased stocking and harvest	Unknown
Alewife	Increasing after general decline	Declines in 1993-1994	Habitat destruction dams, unknown factors
Blueback herring	Increasing	Declines in 1993-1994	Habitat destruction dams, unknown factors
American shad	Increasing	Declines in early 1990s; slight rebound	Habitat destruction dams, unknown factors in 1995
Shortnose sturgeon	Declining; Endangered Species List	Remnant populations	Overfishing, dams, pollution; now harvest restrictions
Atlantic sturgeon	Unknown	Stable or increasing	Overfishing, dams, pollution; now harvest restrictions
Pacific salmon species	Introduced non-native species, strays to Massachusetts	Declining, stocking now terminated; adults will decline	Unknown factors affecting survival at sea
Sea lamprey	Declining	Unknown	Dams

[a] Since 1980

Table 3.3. Status of U.S. Atlantic salmon populations by river (Table from NMFS, 1998).

River System	Population Status
Housatonic River	Extirpated[1]
Quinnipiac River	Extirpated
Hammonassett River	Extirpated
Connecticut River	Extirpated
Thames River	Extirpated
Pawcatuck River	Extirpated
Pawtuxet River	Extirpated
Blackstone River	Extirpated
Merrimack River	Extirpated
Lamprey River	Extirpated
Cocheco River	Extirpated
Salmon Falls River	Extirpated
Mousam River	Extirpated
Kennebunk River	Extirpated
Saco River	Extirpated
Presumpscot River	Extirpated
Royal River	Extirpated
Androscoggin River	Extirpated
Kennebec River	Candidate Species
Sheepscot River	Unique Stock
Pemaquid River	Extirpated
Medomak River	Extirpated
St. Georges River	Extirpated
Ducktrap River	Unique Stock
Little River	Extirpated
Passagassawaukeag River	Extirpated
Penobscot River	Candidate Species
Orland River	Extirpated
Union River	Extirpated
Tunk Stream	Candidate Species
Narragaugus River	Unique Stock
Pleasant River	Unique Stock
Indian River	Extirpated
Chandler River	Extirpated
Machias River	Unique Stock
East Machias River	Unique Stock
Orange	Extirpated
Hobart Stream	Extirpated
Dennys River	Unique Stock
Pennamaquan	Extirpated
Boyden Stream	Extirpated
St. Croix River	Candidate Species

[1] Extirpated status could result from complete blockage of the river or a small population size, both of which indicate that long term persistence is unlikely.

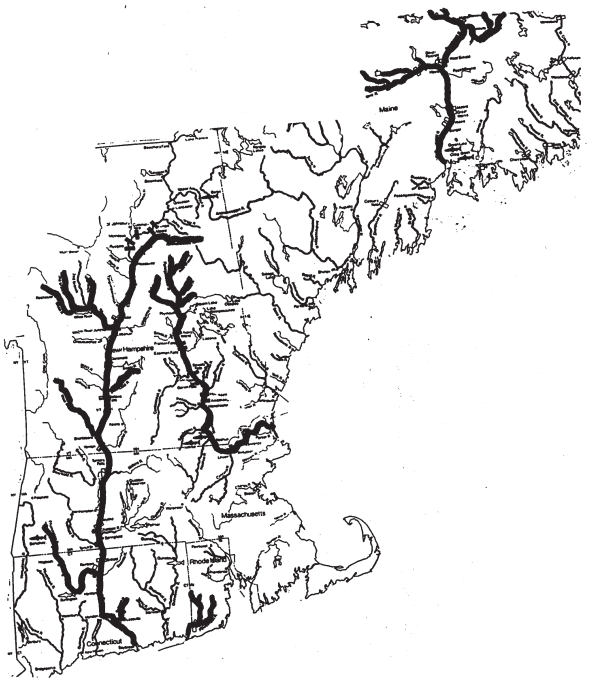

Figure 3.1. Former range of Atlantic salmon in the rivers of New England. Rivers in bold type probably had active salmon runs at the time the first European settlers arrived. Rivers highlighted with the widest line are currently undergoing restoration. (Map from International Atlantic Salmon Foundation, 1979).

Figure 3.2. The aboriginal distribution of Atlantic salmon in North America (map from Watt, 1988). The solid black area represents habitat where the salmon have been extirpated by about 1870. The Atlantic salmon stocks in the rivers flowing into Lake Ontario and Lake Champlain were probably largely landlocked.

Table 3.4. Historic and currently accessible Atlantic salmon habitat listed from south to north. One habitat/production unit = 100 m^2. (Table from NMFS, 1998).

New England Rivers	**Accessible Habitat Units**	**Historic Habitat Units**
a. Large Production Rivers (>10,000 production units)		
Connecticut River	89,250	262,500
Merrimack River	251	83,600
Saco River	12,540	30,692
Androscoggin River	3,177	71,060
Kennebec River	1,003	114,700
Penobscot River	102,744	108,680
Union River	8,360	11,202
St. Croix River	29,260	38,874
b. Small Production Rivers (<10,000 production units)		
Pawcatuck River	167	8,778
Kennebunk River	84	627
Sheepscot River	1,672	1,672
Ducktrap River	585	585
Passagassawaukeag River	167	418
Narragaugus River	585	585
Pleasant River	5,768	5,768
Machias River	269	502
Tunk Stream	6,939	6,939
East Machias River	2,174	2,174
Hobart Stream	84	84
Dennys River	2,090	2,090
Boyden Stream	84	84

SPECIES STATUS

ATLANTIC SALMON

Almost all runs of sea-run Atlantic salmon (*Salmo salar*) were eliminated in New England by the twentieth century, especially in southern New England (Table 3.3 and Figures 3.1-3.2). Salmon runs may once have extended as far south as New Jersey, but the major populations were found in the Connecticut and Merrimack watersheds of southern New England, and the Penobscot and Kennebec drainages of Maine. No one knows for sure how many salmon were present when the colonists first arrived, but the best guess based on available habitat is 300,000 adult salmon in New England waters, including 30,000 in the Merrimack (Stolte, 1981). The causes of the decline and extirpation of stocks have been well documented and attributed to dams and pollution (Table 3.4). As the species declined, commercial fishing accelerated the extirpation. The species was familiar to the first colonists (Stolte, 1981, 1994) and stocks have been exploited for three-and-a-half centuries. After a long period of fish absence, restoration programs began in earnest in the 1960s, after the construction of fish passage facilities and the enactment of several water quality laws.

Restoration success in the Connecticut and Merrimack rivers has been only marginal, while that in Maine rivers has been somewhat better. Despite decades of stocking in the Merrimack River, restoration of Atlantic salmon cannot yet be considered successful (Table 3.5). A peak run of 332 adult salmon was achieved in 1991, but runs only ranged from 23 to 248 in the previous decade, and dropped to just 61 fish in 1993 (Stolte, 1994), 21 in 1994, 34 in 1995, 76 in 1996, and 71 in 1997 (New England Salmon Association, 1995, 1996; United States Fish and Wildlife Service [USFWS] trap records). Returns in recent years (123 in 1998 and 192 in 1999) are still disappointing. Despite the stocking of large numbers of fry (e.g., 3.1 million in 1994), the chances of full restoration seem quite distant. Less than 0.0007% of the fry that are stocked return as adults.

The situation is only slightly better on the Connecticut River. Several hundred thousand smolts are stocked each year, along with several million fry (4 million in 1993, 6 million in 1994). Yet, the total returns for the past 10 years have been just over 2,500 salmon. That reflects an annual return rate of only 0.0004 to 0.4% from stocked smolts and 0.003 to 0.02% from stocked fry (Meyers, pers. comm.). Only 188 adult salmon returned in 1995, 260 in 1996, 199 in 1997, and 300 in 1998 despite the expenditure of more than $70 million on the program in the past three decades (Freeman, 1995; New England Salmon

Table 3.5. Estimated returns of Atlantic salmon to the Merrimack River, 1867-1999, based on trap counts. Data are taken from Stolte (1981), Rideout and Stolte (1988), the New England Salmon Association (1996) and trap records of the U.S. Fish and Wildlife Service.

Year	**Estimated number of returning grilse, 2-sea-year, and 3-sea-year fish**
Historical run size	17,880 (range of 8,940 to 26,820)
First attempts at restoration	
1867-1875	0
1882	166
1883	54
1884	330
1885	760
1886	588
1887	1,500
1888	1,927
1889	1,141
1890	1,796
1891	1,653
1892	3,062
1893	3,600
1894	929
1895	1,776
1896	1,034
1897	241
1898	16
Second attempts at restoration	
1975-1977	0
1978	13
1979	33
1980	53
1981	125
1982	23
1983	114
1984	115
1985	213
1986	103
1987	139
1988	65
1989	84
1990	248
1991	332
1992	199
1993	61
1994	21
1995	34
1996	76
1997	71
1998	123
1999	192
Recent returns	
1993	61
1994	21
1995	34
1996	72

Association, 1995).

Returns of adult salmon to waters in Rhode Island and New Hampshire have been minimal. Farther to the north, there has been more success in restoring Atlantic salmon to the Penobscot River, Maine. Yet, despite a peak run of 4,125 in 1986, recent numbers of adults returning to the river have shown declines (1,578 in 1991, 1,650 in 1993, 1,342 in 1995, 2,052 in 1996, 1,342 in 1997 and 1,210 in 1998. Only 969 adult salmon returned to the Penobscot River in 1999, and the total run for all Maine rivers was only 1,164 (trap records, Maine Atlantic Salmon Authority). Similar declines have been evident on many rivers, including those in eastern Canada and the runs in wild rivers of Downeast Maine. Clearly, some unknown factors are affecting survival of salmon.

The database for documenting returns of Atlantic salmon is good, but the causes of the low numbers of returning adults in recent years are not known. Many experts suspect factors in the marine environment are dictating return survival, possibly ocean warming along southern portions of oceanic habitat or even ocean cooling near Greenland (Friedland et al., 1993). But there are two other obvious factors that likely play a role in restoration success in southern New England waters. First, even though upstream fish passage facilities exist at the lower dams of the Connecticut and Merrimack rivers, and downstream passage is planned for the lower dams, moving up or down past any dam involves some level of mortality, even with the most efficient passage facilities. Thus, despite fish passage improvements, survival of Atlantic salmon in New England streams continues to be low compared to waters in eastern Canada, where there are generally fewer dams and higher fish populations. Second, the salmon brood stock used to initiate restoration was taken from the progeny of fish returning to the Penobscot River, Maine, a source quite distant from the Massachusetts locations where the native stocks were extirpated and the habitat altered. The Penobscot strain, in turn, is a hybrid, derived from a combination of non-extirpated fish returning to Downeast rivers of Maine and fish returning to Canadian rivers--because the native Penobscot strain was also extirpated. Thus, restoration efforts in southern New England rely on a hybridized fish whose relatives were adapted for life in waters much farther to the north; this is especially true for fish in the Connecticut River.

Because salmon in Massachusetts are in the southern margin of the historical range of the species, the task of restoration is especially difficult. Add to this the recent trends in global warming, and at least one scientist has predicted that "Global warming could ultimately make restorations on the southern edge of Atlantic salmon, the Merrimack and Connecticut, difficult or nearly impossible" (Bielak, 1994). How native Atlantic salmon of southern New England may have responded to even subtle warming trends is a moot point, as restoration now involves a hybridized, non-native stock. Return rates of salmon from the Connecticut and Merrimack rivers average only 12 and 27%, respectively, of those on the St. John River, New Brunswick, while those of Atlantic salmon from the Penobscot River, farther north, average 89% of the return percentages from the St. John River (USFWS and NMFS, 1995).

AMERICAN SHAD

Commercial landings of American shad (*Alosa sapidissima*) peaked in 1970 when about 3,000 mt were taken in northwestern Atlantic waters. But recent harvests have only been one-third of that level (NMFS, 1992). Statistics specific to New England are unreliable through the 18th and 19th centuries; however, landing records for the entire Atlantic coast suggest that 1896 was a peak year with catch figures about six times what they were in 1960. Historically, runs declined due to pollution and inadequate fish passage, but some of these declines likely were masked by the natural cyclical nature of year classes.

Except for the most recent run years, there have been modest success stories in southern New England, such as runs on the Connecticut River, but most populations in other rivers have been depressed, especially during 1993-1994. Most runs in Maine, for example, were eliminated due to impassable dams and pollution and are only now showing some successes in restoration in a few rivers, primarily due to stocked, transplanted fish.

Those shad that survive spawning, along with immature adults, generally migrate to the Bay of Fundy, and remain there during the summer and into the fall (Melvin et al., 1992). During winter months, shad from New England move into an area between Long Island and the mid-Atlantic coast. Thus, shad are influenced not only by conditions in freshwater but by conditions in several areas of the Gulf of Maine and southward as well.

Runs of American shad have generally increased in Massachusetts waters in the 1990s, a rehabilitation success story that has not been completely smooth. After gradually increasing until 1993, runs on the Merrimack River declined by about half each year in 1993 and 1994 before increasing again in 1995. Recent returns have been impressive (22,586 in 1997, 27,891 in 1998, and 56,465 in 1998), the highest three annual runs since records have been kept (USFWS trap records). Apparently, the 1993-4 decline was due to some unknown factor at sea, as similar declines were experienced in the Connecticut River stock, where shad runs decreased to record lows in 1994 and 1995 (B. Kynard, U.S. Geological Survey, Turners Falls, MA, pers. comm.). Only 300,000 shad returned to the Connecticut in 1995, less than

19% of the run of 1992, although returns in 1996-1998 have exceeded 600,000.

ALEWIVES AND BLUEBACK HERRING

Adult alewives (*Alosa pseudoharengus*) and blueback herring (*Alosa aestivalis*) (sometimes grouped together as "river herring") usually return to saltwater after spawning, and may spawn more than once. Individual stocks have been reduced due to pollution and dams that altered habitat and blocked access to spawning sites (Jury et al., 1994). The National Marine Fisheries Service (1992) considers alewife and blueback herring (river herring) stocks as variable, dependent on local conditions. Commercial catches for these river herring peaked in the 1960s along the northeastern coast, when 27,000 mt were harvested annually. In recent years, the harvest has only averaged 1,200 mt (NMFS, 1992).

River herring runs on the Merrimack River had been steadily increasing, until 1992, but have declined ever since. The 1999 run of 7,898 was only 2% of the size of the run in 1991, but has been an improvement over the historic low return of just 51 fish in 1966 (USFWS trap records). The recent years of poor returns have not been linked to any specific cause. There does not appear to be any in-river change that could account for this decline. Without a specific freshwater factor, all indicators point to some unknown factor at sea that is increasing mortality of northern Massachusetts stocks. Similar declines have been noted for runs in the Connecticut River, where spawning numbers of 410,000 blueback herring in 1991 declined to 12,000 in just seven years, despite a Juvenile Alosid Index prepared by the State of Connecticut that predicted record runs. The actual returns were half and one-quarter of the predicted runs over the two years, respectively (B. Kynard, pers. comm.).

RAINBOW SMELT

Although never experiencing the widespread extirpation of runs as have other anadromous species, the distribution of sea-run rainbow smelt (*Osmerus mordax*) in coastal rivers has been affected by natural and human-made obstructions, siltation, decline of substrate quality, poor water quality, and other unknown factors as well (Buckley, 1989). No single cause has been implicated. Rather, declines in individual streams are due to site-specific combinations of the above factors.

Runs of rainbow smelt are extremely variable, but the long-term trend indicates a decline of coastal populations in southern New England. In the past 15 years, only runs in 1989 and 1994 were considered good. In all other years since the early 1980s, numbers of returning adults have been extremely low, particularly in the years 1990-1993 (B. Chase, Massachusetts Division of Marine Fisheries, Salem, Massachusetts, pers. comm.). The cause of the declines is unknown but increased predation by aquatic birds, and spawning substrate degradation are strongly suspected. Populations of several species of coastal birds have increased dramatically in recent years (Krohn et al., 1995), and the deterioration of small coastal tributaries has been shown to reduce spawning potential (S. Chapman, Darling Marine Center, Walpole, Maine, pers. comm.). Surprisingly, runs have shown improvements in some urban rivers, tributaries to Massachusetts Bay, while runs are declining in less urbanized rivers with better water quality. The issue is unresolved, but may involve other undocumented factors as well, such as the success of Chesapeake Bay and Hudson River striped bass rehabilitation, which has led to the recent large increases in striped bass from those areas feeding in inshore waters and river mouths, particularly the Charles and Weymouth Fore rivers. Additional monitoring will be required to identify specific causes of declines, as they likely are a combination of factors.

Similar declines in interior waters of Massachusetts have been linked to acid precipitation and a resultant decrease in pH (B. Kynard, pers. comm.), and this may be a factor in anadromous runs of rainbow smelt as well, where low pH levels also have been recorded in coastal tributaries (Haines, 1987). Rainbow smelt do not utilize fish ladders and even modest amounts of woody debris in streams will block upstream passage. Thus, urban development and related land use activities in the lower portions of coastal streams can result in increased instream debris and can effectively block upstream access.

There also has been a noticeable decline in substrate quality in many coastal streams. Smelt eggs are adhesive and naturally stick to instream rocks, aquatic vegetation, and submerged branches. However, in recent years, there have been noticeable declines in aquatic vegetation and increases in algal growth on instream cover (B. Chase, Massachusetts Division of Marine Fisheries, Salem, Massachusetts, pers. comm.). Eggs do not adhere to unstable substrate, such as algae, and some reproductive potential may be lost. The widespread growth of algae may be linked to changes in water quality and water temperature.

Even a modest increase in angler effort in coastal rivers can affect rainbow smelt populations. Murawski and Cole (1978) used predictive models to show that populations could be severely impacted by increased angler harvest in some Massachusetts rivers. Massachusetts has had a modest enhancement program in the past where eggs were collected and transported to waters with depressed smelt populations. However, the technique has been temporarily suspended because of the lack of documented success of such practices. A Massachusetts Bay smelt monitoring program introduced eggs into an unaltered stream in 1995 to assess the suitability of egg transfers (B. Chase, pers. comm.). Enhancement of stocks, through transportation or other means, has high public appeal, but transfers have had only mixed success in New England, partly because the extreme variability of year-class strength can mask the success of rehabilitation techniques.

ATLANTIC AND SHORTNOSE STURGEON

The shortnose sturgeon (*Acipenser brevirostrum*) is on the Federal Endangered Species List and considerable research is underway in Massachusetts on spawning behavior and movements, and in Pennsylvania on fish culture. The species is relatively abundant in coastal waters of northern New England and preliminary estimates have been derived for the populations in the Kennebec River, Maine (T. Squiers, Maine Department of Marine Resources, Hallowell, Maine, pers. comm.). Atlantic sturgeon (*Acipenser oxyrhynchus*) are rare in northern New England (only seven were netted in the Kennebec River in 1994), but more abundant in southern New England. Both species, however, have been affected by a combination of urban development, destruction of spawning grounds, obstruction of fish passage, and over-exploitation. Atlantic sturgeon were once harvested in the Merrimack River in the nineteenth century, but reproducing populations no longer exist.

Atlantic sturgeon are considered quite abundant in the Massachusetts Bay area, arriving from Maine waters or the Hudson River (or offshore waters in Massachusetts Bay) in May, and remaining around the many islands of the Bay through the summer (B. Kynard, pers. comm.). Juvenile Atlantic sturgeon enter the lower Merrimack River in summer, then leave to go offshore in the fall, possibly joining adult Atlantic sturgeon in Massachusetts Bay. The North Shore and northern Massachusetts Bay coasts are apparently used as heavy forage areas for Atlantic sturgeon. Although population estimates are unavailable, biologists currently do not consider the species threatened in the southern Gulf of Maine, as they are in the northern Gulf region (Kieffer and Kynard, 1996; B. Kynard, pers. comm.).

Shortnose sturgeon are much less abundant in the North Shore/southern Gulf of Maine region than in the northern Gulf of Maine. A remnant population of less than 100 shortnose sturgeon exists in the Merrimack River, moving up and down the river (B. Kynard, pers. comm.). A few fish are encountered in the Connecticut River. One area of concern is that preferred spawning areas in the Merrimack River are primarily in heavily urbanized sections of the river, such as the concrete-lined sections in downtown Haverhill (Kieffer and Kynard, 1996). As a consequence, the areas needing particular protection are often the areas most susceptible to impacts by humans.

STRIPED BASS

> *I myself at the turn of the tyde have seene*
> *such multitudes pass out of a pounde that*
> *it seemed to me that one mighte go over*
> *their backs drishod.*
> *-Captain John Smith, 1622*

Harvested since the arrival of the first colonists to New England, the striped bass (*Morone saxatilis*) was largely extirpated from most rivers of New England, due directly to habitat destruction, pollution, dams, and overfishing. Each has played a major role in the extirpation of native stocks, but dams and overfishing were probably most responsible (Cole, 1978; Moring, 1986; Squiers, 1988). Such declines were not limited to New England, but have been present throughout the range of striped bass. By the early 1980s, larval fish counts in Chesapeake Bay and elsewhere were the lowest on record. However, the success of several measures in the 1990s has led to a remarkable recovery of Hudson River and Chesapeake Bay stocks.

Unlike the restoration techniques employed for Atlantic salmon and American shad, providing fish passage is not the answer. Striped bass will not use fish ladders, so river systems with dams, however efficient the fish passage facilities, effectively block upstream distribution of striped bass. Thus, as long as dams remain in place on lower portions of New England rivers, full restoration of striped bass to historical spawning grounds will never be achieved. On a positive note, the removal of the Edwards Dam on the Kennebec River at Augusta, Maine in 1999 provided striped bass and other anadromous fishes access to the river upstream to Waterville for the first time since the 1830s. Full restoration of the species to the lower portions of rivers also depends on remediating any water pollution problems, which is a process that effectively began in the 1960s.

Individuals migrating in coastal waters from Connecticut northward to Maine in summer are often derived from populations in the Hudson River and Chesapeake Bay (Flagg and Squiers, 1991). Although some striped bass tagged in Canadian waters have been recaptured in New England (Rulifson and Dadswell, 1995), striped bass generally move northward from the Hudson River and Chesapeake Bay in summer, as water temperatures increase. They travel singularly or in groups, feeding on inshore fishes and sea worms, and entering the lower portions of rivers in the Massachusetts Bay region. Thus, recent increased numbers of adult and immature striped bass in Bay waters is a reflection of restoration success elsewhere, not in Massachusetts, as native populations here have been extirpated.

Populations along the east coast declined rapidly in the 1980s, but restrictions on sport and commercial catches, habitat improvement, and reduced pollution have recently led to a marked increase in population numbers. Larval counts and numbers of adults have shown remarkable improvement in the last two years in Maryland waters, and increased numbers of migrating adults have been subjectively noted in Massachusetts coastal regions and river mouths as well (B. Kynard, pers. comm.). It is difficult to conclude which rehabilitation technique has been responsible for the recent success and the subsequent increased number of fish entering waters of coastal Massachusetts, as all techniques were initiated simultaneously. However, it is likely that Federal pressure to reduce harvest by 50% had significant influence.

There are more striped bass today in the lower Merrimack and Connecticut rivers and along the North Shore of Massachusetts than were present even a few years ago (former National Biological Service, National Marine Fisheries Service, unpub. records). Since there has been no direct evidence of natural reproduction in Massachusetts waters for decades, the striped bass present along the Massachusetts Bay coastline in summer must originate elsewhere. Tagging studies indicate that Hudson River striped bass are the ones most likely captured by sport anglers in New England, although some scientists suspect that Chesapeake Bay stripers visit the area as well.

Most New England states have some level of restoration program for striped bass. Maine has been stocking juveniles into the Kennebec River, obtained from hatcheries in New York, for the past decade. Until recently, striped bass captured along the Maine coast in summer were the product of fish born elsewhere, as is the case with Massachusetts. However, there has been evidence of natural reproduction every year since 1987 in the Kennebec River and, most recently, in the Sheepscot River (L. Flagg, Maine Department of Marine Resources, Hallowell, Maine, pers. comm.). Thus, some larger striped bass encountered in the summer from Cape Cod to Maine may be the offspring of fish naturally produced in Maine.

There has been no direct evidence of natural reproduction in Massachusetts waters for decades. Re-establishing actual spawning runs in Massachusetts waters will require stocking of spring yearling fish.

SEA-RUN BROOK TROUT AND SEA-RUN BROWN TROUT

Localized populations of wild sea-run brook trout (*Salvelinus fontinalis*) are found in coastal streams of New England and eastern Canada. Such populations are isolated in New England and more common—even abundant—northward into rivers of eastern Canada in areas where stream pH is not low. Some information is available on Gulf of Maine populations and at least two states (Massachusetts, Maine) have considered modest management programs.

In all likelihood, sea-run brook trout populations are more common than long assumed. Runs in eastern Maine (Ritzi, 1953) and southern Maine (M. Dionne, Wells National Estuarine Research Reserve, pers. comm.) have survived largely because of minimal urban development and pollution and limited angling pressure. The surviving native populations are in isolated, little-developed coastal streams. Because of the popularity of this anadromous form in eastern Canada, fisheries managers in Maine consider coastal runs to be candidates for new and expanded fisheries (R. Owen, former Commissioner, Maine Department of Inland Fisheries and Wildlife Augusta, Maine, pers. comm.). Thus, increased exploitation of these wild runs is probable in the future. Remnant runs on Cape Cod have co-existed with human populations since colonization by the first Europeans. Management strategies in Massachusetts today include stocking of alternative species, such as brown trout, *Salmo trutta*, to lessen fishing pressure on these unique wild stocks (Bergin, 1985).

The brown trout is a species introduced from Europe, and both the true sea-run form and the non sea-run form have been introduced into lower rivers and estuaries of New England states. Several significant coastal sport fisheries have developed, particularly in major rivers where pollution has been reduced. Although brown trout are exotic fishes, they have been established in waters of North America for over 100 years and represent an increasing resource opportunity. There is almost no information on sea-run brown trout after their release into coastal rivers except that they experience rapid growth in summer months in coastal estuaries.

PACIFIC SALMON

All five species of Pacific salmon (*Oncorhynchus* spp.) have been stocked in New England waters, but are not native here and natural spawning has generally not been documented. Three species also have been reared in coastal aquaculture operations in the past twenty years (pink salmon, *O. gorbuscha*; chum salmon, *O. keta*; and coho salmon, *O. kisutch*). In recent years, New Hampshire has had a significant program of stocking coho and chinook salmon (*O. tschawytscha*) to create coastal sportfisheries, but stocking of coho salmon was terminated in favor of chinook salmon. Stocking of the latter species was terminated in December 1993 because of low returns and encouraging, early returns from stocked Atlantic salmon.

Pacific salmon are non-native species and the potential for competition with native freshwater and anadromous fishes has never been fully explored. However, numbers introduced into New England waters have decreased dramatically. Chinook salmon stocked in 1993 will continue to be encountered in southern Gulf of Maine waters for several years, then will disappear if natural reproduction does not occur. Ocean ranching operations in southern Maine in the early 1980s for pink salmon and chum salmon were largely unsuccessful, apparently due to inadequate water temperatures. Expansion of the species and associated fisheries into Massachusetts seems unlikely, particularly with the current emphasis on restoration of Atlantic salmon.

SEA LAMPREY

The sea lamprey (*Petromyzon marinus*) has been commercially important in the past. The fish was used by Native Americans for centuries and was taken commercially in large numbers from the Merrimack and Connecticut rivers in colonial times. Runs declined rapidly due to the construction of dams, especially on the Merrimack River (Bigelow and Schroeder, 1963). Currently, runs are stable, although lower than in historical times, yet harvest and consumption are almost non-existent. For unexplained reasons, the trap count of sea lampreys tripled on the Connecticut River in 1998, the highest numbers on record, but numbers in other years have remained steady. There are, however, efforts underway to develop a market for lamprey skins to be marketed in Asia for the creation of purses, wallets, and other products, much in the fashion of wolffish skin and shark skin.

HISTORICAL TRENDS IN NEW ENGLAND

> *Salmon, shad and alewives were formerly abundant here (the Merrimack River), and taken in weirs by the Indians, who taught this method to the whites, by whom they were used as food and manure, until the dam and afterward the canal at Billerica, and the factories at Lowell, put an end to their migrations hitherward; though it is thought that a few more enterprising shad may still be seen.* *-Henry David Thoreau, 1849*

Although there is some question as to the magnitude of runs of anadromous fishes before the Little Ice Age (ca. 1450-1800 A.D.), particularly with Atlantic salmon (Carlson, 1988), Native Americans surely utilized several species of anadromous fishes. Atlantic salmon were plentiful at the time of the first European settlements in New England, although no one can accurately assess the sizes of runs. Stolte (1986) estimated 300,000 adult Atlantic salmon, based on habitat availability. The construction of a dam 160 km upstream from the mouth of the Connecticut River in 1798 marked the beginning of a decades-long extirpation process of anadromous fish runs in New England. Locks and canals on the Merrimack River started to appear at the end of the eighteenth century and the construction of a dam near Bristol, New Hampshire, blocked the upstream passage of fishes in 1820 (Stolte, 1981). Whatever the specific local situation, American shad, Atlantic salmon, alewives, blueback herring, striped bass, and rainbow smelt were all declining in southern New England by 1870 (Bowen, 1970; Moring, 1986). The causes were primarily the impassable dams located at numerous locations along the Connecticut, Merrimack, and other major rivers of New England, and the heavy pollution near towns and mills. The first dam on the Connecticut River was constructed in 1798 at Turners Falls, Massachusetts. It was 16 feet high and impassable to migrating fish. Others soon followed on the Connecticut, Merrimack, Kennebec, Penobscot, and other rivers of New England.

Probably the first species to receive supplemental stocking to reverse the declining runs was the American shad. Over 200 million eggs were artificially hatched in northeastern states during a five-year period, 1866-1871 (Bowen, 1970). By 1866, runs of Atlantic salmon were severely depleted in southern New England and eggs were brought from the Miramichi River, New Brunswick, and implanted in gravel in the Merrimack River. Several states were actively purchasing Atlantic salmon eggs in the late 1860s to booster declining stocks of salmon.

The first salmon hatchery in the United States was at Craig Brook, near Orland, Maine; it distributed up to six million fry in the early 1870s to states as far south as New Jersey (Bowen, 1970). These efforts continued until the twentieth century, when the once abundant runs of Atlantic salmon in Maine began to suffer a similar fate to those in Massachusetts, Connecticut, and Rhode Island. Eventually, there were few brood stock available to supply eggs or fry. By the early decades of the twentieth century, runs of Atlantic salmon were extirpated from all the rivers of New England, except for several smaller streams in Downeast and central Maine (Moring et al., 1995).

In concert with the declines of Atlantic salmon, rainbow smelt were similarly declining due to blocked passage, pollution, and habitat destruction in lower rivers. As early as 1874, regulations were enacted to limit commercial catches (Murawski and Cole, 1978). Striped bass were eliminated from many rivers of New England as early as the 1830s, due to dams and pollution, while runs of American shad, blueback herring, and sea-run alewives declined due to fish passage problems (many runs in northern Maine were even extirpated). Unregulated commercial harvest of Atlantic sturgeon in the late nineteenth century led to depletions and extirpations of populations that were fished until they were no longer economically profitable.

Most restoration and rehabilitation efforts in the Gulf of Maine region have occurred since 1960. The major focus has been on the construction of fish passage facilities and the improvement of water quality in rivers. Once the issues of dams and water pollution were addressed in individual rivers, stocking and harvest regulation could proceed with some possibility of management success. Many New England rivers still have dams and pollution issues that affect fish populations, so the process of rehabilitation of fish runs is of necessity an ongoing process. Although population levels of all anadromous species are well below historical numbers, American shad and blueback herring programs have shown the most success in the Massachusetts Bay/southern Gulf of Maine region.

CURRENT AND FUTURE CONSTRAINTS

OBSTRUCTIONS AND FISH PASSAGE

By their very nature, anadromous fishes must descend rivers, later ascend rivers and, in some cases, repeat the process more than once. Downstream passage mortality associated with dams varies with fish passage design, but has reached 62-82% for American shad and blueback herring on the Connecticut River (Taylor and Kynard, 1985) and with similar values on other northeastern waters (DuBois and Gloss, 1993). More recent studies have estimated much lower mortalities, due to higher recapture rates and lower mortalities in control groups (Mathur et al., 1994). Mortality rates for passing Atlantic salmon have ranged from 9-23% in waters of Massachusetts, New Hampshire, and Maine (Stier and Kynard, 1986; Moring, 1993).

Stream blockage is likely one of the causes of the decline of rainbow smelt in the past decade and has certainly led to declines in striped bass. As these two species do not utilize fish ladders, re-establishing historical levels of these fishes will require removal of some dams (such as recently occurred on the Kennebec River) or refinement of fish lift technology.

WATER QUALITY

Freshwater tributaries of the Massachusetts Bay/North Shore area continue to be sources of pollutants (Brown, 1987), although water quality is less of a constraint for re-establishing historical levels of anadromous fishes than was the case decades ago. Since the passage of a number of federal and state laws in the 1960s and 70s that promoted water quality improvements, many rivers of New England have shown substantial improvement from the heavily polluted condition of earlier times. Nevertheless, in 1995, more than half of the lower Charles River, between Newton and Waltham, was covered with aquatic vegetation--primarily water chestnut. Such dense vegetation can affect water quality and even block fish passage. The principal cause was the heavy nutrient load to the river from human sources, such as lawn fertilizers, septic tank input, and raw sewage (Allen, 1995). Some of this material eventually reaches Massachusetts Bay and influences water quality as anadromous fishes enter coastal streams. Levels of PCB and mercury continue to be accumulated by fish, necessitating fish consumption health advisories for humans (see Chapter 4 by Thurberg and Gould). There is no evidence that contaminants are today adversely affecting the health of adult anadromous fish in New England. However, there have been recent studies with freshwater and estuarine fishes that indicate fish subjected to higher levels of mercury in the environment may exhibit behavioral changes or reduced reproductive success (Wiener and Spry, 1995; T. Haines, U.S. Geological Survey, Orono, Maine, pers. comm.). Specifically, avoidance of predators may be impaired and hatching success reduced. This has been untested for anadromous species of North America, but such subtle impacts could result in lower survival of fish in streams with higher mercury content. It is likely that some water quality factors continue to influence spawning runs and spawning success of rainbow smelt and may affect other species as well, such as PCB problems affecting striped bass in the Hudson River.

LAND USE PRACTICES

Several types of land use activities can disturb aquatic ecosystems and potentially can have a significant impact on anadromous fish during their freshwater residence. Moring et al. (1994) summarized the physico-chemical and biological changes that can occur when forest canopies are opened, riparian vegetation is removed, stream banks are altered, or roads are constructed. These consequences include increased sedimentation, excess particulates in gravel, increased streamflow, increased temperature, decreased dissolved oxygen, decreased insect drift, decreased fish populations, losses of nutrients from watersheds, and ecological shifts in energy input and fish and macroinvertebrate species diversity.

Such disturbances occur from agricultural use, road construction, logging, and urban development, and can directly affect nursery and spawning habitat of anadromous fishes. An extensive review of the biological consequences of such land use practices indicates few definitive studies in New England (Moring and Finlayson, 1996), except for the Hubbard Brook studies in New Hampshire (e.g. Likens et al., 1970) and several logging studies in northern Maine (Moring et al., 1994). There have been no studies in New England that directly link such land disturbance activities to losses of anadromous fishes. However, urbanization along coastal tributaries and logging in upper watershed tributaries of the Merrimack River likely influence different stages of anadromous fishes.

EXPLOITATION

Overexploitation, primarily by commercial fishers, has contributed to the decline in striped bass, Atlantic salmon, and Atlantic sturgeon. Maryland is the major spawning area for striped bass that migrate along much of the northeast coast and the state where the bulk of the commercial catch has been centered. In the 1960s and 1970s, the total catch of stripers by commercial and recreational fishers in Maryland was roughly the same, thus both probably contributed to the collapse that ensued. Once the stock collapsed, however the sport fishery dropped to less than 20% of the commercial catch while the commercial fishery continued their harvest until drastic management measures were imposed. In New England, commercial fishing for stripers has traditionally comprised a smaller percentage of the catch than that of recreational anglers so its impact has probably been less.

For Atlantic salmon, Maine has long had a recreational fishery which was finally banned in 2000. Now that the days of river trapping (19th and early 20th centuries) is over, the major harvest of these fish is by commercial fishers on the high seas beyond US territorial waters. For sturgeon, there is currently little take these days, but previous sport and commercial fishers were responsible for limited harvests.

More effective fisheries management plans will minimize the effect of harvest on anadromous species, particularly numbers of spawners. Two recent management actions show how this can benefit these fishes. Federally-mandated increases in the minimum size limit for striped bass have apparently achieved the desired effect of reducing harvest by 50%. Maine (the only state that always had some level of sportfishery for Atlantic salmon, until 1994, with catch-and-release from 1995-1999) reduced the sportfishing-induced mortality from 20% of the run to zero. Two other management actions just adopted will hopefully have positive effects on anadromous fish populations. Recent negotiations with Canada and the buy-out (at least temporarily) of the West Greenland fishery for Atlantic salmon may prove fruitful in the next several years if higher numbers of adults survive ocean fisheries and return to Massachusetts waters to spawn. A new fisheries management plan for shad and river herring has just been adopted by the Atlantic States Marine Fisheries Commission in response to declining runs of those species.

In addition to fisheries management plans, the exploitation of some anadromous species is restricted under other environmental regulations. The shortnose sturgeon is protected under the Federal Endangered Species Act. The Atlantic sturgeon may be listed soon.

COASTAL AQUACULTURE

In recent years, Atlantic salmon, rainbow-steelhead hybrids (*Oncorhynchus mykiss*), and Arctic char (*Salvelinus alpinus*) have been reared in offshore cages to meet market demands for salmon and trout. At present, most such activity is clustered around the Maine-New Brunswick border, although aquaculture operations have been established from Cape Cod northward at various times in the past. The major biological concern is that escapees from these cages are now entering U.S. rivers. A storm in fall 1994 resulted in the escape of thousands of Atlantic salmon, many of which were subsequently encountered in coastal rivers of eastern Maine. Generally, 10% of caged salmon are likely to disappear from cages during rearing (Moring, 1989). If they originate from non-native sources, these escapees can and do hybridize with native stocks, thus altering the genetic components of salmon used in restoration. Even though the stock of Atlantic salmon that is used in restoration of the Merrimack River is non-native, selection is favoring an artificial "site-specific" stock that does return and reproduce. Thus, genetic dilution of such fish could further reduce the possibility of restoration. Another threat is represented by the recent outbreaks of two diseases in both cultured and wild stocks of Atlantic salmon of eastern Maine and New Brunswick. The cultured stocks may be acting as reservoirs of diseases that get passed on to the wild stocks.

SYNERGISTIC CONSIDERATIONS

Increasing population numbers of anadromous fishes is a difficult management premise because no single species can be managed in a vacuum. For example, as striped bass numbers increase, more of these predators will be entering the lower portions of New England rivers. The success of their feeding in these habitats will depend, among other things, on the success of programs to increase runs of Atlantic salmon and American shad. Survival of Atlantic salmon smolts, rainbow smelt, and alewives, in turn will be influenced by the breeding success of federally-protected aquatic birds, such as double-crested cormorants (*Phalacrocorax auritius*), the popularity of new sea-run brown trout sportfisheries (potential predators of smolts and prey for striped bass), and the success of striped bass restoration programs, as well as fishing and non-fishing human activities. Thus, rehabilitation programs must be managed from a broader perspective.

The interaction of anadromous fishes and other forms of wildlife are most obvious with respect to aquatic birds. Federal protection of double-crested cormorants and other species since 1916 have recently led to exponential increases in coastal breeding populations in Massachusetts (Krohn et al., 1995). The number of breeding pairs of double-crested cormorants in Massachusetts has more than tripled in less than 15 years, from 1,760 to 7,000, and the number of bird colonies has almost doubled during that time (Krohn et al., 1995). These birds are known predators of such anadromous species as rainbow smelt and Atlantic salmon smolts. As the bird populations have increased, pressure on anadromous stocks also increased. A recent estimate concluded that over seven percent of the Atlantic salmon smolts in the Penobscot River are consumed by cormorants in the spring as the fish migrate downstream (Blackwell, 1996). It is difficult to reach conclusions on a New England-regional basis as cormorant predation varies locally, dependent upon the presence of dams and breeding populations.

Although cormorants have been the birds most studied as salmon predators, at least six other bird species are known predators on Atlantic salmon smolts in New England and eastern Canada (Moring et al., 1998). As cormorant populations have increased (or in some cases stabilized), other birds, such as terns, have concurrently declined. Thus, the overall impact of bird predation on salmon and other fish species still needs to be assessed.

Links between Massachusetts anadromous fishes and inshore and offshore marine fishes are less clear. Successful restoration of Atlantic salmon, for example, may have been hampered in the early 1980s by substantially increased harvests of capelin (*Mallotus villosus*) on the high seas. This osmerid is the favorite food of Atlantic salmon in offshore waters.

SUMMARY OF ISSUES FACING ANADROMOUS FISHES

Anadromous fishes, by their very nature, are influenced by conditions in freshwater as well as at sea. Stocks of all anadromous fishes in the Massachusetts Bay region have declined from historical levels, principally due to dams, habitat alterations and pollution. To a lesser extent, overfishing on declining stocks also has played a role.

Initially, impassable dams blocked upstream passage of anadromous fishes, thus preventing fishes from reaching most spawning grounds. The first types of fish ladders and lifts were quite inefficient, but state-of-the-art designs of today still involve some mortality in passage. Downstream fish passage has lagged behind concerns for upstream passage, yet this is equally important. Mortality of downstream-migrating juvenile and adult fishes can be significant due to mortality from turbines and predation, but recent advances with bypass systems, especially with Atlantic salmon, have shown promising results. Water level changes due to competing demands for surface waters also may be an important factor, but this needs more study.

A significant portion of the habitat once utilized by anadromous fishes is no longer available. Thus, the reproductive and nursery carrying capacity of New England freshwaters is no longer as high as it was in Colonial times due to blocked passage and land disturbance activities, such as agriculture and logging.

Since the mid 1800s, pollutants have continued to influence the freshwater life cycles of anadromous fishes. Impaired water quality together with blocked fish passage was a major cause of declines and extirpation of anadromous stocks. Mill and tannery wastes, sewage effluents, heavy metals (particularly from industrial plants and paper companies), and sawdust and pulp from sawmills and lumber companies have all entered rivers tributary to the Gulf of Maine.

The most severe pollution occurs in coastal waters, obviously because these areas are adjacent to the land-based and discharge sources of pollutants. Anadromous fishes can be severely impacted by heavy metals, pesticides, hydrocarbons, and effluents because they pass between the coastal waters (as well as polluted rivers) in their migrations between freshwater and saltwater. In addition, as Thurberg and Gould summarized in Chapter 4, many pollutants have more pronounced effects on immature stages of fishes. The majority of species of anadromous fishes in New England do not travel far from shore; larvae and juveniles in the inshore, more heavily-polluted waters are faced with the consequences of pollution. Although pollution is significantly lower in the twenty-first century, levels of heavy metals, PCBs, and acid precipitation still may be adversely affecting anadromous fishes, especially through levels in sediments.

When a fish stock is declining, commercial or sport exploitation only accelerates the decline of anadromous fishes. It is an additive factor that can sometimes become critical in the presence of other contributors to stress and mortality. Anglers have taken a portion of returning stocks, yet this source has never been considered a primary factor in the overall declines of stocks. Commercial fisheries (first in freshwater, then in saltwater) have played a role in the decline of striped bass, Atlantic salmon, and Atlantic sturgeon. With current management efforts reducing the commercial and sport catches of striped bass and Atlantic salmon, harvest activities will likely have less negative influence on restoring anadromous stocks.

There are restoration or rehabilitation programs underway for almost all anadromous species in the Massachusetts Bay region, some modest and some quite extensive. The results have varied (Tables 3.1-3.3). Runs of American shad have shown the most improvement, although progress with river herring seems a distant prospect. Numbers of striped bass along the coast have increased recently, but not as a result of natural spawning in Massachusetts. Coastal runs of rainbow smelt appear to be declining over a broad geographical area; returns in 1994 were high, but this may be an anomaly. Native runs of sea-run brook trout appear to be stable. Atlantic salmon restoration has shown little success despite decades of heavy stocking of fry and smolts. Drought conditions in 1995 do not appear to have adversely affected juvenile Atlantic salmon in streams (R. Spencer, Maine Atlantic Salmon Commission, Bangor, Maine, pers. comm.), but winter survival may have been reduced as a consequence.

Restoration or improvement of anadromous fish populations in the region seems to be a function of many factors, each of which must be improved in order to show tangible results, particularly: improved fish passage, increased availability of appropriate habitat, improved water quality, and well managed sport and commercial harvest. The priority for restoration and rehabilitation of anadromous fish runs in the Massachusetts Bay region and other areas of northeastern United States should be to provide suitable habitat for species and access to that habitat. If that is accomplished, the impediments to restoration will largely be those of natural variability in ocean conditions.

RESEARCH NEEDS

Despite the long history of exploitation of anadromous fishes, and the lengthy database, several areas of research are necessary for future management, restoration, and rehabilitation. Runs of alewives, blueback herring, and Atlantic salmon are being influenced by some undocumented marine factors. In order to properly evaluate management and regulatory actions performed on behalf of these anadromous stocks, it is necessary to know why runs have declined despite improvements in water quality and fish passage. This requires research on ocean warming and cooling trends, commercial harvesting, and the dependency of these anadromous species on certain prey items, such as capelin.

Another area of concern is restoration of striped bass stocks to southern New England and elsewhere. The modest, but consistent, success of Maine's restoration program provides some evidence that important gains can be achieved with moderate financial investment. Factors influencing distribution and spawning success of striped bass need to be investigated. Third, the fragile existence of sea-run brook trout stocks from Cape Cod northward in the United States needs to be identified and managed. Our knowledge of the biology, seasonal movements, and habitat requirements of this anadromous form is primarily from waters of the Maritimes.

Fourth, broader-scope investigations need to be initiated that involve multi-species approaches. For example, striped bass restoration in waters containing sea-run brown trout or sea-run brook trout should be conducted after studies conclude that predator-prey influences are not self defeating. The success of one program should not be at the expense of another program, just because the two are conducted by separate agencies working in the same geographic area.

ACKNOWLEDGMENTS

I wish to thank Duncan McInnis, New Hampshire Fish and Game Department; Brad Chase and Randy Fairbanks, Massachusetts Division of Marine Fisheries; Richard Hartley, Massachusetts Division of Fish and Wildlife; Larry Stolte, Joe McKeon, and Ted Meyers, U.S. Fish and Wildlife Service; Norm Dubè and Randy Spencer, Maine Atlantic Salmon Commission; Lew Flagg and Tom Squiers, Maine Department of Marine Resources; and Boyd Kynard, U.S. Geological Survey, for background information and unpublished data.

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Chapter IV

Pollutant Effects upon Cod, Haddock, Pollock, and Flounder of the Inshore Fisheries of Massachusetts and Cape Cod Bays

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Nowhere is man's ecological naiveté
more evident than in his assumptions
about the capacity of the atmosphere,
soils, rivers, and oceans to absorb
pollution.
-Paul R. Ehrlich & Anne H. Ehrlich, 1970

INTRODUCTION

Of the three possible impacts being discussed in this volume (contamination, habitat alterations and overfishing), the effects of contaminants on demersal fish populations may well be the hardest to document. Catastrophic impacts of contaminants (e.g. oil and hazardous waste spills) exhibit readily apparent effects, but are relatively rare and are localized. Thus, they probably have little effect on overall fish population structure. The more important contaminant effects are from chronic, low level exposures. As with catastrophic effects, even these chronic impacts are localized geographically.

Contaminant impacts may be difficult to discern from habitat alterations and overfishing because the overall impact may be due to the combined effect of low levels of multiple contaminants (acting in an additive, synergistic or antagonistic manner). In addition, contaminant impacts may be masked by natural fluctuations in demersal fish populations due to such factors as weather, variable predation, gradual climate change, food availability, temperature, salinity, and interactions among environmental variables. Contaminants may be adding additional stress to populations already stressed by natural environmental fluctuations, anthropogenic habitat alterations and overfishing, thereby contributing to declines in fish populations in some areas.

This chapter focuses on the effects of four major pollutant categories (metals, petroleum hydrocarbons and PAHs, PCBs, and pesticides), each of which contain individual contaminants that are listed on the Environmental Protection Agency’s list of “priority pollutants” (Table 4.1). Toxicological impacts are assessed on several commercially important Northeast groundfish species including cod, haddock, pollock and four flounder species, of which cod and flounders have been the subject of considerable research. The European literature abounds in both experimental and field work with the Atlantic cod, *Gadus morhua*, from which the more relevant reports have been abstracted for inclusion here with American and Canadian papers. In the American Northeast, the winter flounder, *Pseudopleuronectes americanus*, has also been the focus of much experimental work, and is perhaps the most widely studied marine fish with respect to pollutant toxicity. It is widely distributed, is highly visible as an important commercial and

Table 4.1. The U.S. Environmental Protection Agency's Priority Pollution list (U.S. EPA, 1984). Contaminants marked with an asterisk have been routinely monitored since 1986 in sediments, marine mussel tissue and groundfish tissues by NOAA's National Status and Trends Program (Benthic Surveillance Project (NOAA/NS&T, 2000) and Mussel Watch Project; Lauenstein and Cantillo, 2000).

Metals
Ag*, As*, Be, Cd*, Cr*, Cu*, Hg*, Ni*, Pb*, Sb*, Se*, Tl, Zn*
PCBs*
all congeners and Arochlors
PAHs*
acenapthene*, acenaphthylene*, anthracene*, benzo(a)anthracene*, benzoperylene, benzo(a)pyrene*, benzo(b,k)fluroranthene*, chrysene*, dibenz(a,h)anthracene*, fluoranthene*, fluorene*, indeno-pyrene*, naphthalene*, phenanthrene*, pyrene*,
Pesticides
aldrin*, chlordane*, DDT*, DDE*, DDD*, dieldrin*, endosulphans, endrin, endrin aldehyde, heptachlor*, heptachlor epoxide*, hexachlorobenzene*, lindanes (BHC)*, toxaphene
Other halogenated compounds
bromoform, carbon tetrachloride, chlorinated benzenes, chlorinated naphthalenes, chloroalkylethers, chlorobenzene, chlorodibromomethane, chloroform, dichlorobenzenes, dichlorobenzidine, dichloroethane, dichlorobromomethane, dichloroethylenes, dichlorophenol, dichloropropanes, dichloropropene, dichloropropylene, dioxins (TCDD), haloethers, halomethanes, hexachlorobutadiene, hexachlorocyclohexanes, hexachlorocyclopentadiene, hexachloroethane, methylbromide, methylchloride, methylchlorophenol, methylenechloride, pentachlorolphenol, tetrachlorobenzene, tetrachloroethanes, tetrachloroethylene, tetrachlorophenol, trichlorinated ethanes, trichloethylene, trichlophenol, vinyl chloride
Phthalates
butylbenzylphthalate, dibutylphthalate, diethylphthalate, dimethylphthalate, diethylhexylphthalate, phthalate esters
Nitro-compounds
dinitrophenol, dinitrotoluene, dinitro-o-cresol, nitrobenzene, nitrophenols, nitrosamines, nitrosodibutylamine, nitrosodiethylamine, nitrosodimethylamine, nitrosodiphenylamine, nitrosopyrolidiae, nitrosodipropylamine
Other compounds
acrolein, acrylonitrile, asbestos, benzene, benzidine, cyanide dimethylphenol, diphenylhydrazine, ethers, ethylbenzene, isophorone, phenol, toluene

recreational fish, is easy to handle in the laboratory, and is readily collected from contaminated coastal areas. The number of field and laboratory reports of the effects of pollutants on the other five species (haddock, *Melanogrammus aeglefinus*; pollock, *Pollachius virens*; yellowtail flounder, *Pleuronectes ferrugineus*; American plaice, *Hippoglossoides platessoides*; windowpane flounder, *Scophthalmus aquosus*), whether examining cause-and-effects, circumstantial evidence, or even engaging in informed speculation, is sparse.

Our aim here is to summarize the various types of toxicological effects, both lethal and sublethal, that have been shown to occur at the organismal, tissue, cellular and subcellular levels of biological organization. This review follows the progression of research from the laboratory where toxicity was demonstrated through carefully controlled laboratory exposures, to the field where laboratory observations were confirmed at contaminated field sites. While much of the early laboratory work used high, often environmentally-unrealistic concentrations of contaminants, these studies have nevertheless been useful in determining the initiation, progression, and mechanisms involved in many adverse health effects observed in the field. Additional laboratory studies showed effects upon very sensitive early life stages as well as effects upon reproductive processes. The next critical stage, linking these laboratory, field, and reproductive biology studies to population effects, has been attempting to demonstrate that pollutants do have a measurable effect upon fish recruitment processes and thus effect population structure. This has been a difficult and elusive effort. The multitude of potential stressors in the environment (i.e., temperature, season, predators, parasitism, siltation, food availability, to name a few) in addition to the possible antagonistic and synergistic effects of multiple pollutants, both organic and inorganic, found in contaminated habitats, clouds the determination of specific effects of pollutants on fish populations as a whole. Mathematical models are being developed to consider the interactions of these multiple factors with contaminant effects, but the definitive answer is still some time away. This at least partially explains the lack of information on direct effects of pollutants on fish populations and communities.

CONTAMINANT DISTRIBUTION IN FINFISH

> *The quality of Massachusetts' marine environment is degraded. Pollution is impacting fish and shellfish health and impacting fisheries by rendering fish and shellfish in some locations unfit to eat or creating the unwarranted public perception that all seafood should be shunned.* *-MA DMF, 1985*

Various metals, pesticides, polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs) and other petroleum hydrocarbons have been measured in fish taken from the wild (Table 4.2). Elevated pollutant levels in fish tissue are, at the very least, an indication of habitat contamination. Temporal changes in contaminant body burdens have been documented, at least for some locales (e.g. Canadian *G. morhua*, Freeman and Uthe, 1984; Misra et al., 1988; Norwegian *G. morhua*, Skåre et al., 1985). Decreases in environmental exposure to these contaminants are generally reflected in declines in fish contaminant bioburdens over time.

METALS

Higher concentrations of metals in seawater and sediments have generally, although not universally, been correlated with higher concentrations of metals in various fish tissues. Geographical differences in concentrations of Zn, Cu, and Cd in Atlantic cod muscle, for example, reflected local

Table 4.2. Examples of contaminant bioaccumulation in demersal fish species from the western North Atlantic.

Species of Fish	Tissue Examined	Collection Site	Toxicant Measured	Reference
Atlantic cod	muscle	Baltic Sea	Cd, Cu, Zn	Perttila et al., 1982
Atlantic cod	muscle	Norway	Hg	Julshamm et al., 1982
Atlantic cod, American plaice	muscle	Newfoundland & Labrador	As	Kennedy, 1976
Atlantic cod	liver, muscle	Nova Scotia, New Brunswick	DDT	Sims et al., 1975
Atlantic cod	liver	Canada	organochlorines	Freeman and Uthe, 1984
Haddock	muscle	offshore	PCB	Capuzzo et al., 1987
Winter flounder	muscle	NY Bight, Hudson Shelf	various metals	Reid et al., 1982
Winter flounder	liver	Long Island Sound	Cu, Mn, Zn	Greig and Wenzloff, 1977
Winter flounder	liver	Boston Harbor	various metals	MacDonald, 1991
Winter flounder	muscle, liver	NY Bight	PAH, organochlorines	MacLeod et al., 1981
Winter flounder	whole body	New Bedford Harbor	PCB	Connolly, 1991
Winter flounder	liver	NY Bight	PCB	Reid et al., 1982
Winter flounder	gonad, liver	Long Island Sound	PCB	Greig and Sennefelder, 1987
Winter flounder	liver	New Bedford Harbor	PCB	Elskus et al., 1994
Windowpane flounder	liver	Long Island Sound	PCB, various metals	Greig et al., 1983
Windowpane flounder	muscle	Delaware Bay	Hg	Gerhart, 1977

concentrations in seawater (Perttila et al., 1982). American plaice collected off Newfoundland and Labrador had arsenic concentrations in muscle tissue that were similar to levels in sediments, much higher than those found in Atlantic cod, redfish (*Sebastes marinus*) and turbot (*Reinhardtius hippoglossoides*) from the same area, but lower than concentrations in the local shrimp upon which they prey (Kennedy, 1976).

In contrast to these two examples where metal body burdens correlated with environmental contamination, metal body burdens in winter flounder did not reflect the high metal levels found in the sediments of the more contaminated sites sampled from the New York Bight and Long Island Sound (e.g. Christiansen Basin, and the "Mudhole" 30 km SSE of the Basin) (Reid et al., 1982; Carmody et al., 1973). In a comparison of two sites in the west-to-east pollutant gradient in Long Island Sound, Hempstead Bay (westernmost) was considered heavily polluted compared to waters off Shoreham NY (mid-Sound) based on metals in sediment data, yet Cu, Mn, and Zn concentrations in livers of winter flounder were twice as great for Shoreham as for Hempstead (Greig and Wenzloff, 1977). The absence of a correlation between metal concentrations in tissues and the environment was also apparent in Gerhart's (1977) study of Hg in eleven fish species including winter flounder from Delaware Bay, and McDonald's (1991) review of data on winter flounder from within Boston Harbor. Winter flounder is a pollution-tolerant species that migrates between shallow and deeper waters seasonally, and therefore may not reflect contaminant levels from a single site.

LIPOPHILIC CONTAMINANTS

Lipophilic contaminants such as chlorinated pesticides (DDT, DDT analogs, chlordane, dieldrin, etc.), PCBs, PAHs and other petroleum-derived contaminants, have the ability to accumulate in the lipid-rich tissues of fish. For example, DDTs were measured in cod liver in quantities far exceeding the amount present in the flesh (Sims et al., 1975), a fact more important to the health of the fish than to the consumer of cod flesh.

In general, concentrations of these lipophilic contaminants in fish tend to correlate with contaminant levels in the environment. Elevated levels of polycyclic organochlorine pesticides were found in livers of male American plaice sampled in the North Sea near areas of major riverine input and other sources of anthropogenic pollution (Knickmeyer and Steinhart, 1990). Pesticide levels were also elevated in winter flounder from a tributary of Buzzards Bay MA, indicating significant levels of contamination in this area (Connolly, 1991). The highest concentrations of pesticide contaminants were found in coastal harbors and industrialized centers (MacLeod et al., 1981), whereas offshore areas had very low levels (Connolly, 1991). Similarly, levels of PCB congeners in liver samples of male and female Atlantic cod reflected a decreasing PCB pollution gradient away from the mouth of the Glomma, Norway's largest river (Marthinsen et al., 1991). Offshore haddock fisheries had very low levels of PCBs, presumably reflecting the low PCB exposures as distance from the mainland is increased (Capuzzo et al., 1987). PCB concentrations were higher in winter flounder gonad and liver samples from the more polluted sites in Long Island Sound (Greig and Sennefelder, 1987), although no relation was found between windowpane flounder liver concentrations of PCBs and a Long Island Sound pollution gradient (Greig et al., 1983).

Fish may obtain these organic contaminants not only via the soluble phase, but also through ingestion of contaminants present in or bound to prey items, as well as by contact or ingestion of contaminants bound to particles and sediments. The rate of uptake depends on the lipophilicity of the compound. Winter flounder exposed to crude oil-spiked sediments for 4 months accumulated more of the low molecular weight PAHs than the more lipophilic, higher molecular weight compounds (Hellou et al., 1995). The more lipophilic the contaminant, the more important is the ingestion route in uptake. Connolly (1991), for example, demonstrated, using a food-chain model, that uptake of soluble PCBs across the gill of winter flounder was exceeded by dietary uptake of PCBs. Contaminated prey species provided 80-95% of the PCB body burden. Assimilation efficiency of PCB declined from high values for trichlorophenyl to low values for the more highly-chlorinated (more lipophilic) congeners (Connolly, 1991).

Some of these lipophilic contaminants are quite persistent in the environment, as reflected in

Table 4.3. Toxicological effects and diseases possibly associated with concentrations of toxicants in fish. Potential correlations made between body tissue burdens with sublethal toxicological effects.

Species of Fish	Tissue Examined	Collection Site	Toxicant Measured	Associated Measure of Toxicity	Reference
Atlantic cod	liver, kidney, skeleton	Baltic Sea	Cd	skeletal deformities	Lang & Dethlefsen, 1987
Atlantic cod	liver	North Sea	PCB	ulcus syndrome	Stork, 1983
Atlantic cod, Haddock	liver	near oil rigs	assumed oil	elevated AHH activity	Davies et al., 1984
Atlantic cod	liver	Halifax Harbor	assumed, metals, organics	'fatty change' in liver	Freeman et al., 1981a,b
Atlantic cod	behavior	Norway, Sweden	ambient seawater	slower reaction times	Olofsson & Lindhal, 1979
Atlantic cod	gill	lab exposure	PAH	parasitic infect., less host resistance	Khan, 1990
Atlantic cod	liver, testes	lab exposure	PCB	reduced survival & reproduction	Sangalang et al., 1981
Yellowtail flounder	epidermis, skeleton	NY Bight	ambient seawater	fin erosion	Ziskowski et al., 1987
Yellowtail flounder	liver, skeleton	western N. Atlantic		parasitism, liver lesions, skeletal abnormalities	Despres-Patanjo et al., 1982; Murchelano et al., 1986
Winter & Summer flounder	intestine	Sandy Hook Bay, NJ	Hg	leucine absorption from gut	Farmanfarmaian et al., 1981
Winter flounder	epidermis, liver	NY Bight	PCB or DDT	increased liver size, fin erosion	Sherwood, 1982
Winter flounder	liver	New Bedford Hbr, Gaspee Pt, Fox Is.	PCB	increased P-450 protein (but no change in EROD)	Elskus et al., 1989
Winter flounder	gonad, larvae, embryos	Long Island Sound	PCB	lowered reproductive rates; sm larvae, increased embryonic abnormalities	Nelson et al., 1991
Winter flounder	eggs	New Bedford Hbr	PCB	smaller larvae	Black et al., 1988
Winter flounder	liver	Boston Harbor	PAH	liver tumor & other liver pathologies	Murchelano & Wolke 1991; Moore 1991, Smolowitz et al. 1989; McMahon et al. 1988a, b; Turgeon & O'Connor 1991
Winter flounder	liver	New Haven	ambient seawater	liver macrophage aggregates	Wolke et al., 1985
Winter flounder	epidermis, sktn	New York Bight	ambient seawater	fin erosion	Ziskowski et al., 1987
Winter flounder	embryos, eggs	Long Island Sound	ambient seawater	egg cytotoxicity, cell necrosis, chromosome damage, slower dev.	Perry et al., 1991
Winter flounder	liver, blood cells	New Haven CT, Long Island Sound, NY Bight	metals, PCBs	liver lesions, blood cell abnormalities, liver DNA damage, liver neoplasms	Greig & Wenzloff, 1977, Greig & Sennefelder, 1987, Gronlund et al., 1991.
Winter flounder	blood cells	NY Bight A, NJ, VA	ambient seawater	micronucleii in rbc	Hughes & Hebert, 1991;
Winter, windowpane flounder	blood cells	coastal mid-Atlantic	ambient seawater	erythrocyte mutations and micronucleii	Longwell et al. 1983
Winter flounder	blood cell ct	Boston Harbor, NE	ambient seawater	higher no. immature erythrocytes	Daniels & Gardner, 1989
Winter flounder	larvae	lwr Narragansett B.	ambient seawater	sm. yolk-sac larvae; lwr rate survival	Buckley et al., 1991
Winter flounder	liver, plasma, brain, muscle, epidermis	Boston Harbor	ambient seawater	reduced: hepatic, pectoral fin ascorbic acid conc; helpatic glycogen, lipid; plasma glucose; brain serotonin, norepinephrine, amino acid conc muscle	Carr et al., 1991
Winter flounder	liver	Halifax Harbor	ambient seawater	liver: necrogenic effects (hepatocyte basophilia, macrophage aggregation, hepatic epithelial vacuolation)	Tay et al., 1991
Winter flounder	liver	Long Island Sound	ambient seawater	histological ch., DNA alter. in liver	Gronlund et al., 1991
Winter flounder	whole body	lab exposure	DDT & dieldrin	reduced survival of embryos	Smith & Cole, 1973
Windowpane fl.	blood	Long Island Sound	ambient seawater	increased Hct & hemoglobin	Dawson, 1990
Windowpane, Yellowtail, & Winter flounder	epidermis, skeleton	New York Bight	ambient seawater	fin erosin	O'Connor, 1976
Windowpane flounder	eggs	Long Island Sound	ambient seawater	mitotic abnormalities in eggs	Longwell et al., 1992

temporal monitoring of fish tissues. Most organochlorine compounds in livers of cod caught off the east coast of Canada in 1980 showed no change in concentrations over the previous 8 years, with the exceptions of DDT and the PCB group, in which there was a general decline between 1972 and 1975, with no significant change thereafter (Freeman and Uthe, 1984). Following a 1972 DDT ban in Norway, cod liver samples showed decreasing concentrations of DDT; 10 years after the ban, the highest level of cod liver DDT was about one-third of the corresponding 1972 residue level (Skåre et al., 1985).

The persistence of lipophilic contaminants in fish themselves is related to several factors, including the contaminant's degree of lipophilicity, the size of the organism's fat stores, the organism's ability to metabolize the contaminant, and the organism's seasonal turnover of fat. When exposed to a radiolabeled PAH (benzo(a)pyrene) and a PCB congener for 24 h, for example, cod eggs and newly-hatched larvae accumulated both from the seawater. After being moved to uncontaminated seawater, the yolk-sac larvae showed no apparent elimination of the more lipophilic compound, the PCB, although there was a clear elimination of some benzo(a)pyrene (Solbakken et al., 1984). PCB levels in winter flounder liver were significantly correlated with body fat content, although fat content itself did not correlate with the contamination gradient (Reid et al., 1982). Haddock and cod from a Norwegian fjord had clearance rates for DDT and PCB that were slower than those found for wolffish, sea scorpion, a European wrasse, and lemon sole. While demonstrating interspecies differences in contaminant metabolism, the slower clearance of liver DDT in cod as compared to the other species examined may also be attributed to the substantially higher fat content of cod liver (Skåre et al., 1985).

Fat stores are closely tied to the fish's seasonal cycle of reproduction. PCB levels in female cod also varied seasonally (levels in Sept./Oct. greater than corresponding levels in June and Nov./Dec.), although no such effect was seen in male cod (Marthinsen et al., 1991). Similarly, work in Long Island Sound showed concentrations of PCB in winter flounder gonads to be highest (0.73 µg/g wet wt) in the months just before spawning, as compared to levels in other months (0.056-0.36 µg/g; Greig and Sennefelder, 1987). After spawning, PCB concentrations in gonads decreased to very low levels (0.03-0.08 µg/g). In contrast, Elskus et al. (1994) found that the content and concentration of PCB congeners in winter flounder liver taken from New Bedford Harbor fish, did not correlate with either sex or reproductive state. However, they cautioned that the high tissue concentrations of PCBs obtained at this extremely contaminated site may have obscured sex and reproductive condition differences.

TOXICOLOGICAL EFFECTS OF CONTAMINANTS ON FINFISH

> *As crude a weapon as the cave man's club, the chemical barrage has been hurled against the fabric of life - a fabric on the one hand delicate and destructible, on the other miraculously tough and resilient, and capable of striking back in unexpected ways.*
>
> *-Rachael Carson, 1962*

Bioaccumulation of contaminants does not necessarily imply that the contaminants are having an adverse effect on the organism. Nevertheless, there are numerous examples to show that both laboratory and field exposures to various metals, pesticides, PCBs, PAHs and other petroleum hydrocarbons do in fact elicit toxicity. In some cases, toxicity has also been linked to elevated tissue bioburdens of these contaminants (Table 4.3).

METALS

Each of the thirteen metals listed as EPA priority pollutants (Table 4.1) varies in toxicological potency and mode of action. This differential toxicity is best illustrated with data on winter flounder. The order of sublethal metal toxicity (2-5 mo, 10 µg/L metal) for adult winter flounder was $CdCl_2 > HgCl_2 > AgNO_3$ (Calabrese et al., 1977). Metal exposure either elevated (Hg) or depressed (Cd) gill respiration. Mercury, but not cadmium or silver, induced statistically significant hematological responses.

Cadmium, however, was the most potent inducer of transcription of the gene for metallothionein (Chan et al., 1989; Jessen-Eller and Crivello, 1998), the principal function of which appears to be the maintenance of homeostasis for the essential trace metals zinc and copper (Roesijadi and Robinson, 1995). Exposure of winter flounder to low concentrations of Cd induced several significant metabolic responses: (1) the ability of magnesium to promote enzyme-substrate binding in enzymes such as glucose-6-phosphate dehydrogenase (G6PDH) was impaired; (2) G6PDH was induced in gonad, heart, and skeletal muscle; (3) kidney tissue in particular showed an increased expenditure of energy (for synthesis of enzymes to maintain homeostasis under sublethal cadmium stress) and a loss of sensitivity to normal metabolic control (magnesium's enhancement of enzyme-substrate affinities); and (4) in the liver, glycolysis and shunt activity increased (Gould, 1977). These same phenomena were observed in mercury-exposed flounder but to a lesser extent, whereas silver-exposed flounder showed very little effect (Calabrese et al., 1975, 1977).

Metals initiate a number of additional toxicological effects in fish. Acute exposures to high metal concentrations as well as chronic exposures to much lower concentrations can elicit morphological changes (e.g. lesions in winter flounder and haddock olfactory organs following 18 h exposure to 500 µg/L Cu, Bodammer, 1981; abnormal swelling in windowpane flounder gill tissue following 2 mo exposure to 10 µg/L Hg, Pereira, 1988). Important metabolic enzymes can be inhibited (e.g. winter flounder Na, K-ATPase inhibition by Hg, organic Hg and organic As, Musch et al., 1990). Various metals can interfere with ion transport across membranes (Hg and methymercury can inhibit Na transport, Renfro et al., 1974; Farmanfarmaian et al., 1981; Dawson, 1990; Hg, organic Hg and organic As can inhibit K transport, Venglarik and Dawson, 1986; Musch et al., 1990). Methylmercury can increase the energy expenditure for transepithelial electrolyte transport in winter flounder gill and intestinal tissue (Schmidt-Neilson et al., 1977). Ionic Hg, organic Hg and organic As diminish the absorption of some amino acids in flounder species (e.g. leucine, Farmanfarmaian et al., 1981; tyrosine, Musch et al., 1990).

In general, early life stages of fish appear to be more susceptible to toxicity than either adults or membrane-protected embryos. LC_{50} concentrations (the concentration of a metal that is needed to kill 50% of a test population) are typically higher for adult fish and non-hatched embryos (U.S. EPA, 2000). Nevertheless, high concentrations of metals may effect hatching success. Concentrations of silver above 54 µg/L in a flow-through bioassay (18 d, 54 - 386 µg/L) produced greatly reduced percent viable hatch in winter flounder embryos and caused larval mortality (Klein-MacPhee et al., 1984). Embryos exposed to 180 and 386 µg/L hatched earlier than those exposed to lower concentrations, and many had physical abnormalities. Mean total length and mean yolk-sac volume of hatched larvae from the 386 µg/L silver exposure were significantly smaller than the lower Ag exposures. In contrast to Klein-MacPhee's experiments, the percent viable hatch of winter flounder embryos was not effected by silver (0-180 µg/L) but was decreased by cadmium (1,000 µg/L); addition of silver, however, decreased the toxic effect of cadmium on the viable hatch response (Voyer et al., 1982).

OIL

Much of the toxicological research on commercial fish species was spurred on by concerns over oil spills. Petroleum pollution has been shown to correlate with a variety of adverse behavioral, physiological and morphological parameters. Cod avoided concentrations of total petroleum hydrocarbons down to 50 µg/L, either in solution or as an emulsion (Bøhle, 1983). In the laboratory, detection thresholds for behavioral changes (snapping, darting, coughing, and restless swimming activity) in cod upon sudden exposure to oil compounds were observed at concentrations as low as 0.1-0.4 µg/L (Hellstrøm and Døving, 1983). Various fish species, chronically exposed to the water-soluble fraction (WSF) of crude petroleum or to oiled sediments, exhibited reduced growth, food consumption and body condition, depletion of energy stores, reduced gametogenesis and spermiation (release of mature sperm from the Sertoli cells), liver hypertrophy, splenic atrophy, impaired immune response, and morphological abnormalities such as gill hyperplasia, filament fusion, increased skin pigmentation, hepatic granulation, increased gall-bladder size, increased numbers of mucus-producing epithelial cells, capillary dilation, delayed spermatogenesis, and an increase of melanomacrophage centers in the spleen and kidney (Khan et al., 1981; Dey et al., 1983; Burton et al., 1984; Khan and Kiceniuk, 1984; Kiceniuk and Khan, 1987; Payne and Fancey, 1989). Chronic exposure of cod to crude oils, it was concluded, results in severely disabling lesions and reproductive impairment. Yet, mortality due to oil spills among large free-swimming fish has hardly ever been recorded, Bøhle (1983) concluded, because they can move away from contaminated areas.

Exposure to petroleum hydrocarbons typically induces the synthesis of several cellular enzymes, the Mixed Function Oxidase (MFO) enzymes, that metabolize some of these compounds. The most studied of these MFO enzymes are the group of Cytochrome P-450 enzymes (e.g. arylhydrocarbon hydrolase (AHH), ethoxyresorufin O-deethylase (EROD)). Several genetic isoforms of P-450 enzymes have been identified in western Atlantic fish (Wall and Crivello, 1998; Nelson, 2000). While P-450 enzymes are constituitive, elevated levels are biosynthesized "on demand" to catalyze the breakdown of many organic pollutants. Cod and haddock captured close to oil platforms showed significantly higher levels of AHH in their livers than did fish caught in areas well away from oil activity (Davies et al., 1984). These data were the first to indicate that oil in sediments around oil platforms may be bioavailable to fish in the area, probably via the food chain. Chronic exposure to petroleum WSF in the laboratory produced an oil-inducible MFO activity that was elevated 4 times higher in the liver and 3 times higher in the gills than in control fish (4 mo, 300-600 µg/L; Payne and Fancey, 1982). Winter flounder, exposed for 4 mo to oil-contaminated sediment that was weathered for a year had levels of hepatic Cytochrome P-450 that were seven times greater than unexposed controls; fish exposed to freshly oiled sediment (1 liter oil in 45 kg sand) exhibited a thirteen-fold P-450 induction rate (Payne and Fancey, 1982). However, examination of winter flounder collected from the site of an oil spill in 1984, and from a reference site showed that reliance on the measurement of liver MFO parameters alone could lead to false negatives in biological monitoring programs. The kidney provided statistical differences in elements of the MFO system between control and oil sites, whereas the liver did not (Payne et al., 1984). Nevertheless, the same research team, using oiled sediments under a controlled laboratory exposure, later found that biomarkers indicating exposure to oil were (in order of decreasing sensitivity): liver MFO activity, liver condition index (liver wt/total body wt), kidney MFO activity, spleen condition index, and muscle protein and water content. Liver lipid and glucose levels and condition indices for gut, kidney, testis and whole fish were not affected at any exposure level (Payne et al., 1988).

Induction of these enzymes in the liver typically precedes liver damage, although both liver hypertrophy and MFO induction may occur simultaneously. The level of exposure to oil at which liver hypertrophy continues to increase while MFO activity begins to decrease has been called the "point of crossover." It may represent the point at which the detoxication mechanism is overwhelmed (Hutt, 1985).

As indicated earlier, early life stages are often more susceptible to contaminants than adults. Yellowtail flounder eggs collected during the first three days following a gasoline spill near Falmouth MA had an 81% mortality rate (13 of 16 eggs died; Griswold, 1981). Many of the cod and pollock eggs collected shortly after the Argo Merchant oil spill (during the pollock spawning season) had oil adhering to the outer membrane and showed evidence of cytological abnormality of the embryo's cells and nuclear configurations indicative of cell death coupled with division arrest (Longwell, 1977). In the laboratory, cod eggs and larvae exposed to WSFs, suspensions of crude oil, cuts (oil distillation fractions), and some low-boiling aromatics exhibited increased mortality, reduced growth, and several morphological abnormalities: delay and irregularities in cleavage and development, poor differentiation of the head region, malformed upper jaw, protruding eye lenses, abnormally bent notochord, and various levels of inhibition of hatching and assimilation of yolk (Lønning, 1977). Both survival and feeding were further impaired following photodegradation of crude oil components (Solberg et al., 1982a). Kühnhold (1974) found through laboratory exposures that cod eggs were most sensitive to crude oil during the first few hours post-fertilization. Oil retarded development, delayed or prevented hatch, and induced significant mortality by 10 h post-exposure. Those larvae that did hatch showed a high level of abnormal development or abnormal swimming movements, and died within a few days.

Further work with early life stages confirmed the variety of adverse morphological changes. Developing cod embryos exposed to hexane extracts of sea-surface microlayer from 5 marinas located in the North and Baltic Seas showed significant embryo mortality as well as severe deformities in live hatched larvae at two of the sites (Kocan et al., 1987). Other studies noted a significant decrease in growth rate (Tilseth et al., 1981) and a toxicant concentration-dependent reduction in feeding (Solberg et al., 1982a,b,c). Such oil-induced disturbance of physiological and behavioral patterns would reduce feeding capability at the onset of feeding, with consequent high mortality in the field. Morphological disturbances that result in the ultimate death of the larvae, may in turn lead to serious effects on the fish population in the polluted area.

Exposure of cod eggs to the water-soluble fraction (WSF) of crude oils (50-150 µg/L) did not, however, significantly affect surface membrane permeability, nor was osmoregulatory ability of the embryo affected by these ecologically realistic concentrations (Mangor-Jensen and Fyhn, 1985). The WSF of North Sea crude oil (50 µg/L) had no effect on oxygen uptake of the eggs, yet strongly suppressed oxygen consumption by cod larvae at the time of final yolk absorption (5-7 d post-hatch), when the larvae begin to feed (Serigstad and Adoff, 1985). Crude oil extracts can also affect the energetic processes of cod eggs and larvae in addition to causing structural and developmental damage. One such extract (8 mg/L total dissolved hydrocarbons and dispersed oil) had no significant effect upon oxygen uptake in late-stage cod embryos and larvae with functional yolk sacs. Early embryos and starved larvae however showed reduced oxygen consumption when placed in the extract, and the starved larvae became narcotized (Davenport et al., 1979).

A number of behavioral effects were also documented. Larval cod exposed to sublethal amounts of the WSF of crude oil exhibited reduced growth, lower specific weight (neutral buoyancy), reduced feeding ability and swimming speed, and a serious disturbance of the swimming pattern. Larvae exposed to 4.1 mg/L or higher did not recover their feeding ability within 24 h of transfer to clean water (Tilseth et al., 1984). Cod larvae less than 20 mm are the size most harmed by exposure to a 50 ± 20 µg/L WSF of oil (Foyn and Serigstad, 1988). The effect of a 1- to 2-h exposure of cod eggs to the WSF is not acute but instead long-term, leading to starvation of the cod larvae. There is no recovery from the effects of exposure to the oil WSF when cod eggs or larvae are placed in clean seawater (Foyn and Serigstad, 1988).

Goksoyr et al. (1991) exposed cod eggs, larvae and juvenile cod to a WSF of North Sea crude oil (1-6 wk, 40-300 µg/L) and examined them for induction of Cytochrome P-450 enzymes. Although the exposure began during the egg stage, induction response was delayed until after hatching. The P-450 induction response was dose-dependent, and recovery in clean water resulted in normalization of P-450 levels (Goksoyr et al., 1991). Immunochemical response (i.e. induction of the specific Cytochrome P-450 1C as determined by immunochemistry) in the liver of juvenile cod and in homogenates of whole larvae was dose-dependent. Larvae and juveniles that were allowed to recover in clean seawater showed a P-450 decline toward control levels within a few days (Goksoyr et al., 1988). Laboratory exposure of juvenile cod to crude oil and oil dispersant produced significant changes in physiological parameters (heart rate, respiration, gill ventilation rate and amplitude) that did not occur until pollutant concentrations were close to lethal levels (Johnstone and Hawkins, 1980).

Because oil dispersants are often toxic to early life stages of fish, they are often studied together with oil as contaminants. The higher the aromatic content of an oil dispersant, the greater its toxicity to unfed haddock larvae (Wilson, 1977). Larvae were more susceptible to a dispersant's toxic effects immediately subsequent to first feeding. Numerous dispersants have been developed, however, that are 2 to 3 orders of magnitude less toxic than the kerosene-based dispersants available earlier.

PAHs

One component of petroleum hydrocarbons, the PAHs, has received additional attention because of its known ability to induce various MFO enzymes (Solbakken et al., 1980; Foureman et al, 1983; Stegeman et al., 1987). In male and female winter flounder collected over a 2-yr period from a relatively non-polluted area in Nova Scotia waters, seasonal variation (about tenfold) in hydrocarbon-inducible P-450 activity was less than that caused by environmentally realistic levels of pollutants (Addison et al., 1985; Edwards et al., 1988). Liver P-450 activity in this species, therefore, might be used to indicate environmental contamination. However, inducible P-450 activity in American plaice livers was low and did not vary significantly over a presumed organic pollution gradient in New Brunswick (Addison et al., 1991). The inference was that organic pollution was low and uniformly distributed in this estuarine-river system. Similarly, hepatic P-450 1A activity was highly variable in mummichogs (*Fundulus heteroclitus*) sampled from salt marshes in Massachusetts, and did not correlate with gradients from relatively clean to highly contaminated (Moore et al., 1995).

Nonetheless, liver toxicity is thought to be linked to Cytochrome P-450 activity. High levels of PAHs have been found in Boston Harbor sediments (Malins et al., 1985) and are thought to induce winter flounder liver toxicity. Various pathologies seen in liver tissue excised from winter flounder collected from Boston Harbor were attributed to PAH-induced genetic mutations leading to tumor formation (McMahon et al., 1988a,b; Figure 4.1). From the late 1980s to the present, however, tumor prevalence in winter flounder from Boston Harbor (Deer Island Flats) has decreased from a high around 12% to 0% , even though PAH concentrations in the sediments have not changed significantly (Mitchell et al., 1998). Thus, PAH may not be the causative agent for inducing hepatic tumors in winter flounder. Experimental work implied that the MFO system is involved in tumor formation. Cytochrome P-450 enzymes (EROD and AHH) were induced in winter flounder in the laboratory by injection of the PAH β-naphthoflavone (Stegeman et al., 1987). Immunohistochemical treatment of liver tissue from winter flounder further revealed evidence of liver histopathology (Smolowitz et al., 1989). Winter flounder fed chlordane- and benzo(a)pyrene-contaminated food developed proliferative lesions similar to cholangiocellular carcinomas in winter flounder taken from Deer Island Flats in Boston Harbor (Moore, 1991). These studies show that P-450 could play a role in the production of a mutagenic metabolite from environmental chemicals taken up by the fish, but are not sufficient to identify the causal agent.

The breakdown products of several PAHs are often more toxic than the parent compounds. PAH metabolites have been detected in adult cod (Davies et al., 1984). In parallel field and tank studies, fish exposed to nominal levels (50 µg/L) of benzo(a)pyrene in the water showed liver AHH values 5-40 times that of the control. These metabolites can be accumulated by predators via their prey (McElroy and Sisson, 1989). Risk assessments for predators must therefore take into account metabolites produced by prey as well as the parent compound.

PESTICIDES

Chlorinated pesticides are known to elicit neurological effects on organisms, primarily by interfering with acetylcholine/acetylcholinesterase function. Laboratory exposure of 3-yr old cod to low levels of DDT produced tachycardia (rapid heartbeat), a decrease in the frequency of respiration, and disruption of the central nervous system's regulation of muscle contraction in stomach and gut. Upon removal of the toxicant, however, normal functions returned after 6-7 d (Shparkovskii, 1982). Other modes of toxicity are also possible, although generally at high organochlorine concentrations. For example, chlordane in high doses induced severe liver damage, and at subacute doses produced macrophage aggregation and a persistence of necrogenic effects in the liver (Moore, 1991). Hydropic vacuolation, regarded as a precursor of liver neoplasia, appears to be correlated with environmental levels of chlorinated hydrocarbons (Mitchell et al., 1998).

Cod larvae are far more sensitive to the chlorinated pesticide DDT and its breakdown product DDE than are the membrane-protected embryos. Percentages of malformed and dead embryos and larvae increased with increasing concentrations of DDT, which was overall more toxic than DDE (Dethlefsen, 1976). Chlorinated pesticides also exhibit toxicities to winter flounder eggs and adults. Abnormal gastrulation and a high incidence (39%) of vertebral deformities were seen in developing eggs from adult winter flounder experimentally exposed to very low, sublethal concentrations of DDT (1-2 µg/L; Smith and Cole, 1973; Figure 4.1).

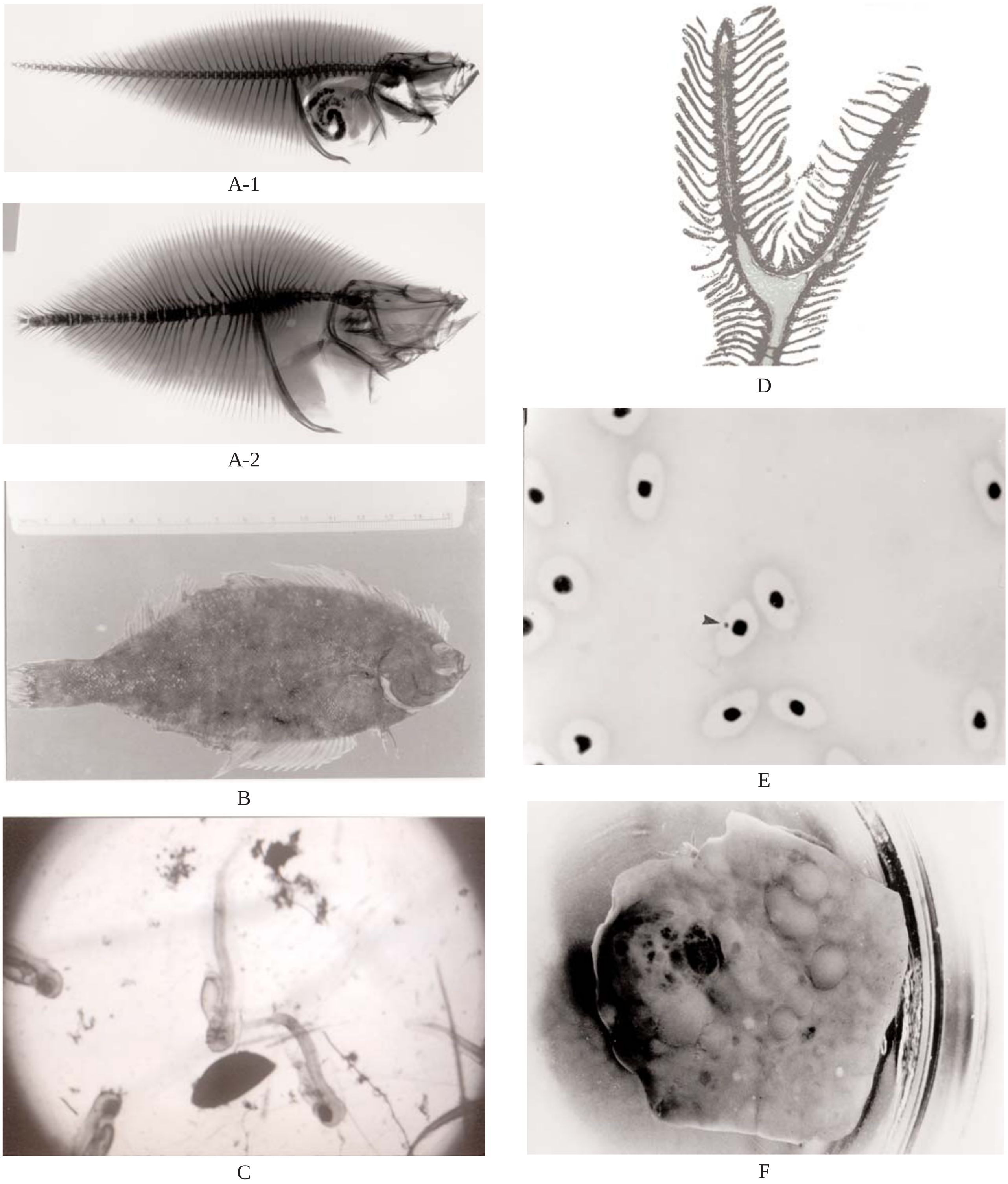

Figure 4.1. Some examples of abnormalities in winter founder, *Pseudopleuronectes americanus*, associated with contaminated environments. Similar abnormalities have been reported for other marine fish species. A. Normal vertebral column (A.1) and fusion and compression of groups of individual vertebrae (A.2) as described in Ziskowski et al. (1987); B. Finrot disease (fin erosion) as described in Ziskowski et al. (1987); C. Abnormal skeletal development in larval flounder as described in Nelson et al. (1991); D. Gill bifurcations associated with contaminated sediments as described in Pereira (1988) and Pereira et al. (1992); E. Red blood cell micronucleii (arrow) as described in Hughes and Herbert (1991); and F. Liver tumors (neoplasia) as described in Murchelano and Wolke (1991).

No such effects were seen in eggs from flounder similarly exposed to dieldrin, nor were residues of either insecticide detected in the milt of exposed or control male winter flounder.

PCBs

Toxicity has been attributed to PCB exposure in both the field and the laboratory. Cod having ulcus syndrome (epidermal lesions thought to be due to an imbalance in corticosteroid metabolism) had significantly higher PCB residue concentrations in liver tissue than did cod without the syndrome (Stork, 1983). In heavily urbanized areas of Long Island Sound having higher PCBs in the sediments (New Haven, Hempstead, Norwalk), winter flounder tended to have lower reproductive success, when spawned in the laboratory, than did flounder from less urbanized sites (Nelson et al., 1991). In that same study comparing several different sites in the Sound, winter flounder embryos from the New Haven site had the most abnormalities and the lowest percent viable hatch. Nelson et al. (1991) also found that flounder with high liver concentrations of PCB (Boston) had small larvae. Black et al. (1988) reported that eggs of winter flounder from New Bedford Harbor (MA) were significantly higher in PCB content (39.6 µg/g dry wt) than those from Fox Island (a relatively clean area in Narragansett Bay, RI), and larvae hatched from these New Bedford eggs were significantly smaller in length and weight. There was a significant inverse relationship between PCB content of the eggs and length or weight at hatch.

PCB concentrations in winter flounder liver sampled from 3 southern New England areas with differing degrees of PCB and PAH contamination (New Bedford Harbor, NBH; Gaspee Point, GP; and Fox Island, FI) reflected the varying degrees of PCB-contaminated liver, with NBH> GP> FI. Liver EROD activity was the same at all three sites, but P-450 concentration (measured immunologically with an antibody against vertebrate P-450 1A1) was significantly higher in the NBH fish (Elskus et al., 1989). The data suggest that P-450's catalytic activity (for EROD) is being competitively inhibited at NBH, (even though P-450 levels are elevated) possibly by some congeners of PCB (Gooch et al., 1988; Elskus et al., 1989). In contrast, sexually mature female winter flounder showed lowered EROD activity and immunologically quantified P-450E concentrations, apparently due to a hormonal effect acting primarily to suppress induction of P-450E, the activation catalyst for EROD (Förlin and Hansson, 1982).

Different PCB mixtures produce different effects in marine fish. Feeding Aroclor 1254 to juvenile cod to produce liver concentrations of ca. 900 µg/g wet wt had variable effects on the Cytochrome P-450 enzyme system. The P-450 enzyme ethoxycoumarin o-de-ethylase (ECOD) was induced 30-fold, but Aroclor 1254 had no effect on ethoxyresorufin o-de-ethylase (EROD) activity. Feeding Aroclor 1016 to juvenile cod induced no increase in P-450 enzyme activity (Hansen et al., 1983).

OTHER CONTAMINANTS

Recent studies have focused on a group of chemicals, the "endocrine disruptors", that interfere with fish (and by implication, human) endocrine systems when present at extremely low environmental concentrations. In plasma of sexually mature male winter flounder exposed to crude oil, for example, total concentrations of the sex hormone androgen (both free and conjugated) were statistically lower than controls (Truscott et al., 1983). During early maturation of the gonads, however, oil exposure had no effect on total plasmatic androgens and estradiol in either female or male flounder.

One subset of these endocrine disruptors, the "environmental hormones", include chemicals that may mimic the function of the sex hormones androgen and estrogen. A broad range of chemicals are known to be estrogenic, including several PCB congeners, dieldrin, DDT, phthalates, and alkylphenols, as well as synthetic steroids (estradiol, ethynylestradiol, etc.). While their individual estrogenic potencies differ, all are potentially present in sewage discharges. Alarm over these chemicals was raised in the early 1990s when excessively high concentrations of vitellogenin were measured in male rainbow trout caged in the effluent of sewage treatment plants along English rivers (Harries et al., 1996, 1997). Ordinarily, vitellogenin, a yolk precursor protein, is only produced in the livers of mature female fish, when signaled by estradiol in the blood. Recent studies have shown that this phenomenon is not restricted to freshwater fish. Male flounder, *Platichthys filesus*, collected from five English estuaries and from the southern North Sea also displayed elevated vitellogenin levels (Lye et al., 1999; Allen et al., 1999). Investigations are now underway to look for environmental hormones in Chesapeake Bay, Long Island Sound and Boston Harbor.

MIXED CONTAMINANTS

Fish are never exposed to single contaminants in nature, but to complex mixtures of organic and inorganic substances of varying potency. Comparisons of fish sampled from relatively clean and polluted sites have documented a variety of adverse impacts, including slower reaction times, skeletal abnormalities, higher prevalences of degenerating hepatic parenchymal cells, and decreases in such biochemical parameters as hepatic and pectoral fin ascorbic acid concentrations, hepatic glycogen and lipid levels, plasma glucose concentrations, brain serotonin and norepinephrine concentrations and the concentration ratio of various free amino acids in muscle tissue (Olofsson and Lindhal, 1979; Despres-Patanjo et al., 1982; Carr et al., 1991). Planktonic stages have been shown to exhibit cytotoxicity and decreased survival rates, mitotic abnormalities, chromosome damage, slower developmental rates, cell necrosis, and smaller yolk reserves (Perry et al., 1991; Gronlund et al., 1991; Buckley et al., 1991; Nelson et al., 1991; Longwell et al., 1992). A variety of other toxicological responses have been demonstrated in fish sampled from heavily polluted harbors (e.g. Halifax, Tay et al., 1991; Salem and Boston, Zdanowicz et al., 1986, Moore et al, 1995, Wall et al., 1998; New Haven CT, Wolke et al., 1985, Gronlund et al., 1991; various harbors in the Northeast, Johnson et al., 1992), and from sites adjacent to sewage effluents (Weis et al., 1992). However, it is generally not possible to attribute the adverse effect to a specific contaminant.

The additivity of some contaminants was recognized early on for the various PCB congeners, and led to the development of Toxic Equivalency Factors (TEF) that are specific for fish (Newsted et al., 1995). Additional models are now being developed and tested to deal with the effects of a broader range of contaminant mixtures in fish (Broderius et al., 1995; Logan and Wilson, 1995; Pape-Lindstrom and Lydy, 1997). Threshold concentrations of additive toxic metal mixtures need not be high to produce toxic effects (e.g. cod exposed to Cu plus Zn, Swedmark and Granmo, 1981). Limited water circulation in estuarine and coastal waters would be most likely to produce examples of such toxic effects in young fish that use these nursery areas.

A drawback to the TEF approach is that it only considers additive responses. A number of studies on fish species have demonstrated the importance of antagonistic and synergistic interactions of contaminants. For example, several lipophilic contaminants interact synergistically to modulate the levels of two important detoxification components in the liver of English sole (*Pleuronectes vetulus*; Nishimoto et al., 1995). Different metals often compete for the same binding sites within cells. Because of this, metals such as Ca can protect against Cd toxicity, as shown in the mummichog, *Fundulus heteroclitus* (Gill and Epple, 1992).

The interactions between metals and organic contaminants have received particular attention. Winter flounder exposed to a PCB (24 h, 1 mg/L) in the presence of added cadmium (200 μg/L), accumulated significantly less PCB in their liver and gills than did those dosed with PCB alone (Carr and Neff, 1988). Additional studies have shown that cadmium and benzo(a)pyrene interact to alter the biotransformation pathways in the liver of the mummichogs (van der Hurk et al., 1998). Zinc and phenanthrene interact to effect the toxicity in sheepshead minnows, *Cyprinodon variegatus* (Moreau et al., 1999). It is therefore clear that much more work needs to be done in order to better understand the impacts of contaminant mixtures on fish populations.

THE CBR APPROACH

Is bioaccumulation useful for predicting impacts? The answer is, in fact, threefold: 'no', 'yes' and 'maybe'. No, it will not be useful for some contaminants either now or in the future. Yes, it

> *appears to be useful now for some contaminants..., and maybe in the future it will be useful for some, but not all contaminants, and not for all organisms.*
>
> *-Peter M. Chapman, 1997*

It is important to point out that most of the studies of toxicological effects of contaminants reported in the literature have related contaminant exposure (i.e. concentrations in the surrounding water) with the toxicological effect. This is typically the approach that has been taken in toxicity testing, where LD_{50}s are computed for exposure concentrations. McCarty and Mackay (1993) have recently advocated correlating toxicity with the concentration of a contaminant that accumulates in the tissues of an organism rather than the concentration of contaminant in the surrounding water. This is referred to as the “Critical Body Residue (CBR)” approach to assessing toxicity. Since toxicity should more directly correlate with the concentration of a toxicant at the site of action in the cells or tissues rather than the exposure concentration, this CBR approach should allow us to better assess toxicological effects by pinpointing a potential causative agent among a suite of multiple exposed contaminants, and should allow us to predict sublethal toxicity from existing bioburden monitoring data.

To support this CBR approach, existing published data have been reexamined to identify those instances where body burdens were measured along with toxicological responses (Jarvinen and Ankley, 1998). For example, cod from the Baltic Sea that were found to have elevated levels of cadmium in liver and kidney tissue also had externally visible skeletal deformities (compressions of the spine and deformities of the jaw; Lang and Dethlefsen, 1987; e.g. Figure 4.1). For the commercially important Northeast species, less than 30 such studies have been identified (Table 4.3). While a direct relationship between contaminant body burdens and the degree of toxicity would be expected, establishing this relationship is often elusive, particularly when studies were not specifically designed as CBR investigations. As described previously, both Cd and Hg induced adverse effects in long-term laboratory studies on winter flounder (Calabrese et al., 1975, 1977). Hg was readily accumulated in blood and gill tissues in a dose-dependent manner, yet there was no statistically significant accumulation of Cd under the experimental conditions employed. The CBR approach has only been directly tested for fish in a few laboratory studies (e.g. Niimi and Kissoon, 1994). Nevertheless, it is clear that this approach holds much promise for assessing the potential of adverse responses to pollutants in field collected fish. Additional work will further refine and validate the approach in the years to come.

POLLUTION-LINKED HISTOPATHOLOGY AND DISEASE

> *Although pollution has been implicated in the high prevalence of lesions in eastern North Atlantic bottom fish, conclusive cause and effect relationships [remain to] be established.*
>
> *-R.A. Murchelano, L. Despres-Patanjo and J. Ziskowski, 1986*

Fish that are collected from contaminated sites often exhibit a broad range of histopathologies and diseases, in addition to the lesions and morphological effects mentioned earlier. Four examples are presented below, as well as a discussion of the interactions between parasites and pollution effects. Investigators have often pointed out the apparent correlation between pollution and these adverse effects, although causation has only been suggested. In only a few cases have laboratory studies been done to investigate possible causative factors.

A variety of liver pathologies have been identified in cod, yellowtail and winter flounder collected from the field (Despres-Patanjo et al., 1982; Freeman et al., 1981a, b, 1983; Murchelano et al., 1986; Murchelano and Wolke, 1991; Turgeon and O'Connor, 1991). Of 100 live cod collected from Halifax Harbor, 73 had histopathological lesions in their livers (Freeman et al., 1983). Histopathological analysis of livers from winter flounder revealed one liver neoplasm in a fish from the western end of Long Island Sound and none in fish from the eastern end of that west-to-east pollution gradient (Turgeon and O'Connor, 1991). High prevalences of liver lesions, blood cell abnormalities, liver DNA damage, liver neoplasms, concentrations of organic chemicals and trace metals, and high levels of PCBs in gonads in winter flounder from New Haven have been found (Greig and Wenzloff, 1977; Greig and Sennefelder, 1987; Gronlund et al., 1991). “Fatty change,” a degenerative process in the liver, has been attributed to exposure to organic contaminants or trace metals (Freeman et al., 1981a,b). Considered with other pathological signals, exceptional accumulation of liver lipid and increases in liver size have been linked to body burdens of PCB, pesticides, and other organic toxicants (Sherwood, 1982; Freeman et al., 1983). Pathological changes in liver seem to become progressively greater with increasing size of the fish (Freeman et al., 1983; Murchelano and Wolke, 1991). In a Boston Harbor field study, for example, tumors were not found in fish smaller than 32 cm in length (Murchelano and Wolke, 1991). The same study revealed a pattern in liver pathology: a progression from necrotic lesions to neoplasia (Figure 4.1), and suggested pollutants as the likely inducers of the lesions (Murchelano and Wolke, 1991).

Investigators have attempted to induce liver tumors in the laboratory, but with limited success. Payne and Fancey (1989) reported that liver hypertrophy in winter flounder increased with increasing oil exposure for 4 months to varying concentrations of crude oil in sediments although the number of melanomacrophage centers in livers was reduced. Boston Harbor sediment extract, injected peritoneally, was acutely toxic to winter flounder; perivascular edema was observed after 10 d in survivors (Moore, 1991). Gardner and Yevich (1988) reported that fish exposed for 90-120 d to sediment from Black Rock Harbor (Bridgeport CT) developed neoplastic or proliferative lesions in the kidney, olfactory and lateral line sensory tissues, gastro-intestinal tract, and buccal (cheek) epithelium but not in the liver; cytopathology and cell necrosis were also detected in the pituitary. In the field, winter flounder collected from Black Rock Harbor and New Bedford Harbor (MA) had similar lesions.

A second well-studied example of histopathological impact is the increase in the incidence of small inclusions (micronuclei, MN) in fish red blood cells (Figure 4.1). MN were elevated sixfold in flounder from the New York Bight Apex as compared to fish from the inshore Gulf of Maine and Block Island Sound, and twice those found in Georges Bank and Long Island Sound flounder (Hughes and Hebert, 1991). Inshore New Jersey and Virginia fish had significantly higher MN frequencies than those from the Gulf of Maine and Block Island Sound. There were higher frequencies of MN in flounder from Hempstead and Shoreham, N.Y. as compared to most other sites in the Sound. Erythrocyte MN were consistently higher (Hughes and Hebert, 1991) in flounder from the more highly contaminated stations examined, New York Bight Apex and Hempstead, which are contaminated with metals and PAH (Carmody et al., 1973; MacLeod et al., 1981; U.S. EPA, 1984). Winter flounder collected from the coastal mid-Atlantic had statistically higher erythrocyte mutation frequencies than those from more offshore waters (Longwell et al., 1983). Winter and windowpane flounder from western Long Island Sound had significantly higher frequencies of micronuclei than those from the New York Bight, with fish from both these areas having significantly higher mutation frequencies than flounder sampled elsewhere (Longwell et al., 1983). The higher incidences suggest a link with environmental pollution.

Several diseases also appear to correlate with pollution. A high incidence of fin-rot disease was observed in windowpane, winter, and yellowtail flounders exposed to materials dumped in the New York Bight (O'Connor, 1976; e.g. Figure 4.1), winter flounder having the highest incidence. The highest rate of fin-rot in yellowtail and winter flounder was seen in fish collected from the inner New York Bight when compared to either offshore waters of the outer New York Bight or inshore within Massachusetts Bay (Ziskowski et al., 1987). PCB concentrations in muscle, liver and brain tissues were higher in winter flounder with fin-rot from contaminated sites (primarily winter flounder) than in fish from reference sites (Sherwood, 1982). The erosion pattern of the fins and the association of higher prevalences of fin erosion with greater degrees of sediment contamination suggest that fin-sediment contact in an area where toxic contaminants have accumulated on the bottom is an important factor in development of the disease.

Lymphocytosis (elevated mean blood lymphocyte counts) is a second example of a potentially pollution-related disease. High lymphocyte counts in winter flounder were correlated with liver necrosis and suspected levels of sediment chemical contamination; winter flounder collected from Boston Harbor had higher numbers of immature erythrocytes than did those from less urbanized environments (Daniels and Gardner, 1989). Disturbances in the distribution of blood cells and alterations in lymphocyte counts were related to neoplastic lesions and indicative of chemical contamination in sediments.

The interactions between fish parasites and contaminant impacts are complex. On the one hand, parasitic infections may make fish more prone to the additional stress caused by pollution. In a laboratory exposure, hemoprotozoan-infected cod reacted more sensitively to petroleum hydrocarbons than did non-infected fish (Khan, 1987), as measured by poor body condition, excessive mucus secretion by the gills, retarded gonadal development, and greater mortality. Similarly, juvenile and adult winter flounder, some infected with the blood parasite *Trypanosoma murmanensis*, were exposed to sediment contaminated with crude oil (6 wk, 2.6-3.2 mg/g) or to clean sediment (Khan, 1987). Mortality was significantly higher (89% for juveniles, 49% for adults) in infected, oil-exposed fish than in fish with either condition alone. On the other hand, the stress caused by pollution may make fish more susceptible to parasitic infection. In the field, parasitic infections accompanied by lowered host resistance were found to be more prevalent in cod after chronic exposure to petroleum hydrocarbons (Khan, 1990). Likewise, exposure to oil in the laboratory, increased the incidence of *Trypanosoma* infection and death (Khan, 1987, 1991). Interestingly, parasites may in some cases be more susceptible to pollutants than the fish host. As a result, fewer parasites might be found in fish exposed to contaminants. For example, Khan and Kiceniuk (1983) observed that there are fewer parasites in fish exposed to oil, and suggested that the lowered parasitism might be attributed to toxicity induced by water-soluble fractions of crude oil and/or modification of the gut environment. The intensity and prevalence of parasitic infections were more pronounced in fish exposed to water-soluble extracts than in those exposed to oil-contaminated sediment (Kahn and Kiceniuk, 1983).

POPULATION EFFECTS

As described above, there is ample evidence to show that commercially important species bioaccumulate contaminants and exhibit toxicological responses to contaminants that they are exposed to. These effects are generally restricted to heavily polluted coastal sites. Although the impacts on individual fish may be of interest to toxicologists, it is the possible adverse impacts on fish populations that are of interest to fisheries managers and environmental regulators. Assessing the impact of contaminants at the population level of biological organization has proven to be difficult, but is currently one of the most active areas of toxicological research. Two avenues are being pursued: (1) using biomarkers; and (2) incorporating contaminant impacts into currently used population dynamic models.

Biomarkers are biochemical, cellular, physiological or behavioral changes that can be measured in tissues and provide evidence of contaminant exposure and/or toxicological effects (Depledge et al., 1995). Investigators have attempted to correlate one or more biomarkers with adverse impacts on fish populations (Johnson et al., 1992; Stein et al., 1992; Wall et al., 1998). For example, the induction and expression of Cytochrome P-450 enzymes have been proposed as biomarkers of organic contaminant exposure (Stein et al., 1992; Moore et al., 1995). However, in some highly polluted environments (e.g. New Bedford Harbor), EROD activity in winter flounder was depressed even though P-450 gene induction was evident (Elskus et al., 1989). In another polluted locale (Newark Bay), both adult and larval mummichogs failed to exhibit P-450 induction at all, suggesting adaptation to pollution stress (Elskus et al., 1999). These examples demonstrate that biomarkers must be asssessed carefully, and the use of a single biomarker is unwise. Biomarkers cannot, at present, be used to predict impacts on future population structure.

Investigators have also attempted to incorporate toxicity data into fisheries models. Initially, modelers simply modified the mortality term in population or recruitment models to include a component which they attributed to contaminant toxicity (Waller et al., 1971; Wallis, 1975). However, these authors provided no rigorous justification for their contaminant-induced mortality terms. Another approach applied multiple linear regression to fish stock data to assess the significance of hydrological factors and contaminant impacts (e.g. sewage loading, dissolved oxygen concentrations, biological oxygen demand) on the historical time series data (Summers and Rose, 1987). This modeling indicated that hydrological factors were far more important than contaminant effects for striped bass stocks in the Potomac, Delaware and Hudson Rivers, whereas pollution was of more significance for American shad. While this approach can explain historical data, it is not useful for forecasting (Summers and Rose, 1987). A third approach combined toxicological data with physical oceanographic data in a risk-assessment model (Spaulding et al., 1983, 1985) in order to assess the probable effects of an oil spill on cod populations. This assessment provided useful predictions: (1) most of the hydrocarbon impact on cod would occur within the first 60 d after a spill; (2) 41.5% of the spawned cod would be adversely affected by oil concentrations in excess of 50 μg/L; (3) cumulative loss to the population would peak at 23.9% in the 7th year after the spill; and (4) of the four seasons, winter and spring spills would have the greatest impact.

Recently, investigators at Oak Ridge National Laboratory have further advanced the methodology needed to incorporate toxicity data into fisheries models. Two of the most critical steps in this process are the application of acute toxicity test data (e.g. 96 h LC_{50} data) to predict thresholds for chronic lethal and sublethal effects, and the extrapolation of data from one tested species to another (Suter et al., 1987; Suter and Rosen, 1988; Barnthouse et al., 1987, 1989). Not only survival, but also sublethal effects, including reductions in fecundity, can then be calculated (along with confidence limits) and applied to fisheries models. Initial work indicated that fecundity is the most sensitive toxicological factor that needs to be incorporated into population models, and has a greater long-term impact than reductions in survival of young-of-the-year fish (Suter et al., 1987; Barnthouse et al., 1989). Barnthouse et al. (1990) then predicted long-term changes in Gulf of Mexico menhaden and Chesapeake Bay striped bass populations using a Leslie Matrix-type life cycle model. They included natural, contaminant and fishing-induced age-specific mortality, as well as age-specific fecundity in their analysis. Because of differences in life history, menhaden and striped bass had different capacities to sustain the same level of contaminant-induced mortality. Menhaden were better able to tolerate pollution. The model also showed that fish populations that have been reduced in numbers by overfishing were much more susceptible to the increased mortality and reduced fecundity caused by contaminants.

Current models used by fisheries scientists to predict long-term effects of exploitation on fish populations are quite imprecise. Adding in the effects of contaminants, with its own order-of-magnitude confidence in the extrapolation from acute toxicity test data adds another layer of uncertainty. Barnthouse et al. (1990), for example, cautioned that the uncertainty in estimating long-term effects on fish populations using their model was generally greater than the range of responses to contaminants, since the largest source of this uncertainty is attributed to the inherent uncertainty of currently used fisheries models. Nevertheless, modeling is already providing considerable insights as to the interactions between contaminants and overfishing on various fish populations. With further refinements, more holistic models should be able to examine a multitude of forcing factors, including fishing pressure, habitat alteration, contaminant effects and natural environmental variability.

SUMMARY AND DISCUSSION

The highest concentrations of chemical contaminants are to be found in coastal, industrialized or heavily urbanized, and waste-disposal areas. Such estuarine, coastal habitats are also the spawning and nursery areas for many important commercial fishes. These early life stages are most susceptible to toxicants, the larvae more so than the eggs, as the latter have the protection of a membrane (Dethlefsen, 1976; Mangor-Jensen and Fyhn, 1985; Foyn and Serigstad, 1988).

In some fish species, tissue concentrations of pollutants do not necessarily reflect sediment concentrations of those pollutants, whether metals or organics (Greig and Wenzloff, 1977; Greig et al., 1983; MacDonald, 1991). The work of Marthinsen et al. (1991) illustrates the difference in this respect between fish species: they found that PCB levels in Atlantic cod reflected a decreasing PCB pollutant gradient from the mouth of Norway's largest river, whereas PCB levels in the European flounder did not. In winter flounder, PCB body burdens were accumulated from prey species in the sediments, rather than from the water column or from sediment contact (Connolly, 1991).

Circumstantial evidence linking disease and abnormalities in various fish species to polluted habitats is abundant. Proliferative lesions in endocrine, exocrine, respiratory, sensory, excretory and digestive organs, alteration of plasma protein and ion concentrations, and interferences in metabolic pathways were found to be characteristic of species such as winter flounder that spend much of their lives in some moderate to highly contaminated inshore areas. Liver disease was found to be absent in populations from uncontaminated offshore areas. The degree of sediment chemical contamination and disease suggest a causal interrelationship (Gardner et al., 1989).

More immediately associated with pollutant exposure, abnormally high levels of detoxifying P-450 enzymes may signal (for a short time) exposure to organic contaminants such as PAHs, PCBs and chlorinated pesticides (Addison and Edwards, 1988). This response varies with sex and gonadal maturation (Spies et al., 1988a,b; George, 1989). Similarly, induction of metal-binding proteins may signal (for a short time) exposure to heavy metals (Fowler and Gould, 1988; George, 1989; Roesijadi and Robinson, 1995).

Unlike PAHs and chlorinated pesticides, different PCB congeners sometimes elicit conflicting effects (Hansen et al., 1983), with some congeners inhibiting others (Gooch et al., 1988). A hormonal suppression by one PCB congener of the detoxication of another PCB, for example, has been observed in sexually mature female winter flounder (Førlin and Hansson, 1982). Black et al. (1988) found a significant inverse relationship between PCB content of eggs and length and weight of larvae at hatch. Goksoyr et al. (1991) have shown that in the early life stages of a fish, the normal protective production of enzymes that break down organic toxicants for elimination is delayed until after hatching.

Tissue concentrations of organic contaminants such as PAHs, PCBs, and pesticides are significantly correlated with body-fat content (Reid et al., 1982); PCB body burdens vary with season in female but not in male fish (Marthinsen et al., 1991). The fattier the liver, the slower the clearance rates of these fat-soluble toxicants (Skåre et al., 1985).

Induction of Cytochrome P-450 enzymes is associated with liver pathology in fish, a circumstance that could play a role in the production of cancer-producing agents from environmental chemicals by generating breakdown products more toxic than the parent compound (Smolowitz et al., 1989). Ulcer-like lesions are considered to be a result of hormonal imbalance caused by PCB assimilation (Stork, 1983). Chronic exposure of adult cod to crude oils produces severely disabling lesions and reproductive impairment (Khan and Kiceniuk, 1984; Kiceniuk and Khan, 1987).

In the case of metal-organic interactions, cadmium strongly depresses several detoxifying enzyme systems (George, 1989), and appears to depress PCB uptake in winter flounder (Carr and Neff, 1988). In contrast, the European flounder, *Platichthys flesus*, when exposed to diesel oil showed no increase in detoxifying activity when copper was added to the oil (Addison and Edwards, 1988). Overloads in marine animals of even essential trace metals (notably copper) can interfere with normal intracellular metal regulation, with consequently lower fish health and often reproductive failure. These phenomena have been observed most clearly in a Gulf of Maine marine bivalve, the sea scallop (Gould et al., 1988; Fowler and Gould, 1988).

From the foregoing review, it is clear that pollutants can alter the normal health and physiology of Northeast fishes. When these fish are exposed to pollutants in the laboratory, a variety of adverse effects have been clearly demonstrated. When fish are collected from contaminated environments, pollutants are measured in the tissues of these fish and associated harmful physiological and biochemical effects can be observed (Table 4.3). The exact contaminant responsible for the effect is often elusive (Wolfe et al., 1982) because multiple pollutants are usually found in a contaminanted environment and the pollutant effect may be due to the combined action (additive, synergistic or antagonistic) of a mixture of pollutants.

The major factor that remains unclear is the link between these observations and the effect that pollutants might have on the population structure of the various fish stocks. Our failure to make reasonable quantitative estimates of these effects at the population level may be a result of our inability to separate clearly effects of pollutant stress from other environemental stresses. Overfishing, climate changes, food availability, habitat alteration, and predation are among the other factors that alter fish population levels and obscure any possible effects of a degraded habitat (Cohen et al., 1991; Sindermann, 1994).

Sindermann (1994) perhaps best sums up the present state of this dilemma:

> *It seems, with the evidence presently available, that factors other than pollution are overriding in determining fish abundance, but we lack sufficient quantitative data to make positive statements about cause and effect relationships of abundance and pollution.*
>
> *-C.J. Sindermann, 1994*

He further suggests that implementation of new pollution monitoring and assessment programs, critical findings from laboratory and field studies, better and longer-term data sets, as well as the development of new simulation models, may provide answers to this vexing problem of distinguishing pollution effects from overfishing, habitat degradation and other environmental factors, and allow us to assign a quantitative estimate of the impacts of pollutants upon fish stocks.

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Chapter V

The Effect of Habitat Loss and Degradation on Fisheries

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> *The importance to the United States of the fisheries on its coasts can scarcely be exaggerated, whether we consider the amount of wholesome food which they yield, the pecuniary value of their products, the number of men and boys for whom they furnish profitable occupation, the stimulus to ship and boat building which they supply, and, not the least of all, their service as a school for seamen, from which the merchant-marine, as well as the Navy of the country, derive their most important recruits. -Baird, 1873*

INTRODUCTION

The notion that the abundance of an organism is to a large measure a function of the quality and quantity of suitable habitat available to it has been an integral part of the science of ecology since that field of endeavor came into existence in the 20th Century. Perhaps because the habitats beneath the sea are hidden from direct view, the connection between fisheries and habitats has only recently received much attention from scientists and fisheries managers. In the terrestrial realm, we know a great deal about how human alterations of the landscape have changed the suite of species that can survive in a given area. We can make reasonable predictions about which species will occupy certain habitat types under certain management regimes – for example, how the return of forests to much of New England as farms were abandoned has influenced the distribution of such species as black bear, white-tailed deer, and moose. For a variety of reasons, our understanding of the relationship between marine fish and their habitats is, by comparison, at a primitive state, and our predictive capabilities are minimal. The purpose of this chapter is to review human impacts on marine fish habitats and address the question of how such impacts might be affecting fish populations superimposed on fishing itself and natural environmental variation.

INSHORE AND OFFSHORE FISH HABITATS

Much of the information that relates the impacts of habitat alterations by humans to fish populations will be taken from studies of nearshore and estuarine fish, since a number of major studies have been carried out there. It has been recognized since the early 1800s that nearshore marine habitats, particularly coastal embayments and estuaries, are important for the survival of some species of fishes and shellfishes. In some parts of the country, coastal species are the major component of commercial and recreational fisheries. Even where the direct harvesting of estuarine and coastal species is not numerically or economically significant, estuaries and coastal embayments may still be essential for fisheries because they serve as nurseries for the juvenile stages of species harvested offshore or for the prey of commercially important species.

The high productivity of bays and estuaries, particularly those with salt marshes and other vegetated habitats, may help support offshore food webs (Odum, 1980; Deegan, 1993). Thus nearshore and offshore habitats are linked ecologically. What we can learn from the impacts of habitat alterations on nearshore fish will be relevant to habitats further offshore, such as the fishing grounds of Georges Bank.

Because it is a popular place to live, visit, recreate, and conduct business, the coast has been measurably altered and degraded by a variety of human activities: (1) urbanization, (2) agriculture, (3) alteration of water flows by roads, railroads, and dredging, (4) diversion of freshwater flows for alternate uses, (5) overharvest of biological resources, and (6) pollution from point and non-point sources (Table 5.1). There is also an understanding of the physical disturbance of benthic habitats by mobile fishing gear. Because this type of disturbance is so widespread and intense both inshore and offshore, many scientists and managers are concerned that it could be impacting fish populations, perhaps inhibiting recovery of overfished stocks.

Table 5.1. Major categories of human impact on coastal aquatic habitats.

CATEGORY	EXAMPLE
Physical habitat loss **P**	drainage, shoreline dredging, spoil disposal
Hydrology changes **H**	freshwater diversion or withdrawal, tidal constriction due to causeways or culverts, hydroelectric plants
Eutrophication **E**	fertilizers, sewage, runoff, septic systems, land use
Sediment delivery changes **S**	increased due to soil erosion, decreased due to upstream dams
Introduced species **I**	*Phragmites*, green crabs
Fishing methods-physical impacts **F**	trawls, clam rakes, scallop dredges
Contaminants **C**	heavy metals, hydrocarbons, organics, waste dumping/debris
Global climate-changes in sea-level **G**	higher water temperatures, increased storm events, flooding of marshes, declines in productivity

DEFINITION OF HABITAT

Essential fish habitat (EFH) means those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity. For the purpose of interpreting the definition of essential fish habitat: Waters include aquatic areas and their associated physical, chemical, and biological properties that are used by fish and may include aquatic areas historically used by fish where appropriate; substrate includes sediment, hard bottom, structures underlying the waters, and associated biological communities; necessary means the habitat required to support a sustainable fishery and the managed species' contribution to a healthy ecosystem; and "spawning, breeding, feeding, or growth to maturity" covers a species' full life cycle (Federal Register, 1997).

We have chosen to define habitat broadly for this paper. Habitat is that part of the environment on which organisms depend directly or indirectly to carry out their life processes. For fish, this includes spawning grounds, nursery areas, feeding areas, and migration routes. The definition of what is a habitat for a particular species will vary according to the number and extent of life processes used by an ecologist to delineate a habitat. In this chapter, we will consider habitat as those parts of the environment that together make a place for organisms to survive and prosper. This includes the physical environment (such as structure provided by plants, sediments or temperature), the chemical environment (such as salinity and dissolved oxygen), and the many organisms (such as plants and invertebrates) that comprise a food web (Cronin and Mansuerti, 1971; Peters and Cross, 1991; Hoss and Thayer, 1993).

Defining and describing what habitats are essential to a species is complicated. Different life history stages may require a different habitat, or an organism may use a variety of habitats during each of its life stages. For example, menhaden require at least four different habitats (Deegan, 1993). They spawn offshore (1), then depend on tidal currents set up by river discharge (2) to bring larvae into estuaries where they use salt marshes (3) for feeding and protection from predators. As juveniles, they use open bay areas (4) to grow to a size where they can move offshore (back to 1). A number of common species in the Massachusetts region have populations that make regular, seasonal migrations between estuaries and nearshore coastal waters. These include winter flounder, summer flounder, three-spined stickleback, bluefish, striped bass, Atlantic silversides and a variety of diadromous species (Table 5.2).

Physical structure is the most visible aspect of a habitat and is therefore the basis for most habitat classifications. Kelp beds, seagrass meadows, intertidal marshes, intertidal and subtidal mud and sand flats, or offshore ledges and banks provide distinct physical structures that serve as habitats for fish and other marine organisms. Less obvious structural components are fronts separating different water masses or plumes of turbid, low salinity water produced by large rivers. Structure alone is not sufficient to provide a functional habitat for an organism. Habitats can be dysfunctional, even though the basic physical structure is present, if aspects such as food webs or primary production have been altered. In addition, environmental properties such as temperature, salinity, and nutrient (food) availability greatly influence the use of these areas.

Some habitats cannot be assigned to a specific location. The convergence of the freshwater plume of a river and the ocean changes its location with the discharge of the river and the tidal regime. These areas are an important pelagic habitat and support dense populations of zooplankton that are critical to the survival of larval and juvenile stages of fishes (Townsend, 1983, Govoni et al., 1989; Grimes and Finucane, 1991; Govoni and Grimes, 1992; Doyle et al., 1993).

GENERAL CATEGORIES OF IMPACTS

From an ecological perspective, it is useful to think of human impacts to marine habitats as fitting into three categories:

1. permanent loss (e.g., filling of a coastal wetland),
2. degradation (e.g., eutrophication), and
3. periodic disturbance (e.g. mobile gear).

The first results in a loss of habitat quantity, the other two in a loss of quality. All three may reduce the ability of a region to support fish, however they differ in that the first is irreversible, the second may or may not be reversible, and the third is generally reversible once the source of disturbance is removed. Recovery times for the second category depend on the nature of the agent causing the degradation (e.g., very slow for PCBs vs. relatively short for nutrients once the source of contamination has been removed) and the physical characteristics of the region (sediment type, hydrodynamics, etc.). Recovery times for the third category will vary depending on the intensity and periodicity of the disturbance (e.g., how frequently trawled) and the nature of the habitat itself. Superimposed on these human-related alterations are natural fluctuations in habitats, such as storms, and long term climatic changes.

HABITAT QUANTITY AND QUALITY

Habitat quantity is a measure of the total area available, while habitat quality is a measure of the carrying capacity of an existing habitat. Documenting the former is reasonably straightforward, particularly for some nearshore habitats, such as salt marshes and eelgrass beds. The extent and rate of actual loss of coastal wetlands have been well documented for parts of New England and elsewhere over the last few decades by a number of researchers and agencies using aerial photography and ground surveys (Dexter, 1985; Hankin et al., 1985; Costa, 1988; Tiner and Zinni, 1988; Dahl, 1990; Field et al., 1991; Tiner, 1991). Maps, however, do not indicate whether a current existing coastal wetland or seagrass bed still functions as it had historically.

Less is known about the distribution of offshore habitats in New England waters than nearshore habitats and about how such habitats have changed as a result of human and natural factors. Certainly scientists and fishermen have long been aware that certain underwater features, such

Table 5.2. List of fish species occurring in coastal Massachusetts waters and their relationship to estuaries. Fishery indicates species historically or currently taken in either commercial (C) or recreational (R) fisheries. Human impact effects are keyed to Table 5.1. Possible impacts on each species are based on documented cases and assessment of habitat requirements of individual species by the authors.

Common Name	Scientific Name	Fishery	Zone	Human Impact							
Freshwater				P	H	E	S	I	F	C	G
Banded killifish	*Fundulus diaphanus*		Benthic								
Blacknose shiner	*Notropis heterolepis*		Pelagic		x						
Bridle shiner	*Notropis bifrenatus*		Pelagic								
Brown trout (introduced)	*Salmo trutta*		Pelagic	x	x	x					
Chain pickerel	*Esox niger*		Pelagic	x		x					
White perch	*Morone americana*	R	Benthic	x	x	x				x	
Diadromous											
Alewife	*Alosa psuedoharengus*	C R	Pelagic	x	x	x				x	
American eel	*Anguilla rostrata*	C R	Benthic	x	x	x	x				
American shad	*Alosa sapidissima*	C R	Pelagic	x	x	x				x	
American smelt	*Osmerus mordax*	C R	Pelagic	x	x	x	x				
Atlantic salmon	*Salmo salar*	C R	Pelagic	x	x	x		x		x	
Atlantic sturgeon	*Acipenser oxyrhynchus*	C R	Benthic	x	x	x	x			x	
Blueback herring	*Alosa aestivalis*	C R	Pelagic	x	x	x	x			x	
Rainbow smelt	*Osmerus mordax*	C R	Pelagic	x	x	x				x	
Sea lamprey	*Petromyzon marinus*		Benthic	x	x	x					
Searun brook trout	*Salvelinus fontinalis*	C R	Pelagic	x	x	x					
Resident											
Blackspotted stickleback	*Gasterosteus wheatlandi*		Pelagic	x	x	x					
Fourspine stickleback	*Apeltes quadracus*		Benthic	x	x	x					
Grubby	*Myoxocephalus aenaeus*		Benthic								
Hogchoker	*Trinectes maculatus*		Benthic	x	x	x	x				
Inland silversides	*Menidia beryllina*		Pelagic	x	x	x	x				
Longhorn sculpin	*Myoxocephalus octodecimspinosus*		Benthic								
Mummichog	*Fundulus heteroclitus*		Benthic	x	x	x	x	?		x	?
Naked goby	*Gobiosoma bosci*		Benthic								
Ninespine stickleback	*Pungitius pungitius*		Benthic	x	x	x					
Northern pipefish	*Syngnathus fuscus*		Benthic	x	x	x					
Northern puffer	*Sphaeroides maculatus*		Pelagic								
Oyster toadfish	*Opsanus tau*		Benthic	x	x	x	x			x	
Rainwater killifish	*Lucania parva*		Benthic	x	x	x	x	?			
Sheepshead minnow	*Cyprinidon variegatus*		Benthic	x	x	x	x	?			
Shorthorn sculpin	*Myoxocephalus scorpius*		Benthic								
Smooth flounder	*Liopsetta putnami*	C R	Benthic	x	x	x	x				
Stickleback	*Gasterosteus spp.*		Benthic								
Striped killifish	*Fundulus majalis*		Benthic	x	x	x	x	?			
Threespine stickleback	*Gasterosteus aculeatus*		Benthic	x	x	x	x				
Nursery											
Atlantic menhaden	*Brevoortia tyrannus*	C R	Pelagic	x	x	x				x	?
Atlantic silversides	*Menidia menidia*		Pelagic	x	x	x					?
Atlantic tomcod	*Microgadus tomcod*		Benthic	x	x	x	x				
Bay Anchovy	*Anchoa mitchilli*		Pelagic	x	x	x					
Bluefish	*Pomatomus saltatrix*	C R	Pelagic	x	x	x					?
Cunner	*Tautogolabrus adspersus*	C R	Benthic	x	x	x	x				
Mullet	*Mugil cephalus*		Benthic			x	x				
Pollock	*Pollachius virens*	C R	Pelagic	x	x	x			?		
Striped bass	*Morone saxatilis*	C R	Pelagic	x	x	x	x			x	
Striped mullet	*Mugil cephalus*	C R	Benthic				x				
Summer flounder	*Paralichthys dentatus*	C R	Benthic	x	x	x	x		?		
Tautog	*Tautog onitus*	C R	Benthic	x	x	x					
Winter flounder	*Pseudopleuronectus americanus*	C R	Benthic	x	x	x	x				

Common Name	**Scientific Name**	**Fishery**	**Zone**	**Human Impact**							
Marine				**P**	**H**	**E**	**S**	**I**	**F**	**C**	**G**
American plaice	*Hippoglossoides platessoides*	C	Benthic								
American sand lance	*Ammodytes americanus*	C R	Pelagic	x	x		x				
Atlantic cod	*Gadus morhua*	C R	Benthic						?		
Atlantic herring	*Clupea harengus*	C	Pelagic						?		
Atlantic mackerel	*Scomber scombrus*	C R	Pelagic								
Atlantic needlefish	*Strongylura marina*		Pelagic								
Atlantic spiny lumpsucker	*Eumicrotremus spinosus*		Pelagic								
Atlantic wolffish	*Anarhichas lupus*	C	Benthic								
Barndoor skate	*Raja laevis*		Benthic						?		
Black seabass	*Centropristis striata*	C R	Benthic								
Blotched cusk-eel	*Ophidion grayi*		Benthic								
Butterfish	*Peprilus triacanthus*	C R	Pelagic								
Cobia	*Rachycentron canadum*		Benthic								
Conger eel	*Conger oceanicus*		Benthic								
Cusk	*Brosme brosme*		Benthic								
Daubed shanny	*Lumpenus maculatus*		Benthic								
Fourspot flounder	*Paralichthys oblongus*	C R	Benthic								
Goosefish	*Lophius americanus*		Benthic								
Haddock	*Melanogrammus aeglefinus*	C R	Benthic								
Inshore lizardfish	*Synodus foetens*		Benthic								
Little skate	*Raja erinacea*		Benthic								
Lookdown	*Selene vomer*		Pelagic								
Lumpfish	*Cyclopterus lumpus*		Benthic								
Mackerel scad	*Decapterus macarellus*		Pelagic								
Northern kingfish	*Menticirrhus saxatilis*	C R	Benthic								
Northern searobin	*Prionotus carolinus*		Benthic								
Northern sennett	*Sphyraena borealis*		Pelagic								
Ocean perch	*Sebastes marinus*		Benthic								
Ocean pout	*Macrozoarces americanus*	C	Benthic								
Ocean sunfish	*Mola mola*		Pelagic								
Offshore hake	*Merluccius albidus*	C	Benthic								
Permit	*Trachinotus falcatus*		Pelagic								
Radiated shanny	*Ulvaria subbifurcata*		Benthic								
Red hake	*Urophycis chuss*	C R	Benthic								
Rock gunnell	*Pholis gunnellus*		Benthic								
Scup	*Stenotomus chrysops*	C R	Benthic								
Sea raven	*Hemitripterus americanus*		Benthic								
Silver hake	*Merluccius bilinearis*	C R	Benthic						?		
Skates spp.	*Raja sp.*		Benthic								
Smooth dogfish	*Mustelus canis*	C	Benthic								
Smooth skate	*Raja senta*	C	Benthic								
Spiny dogfish	*Squalus acanthias*	C	Benthic								
Striped searobin	*Prionotus evolans*		Benthic								
Thorny skate	*Raja radiata*		Benthic								
Weakfish	*Cynoscion regalis*	C R	Benthic								
White hake	*Urophycis tenuis*	C R	Benthic								
White mullet	*Mugil cureme*	C R	Benthic								
Windowpane	*Scophthalmus aquosus*	C R	Benthic								
Winter skate	*Raja ocellata*		Benthic								
Witch Flounder	*Glyptocephalus cynoglossus*	C R	Benthic								
Yellowtail flounder	*Limanda ferruginea*	C R	Benthic								

Physical habitat loss; **H**ydrology changes; **E**utrophication; **S**ediment delivery changes; **I**ntroduced species; **F**ishing methods-physical impacts; **C**ontaminants; **G**lobal climate-changes in sea-level.

as Georges Bank, are attractive to diverse marine organisms and have had a general idea of the bottom types in different regions through bottom sampling by trawls and dredges. The recent development of sidescan sonar, high resolution seismic profiling and echo sounding, and video-equipped remotely operated vehicles (ROVs) and submersibles has stimulated mapping efforts in certain sections of Gulf of Maine, particularly the northeast section of Georges Bank, Stellwagen Bank, Jeffreys Bank, and Swans Island (Valentine and Lough, 1991; Valentine and Schmuck, 1994; Auster et al., 1996).

The degradation of habitat quality, such as through siltation and alteration of salinity, food webs and flow patterns, may be just as devastating to the biological community as a loss in quantity. Siltation caused by land-based erosion may smother a smelt spawning bed in a tidal river just as offshore deposition of dredged materials may smother the benthic eggs of Atlantic herring. The physical structure of the habitat does not need to be directly altered for negative consequences to occur. Anthropogenic alterations in the tidal flow to a marsh frequently results in the invasion of salt marshes by *Phragmites australis*, generally thought to be poorer as a habitat than the plants it replaces. Habitat loss and degradation are interrelated because habitat loss is the ultimate end point of gradual declines in habitat quality.

THE QUESTION OF SCALE

The effect of habitat loss on organisms depends to a large extent on the scale of the loss. A single small loss may not, in itself, cause an observable effect. However, the cumulative impact of many small losses may be quite significant at a regional scale. For example, the diverting or damming of one river, although locally important to the species in that river, may not have a regional impact. However, if enough water is diverted from the many rivers flowing to the Gulf of Maine, the coastal boundary current setup by freshwater discharge and important to the survival of many offshore fish could be altered (Townsend, 1991; Doyle et al., 1993). Similarly, the loss of one acre of salt marsh or the destruction of a small patch of cobble habitat may not have a detectable effect on fish, particularly where there are enough unaffected areas nearby to compensate for the loss. The cumulative impact of many small losses over time may, however, ultimately lead to a severe impact on a fishery.

As unimpacted habitats get smaller and more fragmented, the capacity of the populations to recover from natural catastrophic events as well as human-induced stress is likely reduced. The small incremental increase in a stress, such as an increase in nutrient loading from a new development, may be enough to turn the corner from habitat degradation to loss. Most environmental regulations governing developments require assessments only on a site-by-site basis and cumulative impacts are not considered. Management of fisheries occurs at a regional scale, but the changes in habitats that may ultimately lead to a severe impact on a fishery are the result of cumulative impacts at much smaller scales. Over a long enough period of time the sum of all the small changes may result in a large impact.

LOSS OF HABITAT QUANTITY

> *Nobody knows how much salt marsh existed along the Atlantic coast of North America before the Europeans arrived. No matter how much was here, the settlers began to change the existing amount almost immediately.*
>
> *-Teal and Teal, 1969*

FILLING AND OUTRIGHT LOSS OF COASTAL HABITATS

Loss of coastal wetlands due to residential and industrial development has been severe in the United States. Recent estimates suggest that about 54% of the nation's original 915,000 km^2 of wetlands (freshwater and coastal) have been lost (Tiner, 1984), and over half of the nation's original salt marshes and mangrove forests have been destroyed (Watzin and Gosselink, 1992). A particularly intense period of loss occurred between 1950 and 1970.

Boston Harbor is a local example of this

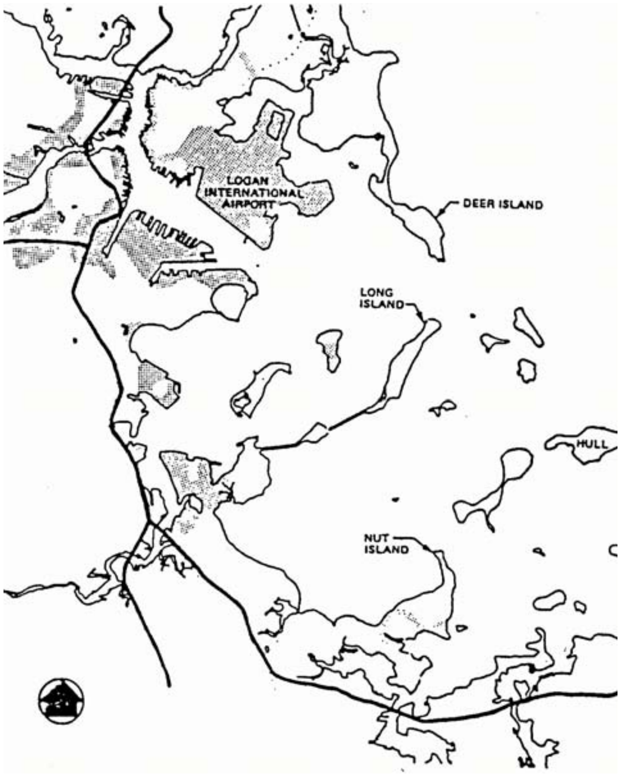


Figure 5.1. Alteration of Boston Harbor by filling-1775 to 1990 (from MAPC Boston Harbor Islands Comprehensive Plan). The grey patterned areas indicate landfill.

national trend (Figure 5.1). From 1775 through the 1970s, the physical shape of Boston Harbor has been drastically altered by filling. The part of Boston that is now the North End and Beacon Hill was originally a peninsula connected to the mainland by a narrow strip of upland (the Boston Neck) with extensive salt marshes. The Back Bay, a densely urban neighborhood harboring some of the tallest buildings in the city, was once a shallow estuary with extensive salt marshes. South Boston and Charlestown were also originally surrounded by tidal marshes that are now destroyed. Filling connected with building Logan airport alone amounted to 2,000 acres.

The wholesale loss of coastal habitats from human activities has been halted, in large part due to federal wetlands protection mandated under the Clean Water Act of 1972. At the state level, Massachusetts has one of the most stringent coastal wetlands protection programs in the country through its Coastal Wetland Protection Act of 1964 and the Wetlands Protection Act and Regulations, passed in the 1970s. Estimates of the change in the acreage of salt marshes that has occurred after the passage of wetlands regulations in Massachusetts indicate that losses vary from vitually zero in less developed parts of the Massachusetts coast to about 9% in a fourteen year period in one highly urban area (studies summarized in Buchsbaum, 1992). Of the approximately 15,500 acres of vegetated estuarine wetlands between Plum Island and Scituate, Massachusetts, about 24 acres (0.15%) were either converted to uplands or changed to nonvegetated wetlands from 1977 through 1985-1986 (Foulis and Tiner, 1994). Although this is a much smaller number than what occurred in the 1950s and 1960s, nonetheless small, incremental losses of coastal habitats still occur from public works projects that are exempt from regulations, from illegal filling, from boater activities, and from the construction of small docks and piers. The latter often involves dredging several hundred cubic yards of material and re-arranging the shoreline to one more suitable to human use.

There is less information on historical changes in acreages of subtidal eelgrass beds and subtidal and intertidal sand and mud flats than there is for salt marshes. Like salt marshes, these areas provide important habitats for a variety of fish and shellfish species (Whitlach, 1982; Thayer et al., 1984; Heck et al., 1989). Undoubtedly, many of these habitats were filled along with vegetated wetlands as population centers grew around estuaries. In addition to filling, these shallow, subtidal areas are affected by other human activities. Power boats, for example, can directly remove submerged vegetation and associated attached organisms (Thayer et al., 1984) and resuspend sediments from tidal flats.

Dredging Subtidal Benthic Habitats

Many New England coastal communities depend on dredging to maintain their harbors for recreational and commercial uses. From 1960 to 1981, enough material was dredged from Beverly, Chelsea, East Boston, South Boston, Gloucester, New Bedford, Fairhaven and Salem, to fill a one square mile hole 4.5 ft deep (Pierce, 1985). The act of dredging, as well as disposal of the dredged material, has an immediate affect on the benthic community. In the past, dredged materials were often used to fill salt marshes and tidal flats to create more dry land, so a number of coastal habitat types were affected by dredging. Much of this sediment had toxic levels of chemicals such as PCBs, PAHs, or metals, hence the impacts go beyond just physical alteration of the habitat. Dredge spoils have also been deposited in offshore disposal areas such as the Massachusetts Bay and Cape Cod Bay Disposal Sites. Although regulations now require that such dredged material meet certain standards in relation to contaminants to be considered suitable for offshore disposal, this practice clearly alters both the inshore dredged and offshore filled benthic communities.

Dredging causes losses of submerged aquatic vegetation, such as eelgrass, through both direct and indirect causes. Dredging shallow subtidal areas has resulted in the direct loss of eelgrass in a number of areas around the coast (Costa, 1988). Declining light penetration and smothering associated with the turbidity plume caused by dredging also cause indirect losses in submerged aquatic vegetation (Thayer et al., 1984). Since they are not as well-delineated as intertidal habitats, eelgrass meadows are more likely to be inadvertently dredged even with current wetlands protection regulations.

Dredging alters intertidal and subtidal habitats by making some areas deeper by sediment removal and others more shallow by filling. In addition to being primary habitats for important shellfisheries, shallow subtidal and intertidal flats are important feeding areas for many flatfish, including winter, summer, and windowpane flounders (Pearcy, 1962; Tyler, 1971a; Bergman et al., 1988; Saucerman, 1990; Saucerman and Deegan, 1991; Ruiz et al., 1993). Dredging eliminates feeding sites for many fish by altering the species composition of the invertebrate prey within the benthos (Whitlatch, 1982). Tyler (1971b) suggested that the loss of tidal flats in the Bay of Fundy through dredging could reduce winter flounder populations by altering food availability.

SHORELINE MODIFICATION

The use of sea walls and bulkheading results in an alteration of intertidal habitat and associated communities (Whitlatch, 1982). Construction of hard surfaces along the coast transforms a soft bottom community to a hard bottom and usually creates a more narrow intertidal and shallow subtidal zone. There are also cascading effects on marsh and tidal flats due to changes in sediment transport processes and long-shore currents. In the long term, development of shorelines may prevent the normal migration of salt marshes inland as they respond to rising sea levels.

A recent concern has been the effects of shading from docks and piers on eelgrass communities. A recent study carried out in Waquoit Bay on Cape Cod indicated that eelgrass plants under docks were lower in a number of growth parameters compared to outside controls (Burdick and Short, 1995). Shading effects from docks is also believed to favor algae-dominated communities over eelgrass. Algae-dominated communities do not support the diversity and biomass of fishes that are typical of submerged aquatic vegetation habitats (Deegan et al., 1997).

DECLINE IN HABITAT QUALITY

The loss of habitat quality or carrying capacity is more subtle than the loss of area. The term habitat quality is used to refer to the functional attributes of an area, such as providing food or shelter, needed to support fish and shellfish. Habitat quality is altered by a variety of factors, including sedimentation, nutrients, toxic chemicals, physical disturbance, and colonization by aggressive alien species of plants and animals. Sources of these impacts include urban sewer systems, industrial outfalls, ocean dump sites, individual septic systems, storm water runoff, agriculture, fishing gear, and the atmosphere. The degradation of habitat quality affects a range of ecological processes: primary and secondary production, trophic dynamics, succession, and species diversity.

Boston Harbor was an example of how a multitude of impacts, including toxic organic contaminants, heavy metals, nutrient loading, and sedimentation, altered the benthic food web. Approximately 60% of the bottom area of Boston Harbor had a benthic invertebrate community that was either moderately or severely impacted by pollution. This trend has been at least partially reversed with the upgrade of the wastweater treatment facility in the late 1990s (Rex et al., 2002).

EUTROPHICATION

One of the major causes of habitat decline associated with human activities along the coast is the increase in nutrient loading, especially nitrogen. In general, the effects of increased eutrophication are negative. Elevated levels of nutrients running into a bay from lawn fertilizers, agricultural fields, and sewage, stimulate primary production, resulting in increased growth of phytoplankton and macroalgae, reduced water clarity, and alteration of the water chemistry. The algal species composition changes to a dominance by species that are not readily incorporated into existing trophic structures (Paerl, 1988). Nuisance macroalgae accumulate in shallow waters (Lee and Olsen, 1985; Valiela et al., 1992) and the abundance of rooted aquatic vegetation declines due to shading by attached or floating algae (Orth and Moore, 1983). If the growth of algae exceeds the ability of higher trophic levels to consume it, the excess biomass accumulates on the bottom where it is decomposed by microbes. Subsequent effects may be low dissolved oxygen events, changes in the species of plants and animals present, and loss of critical habitats such as seagrass beds. These effects are likely to affect small consumer organisms such as zooplankton and amphipods, as well as the fishes that depend on these consumers for food. Changes in the food web may alter species composition at the higher trophic levels, from those desirable by humans as food, such as flatfish, to less desirable species such as gelatinous ctenophores or sea nettles (Purcell, 1992; Caddy, 1993).

LOW DISSOLVED OXYGEN

Decline and even loss of habitat due to inadequate dissolved oxygen (DO) is one of the most severe problems associated with eutrophication of coastal waters. Depletion of some (hypoxia, < 2 mg/L) or all (anoxia) of the oxygen in the water or the sediments causes changes in community composition and even death of organisms. Although some degree of oxygen depletion can occur in natural systems, even in offshore basins, oxygen depletion has been exacerbated by increased sewage and increased nutrient inputs resulting from development and agriculture around estuaries. Oxygen is used up by the respiration of bacteria as the excess algae and other organic matter decays.

Areas that are particularly prone to hypoxia include coastal ponds, subtidal basins, and salt marsh creeks. Water circulation in coastal ponds is often restricted, and in some ponds may be completely isolated from the sea for long periods of time. The coastal ponds on the south shore of Cape Cod and Rhode Island are rich habitats for estuarine fish and shellfish. However, the surrounding watersheds of many ponds have undergone rapid growth in the past thirty years, and they now are routinely hypoxic in summer (Lee and Olsen, 1985; D'Avanzo and Kremer, 1994). Communities surrounding these ponds often propose regular opening of these ponds to the sea as a way of “rejuvenating” fish and shellfish habitats. The bottom waters in subtidal basins may be isolated from the well-oxygenated surface waters during periods of stratification in the summer. For example, the dissolved oxygen levels in Stellwagen Basin occasionally drop below 6.0 mg/L when the waters are stratified (Kelly, 1993).

One approach to understanding the loss of habitat due to low DO is to map the amount of area in a bay that does not meet minimum DO standards (5 mg/L according to EPA standards, 5.6 mg/L in Massachusetts). Parts of the Inner Harbor in Boston frequently violate the 5 mg/L standard and the Charles River Basin often is completely anoxic because of inputs from combined sewer outflows and altered hydrology (Rex et al., 1992). In Chesapeake Bay and western Long Island Sound, extensive areas of the bottom often fails to meet minimum DO levels required for survival of fishes and invertebrates (Welsh and Eller, 1991; Breitburg, 1992).

Even short episodes of low DO can have strong effects on fish populations. A severe fish kill in Waquoit Bay was caused by one low DO event which lasted less than 24 hrs (D'Avanzo and Kremer, 1994). Juvenile winter flounder which used the Head of the Bay site as a nursery area were killed and washed up on the beach along with shrimp, crabs, and other fish and invertebrates (Figure 5.2). The juvenile winter flounder population in the anoxic location did not recover, although populations at other sites which did not have a low DO event were unaffected. If the low DO event had been more widespread, it could have caused the failure of an entire year-class for the population

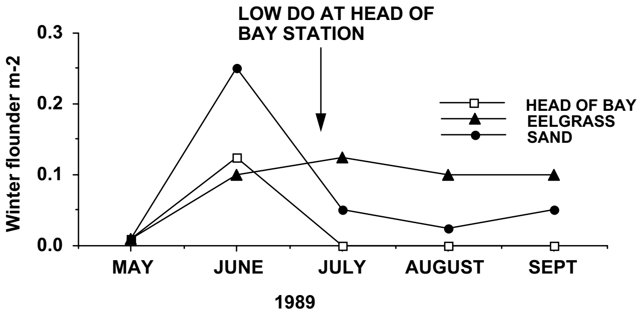


Figure 5.2. Juvenile winter flounder abundance at three habitats in Waquoit Bay. This graph illustrates the effect of a local, 24 hr low dissolved oxygen event on the abundance of juvenile winter flounder. Juvenile winter flounder at the Head of the Bay site were killed by a low dissolved oxygen event that lasted less than 24 hrs (D'Avanzo and Kremer, 1994). Populations at other locations (eelgrass, sand) that did not experience a low DO event were unaffected.

of this estuary, even though it lasted less than 24 hours out of the entire year.

Low, but not lethal, levels of dissolved oxygen can also lower the growth and survivorship of fishes and impact shellfish. Growth of juvenile winter flounder held at oxygen concentrations of 6.7 mg/L was twice that of fish held at 2.2 mg/L (Bejda et al., 1992). Fish held under conditions of diurnally fluctuating DO also showed growth suppression compared to fish held at high DO levels. Some behaviors of young fish such as moving up into the water column and increased swimming activity are also increased under low DO levels making them more susceptible to predation (Bejda et al., 1987). Shellfish growing under low oxygen conditions are stressed (see Brousseau, Chapter 6).

Not all coastal waters are equally susceptible to low oxygen conditions. In general, low oxygen conditions are rare in surface waters and rare in the winter. Low oxygen is most likely to occur in bottom waters at night in the summer because of warm temperatures, high metabolic sediment demand and water column stratification. Areas that are vertically well mixed and well flushed by the tides, which includes most of the New England coastal embayments north of Cape Cod, rarely experience hypoxia. Estuaries in this region may exchange more than 50 percent of their water with well-oxygenated seawater every day. Low DO is not considered a widespread problem in Boston and other urban harbors north of Cape Cod, although it can be locally important.

LOSS OF EELGRASS

The alteration and loss of eelgrass habitats due to eutrophication provides a good example of the effect of habitat degradation on fish communities. The high diversity and abundance of invertebrates and fish in eelgrass ecosystems is due to: (1) increased survivorship due to the physical structure of vascular plants that provides protection from predation (Orth et al., 1984; Bell et al., 1987; Pohle et al., 1991) and (2) greater supply of food (Heck and Crowder, 1991; Deegan et al., 1993). The function of eelgrass and other vegetated habitats in shallow waters parallels that of the cobbles and biogenic structures in deeper offshore waters that are now of such concern because of mobile gear impacts (Auster and Langton, 1999). Thus an examination of studies of how the alteration of the structure of eelgrass habitats has affected fish communities may provide insights for those more offshore habitats as well.

Because eelgrass beds are subtidal, they require relatively clear water so that they have enough light for growth, hence they are sensitive to sediment loading and to eutrophication. In the past twenty years, eelgrass losses related to declining water quality have been documented for Southern Massachusetts (Costa, 1988; Short et al., 1993; Deegan et al., 1993) and Long Island Sound (Rozsa, 1994). Historical records indicate that eelgrass was once widespread in Boston Harbor, but now only a few limited areas (Hingham Harbor) have suitable light penetration to support eelgrass (Chandler et al., 1996). For most of the remainder of the New England coast, there is little documentation of overall trends in the abundance of this habitat, although short term fluctuations have been noted (Short et al., 1986).

Increased nutrient loading causes declines in the habitat quality of submerged aquatic vegetation and eventually complete loss of large areas of this habitat (Costa 1988; Batiuk et al., 1992; Deegan et al., 1993; Short et al., 1993). Eutrophication alters the physical structure of seagrass meadows by decreasing shoot density and blade stature, decreasing the size and the depth of beds, and by stimulating the excessive growth of macroalgae (Short et al., 1993). The initial response of eelgrass to low levels of nitrogen may be positive where the eelgrass itself is nitrogen-limited (Short, 1987). Macroalgae and phytoplankton, however, are able to transform excess nitrogen into growth more rapidly then eelgrass, so they eventually outcompete and smother the eelgrass (Short et al., 1993).

Another typical consequence of eutrophication within an estuary is a geographic shift of eelgrass habitats towards shallower high saline areas near the mouth of the estuary. Eelgrass in deeper areas die first because the plants are very sensitive to reduced light levels (Dennison, 1987; Costa, 1988). Beds near the head of the estuary are also highly susceptible to the effects of eutrophication because of high nitrogen concentrations resulting from the close proximity to runoff from land.

Eelgrass is subject to intense natural population fluctuations which may be exacerbated by eutrophication. An epidemic of wasting disease in the early 1930s wiped out most eelgrass along the entire east coast (Rasmussen, 1977). Eelgrass recovered to some extent, although it never recolonized some former areas. The disease has periodically recurred in local embayments in more recent years (Dexter, 1985; Short et al., 1986). Eutrophication may increase the susceptibility of the plants to the wasting disease by decreasing the plants ability to resist disease and by restricting the plants distribution within the estuary (Buchsbaum et al., 1990; Short et al., 1993). In the past, low salinity areas of the estuary at the upper reaches of estuaries have been a refuge from the disease during outbreaks. As mentioned above, however, eutrophication tends to be most severe in the low salinity, upper reaches of estuaries. This restricts eelgrass to the more saline areas where it is most susceptible to wasting disease.

The replacement of eelgrass by macroalgae in a eutrophic estuary results in a highly modified fish community even before the plants themselves have disappeared (Deegan et al., 1997). In studies of Waquoit Bay and Buttermilk Bay on Cape Cod, assessments of the relative degradation of eelgrass habitats were based on year-round measurements of chemical and physical characteristics (e.g., algal blooms, macroalgae, low DO, high nutrients, dredged channels). Habitats that had moderate water quality had more individuals, more biomass, more species and a higher diversity of fish than areas with low water quality (Figure 5.3). The number of fish species that use estuaries as a nursery area or spawning location was much lower in areas of poor water quality compared to areas with moderate water quality (Figure 5.4). Fish associated with the benthic zone were more strongly affected than fish associated with the pelagic zone. For example, winter flounder, a benthic species that spawns in the estuary, was one of the first species lost from eelgrass habitats under eutrophic conditions.

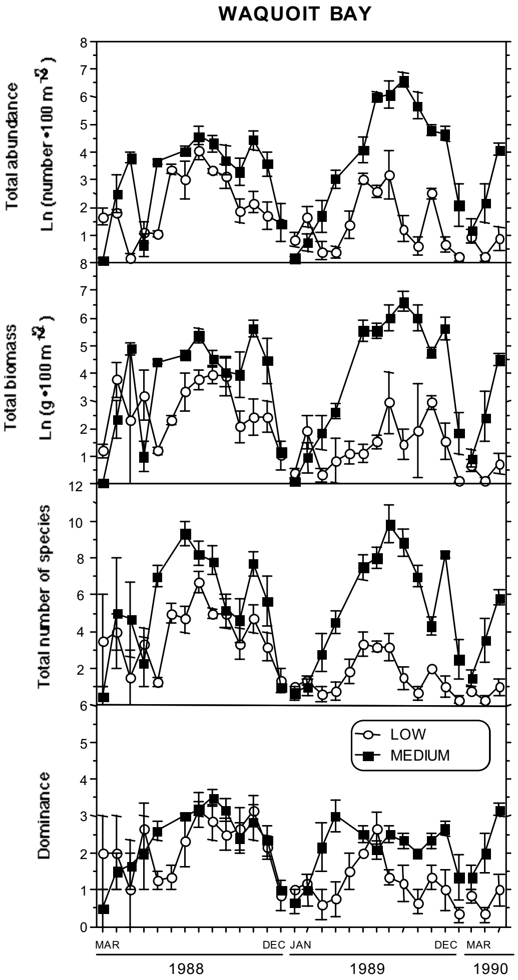


Figure 5.3. Fish abundance, biomass, number of species and dominance all are lower in low compared to medium quality eelgrass habitats in Waquoit Bay, MA (Deegan et al., 1997). *Reprinted with permission, Estuarine Research Federation.*

The largest impact of loss of habitat quality on fish community structure was found in the late summer periods as a result of the cumulative effects of habitat degradation (Figures 5.3-5.4).

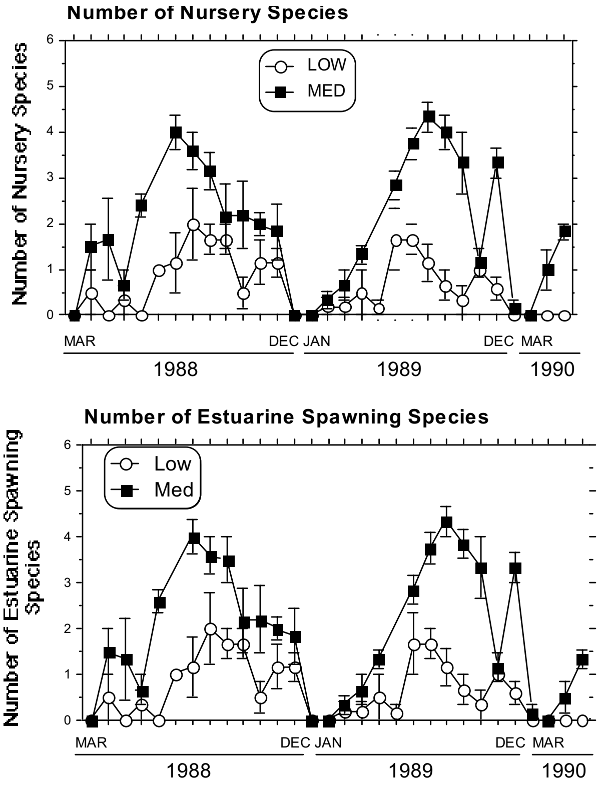


Figure 5.4. Number of species using eelgrass habitats as nursery areas and the number of species that spawn in estuaries were lower in low compared to medium quality eelgrass habitats in Waquoit Bay, MA (from Deegan et al., 1993).

Many species of fish migrate into estuaries in the spring and summer as adults to spawn and feed and as juveniles to find protection and ample food before returning to the open ocean as adults. By the end of the summer, when the fish community has experienced the cumulative effects of low oxygen, higher mortality due to predation and disrupted food webs, there were fewer species, fewer individuals and lower biomass in areas of high anthropogenic stress compared to less disturbed areas (Deegan et al., 1997). This result is likely a combination of lowered fish production, higher mortality, and migration away from degraded habitats.

Declines in eelgrass habitats and commercially or recreationally important finfish were apparent in New England over a 30 year time period in Waquoit Bay (Table 5.3). Comparison of the number of species and the percent composition of the catch in eelgrass habitats in Waquoit Bay indicate a sharp decline in the number of species and abundance of recreationally or commercially important species.

Table 5.3. Changes in the fish community composition of eelgrass habitat in Waquoit Bay over time. Data are from Deegan et al., 1997 and Curley et al. 1971.

	Number of Species			% Composition		
Year	1967	1988	1995	1967	1988	1995
Fishery	10	7	3	63	2	9
Forage	14	13	6	37	98	81
Total	24	20	9	100	100	100

Species common in commercial and recreational fisheries, such as winter flounder, white hake and pollock declined during this time period. The number of fisheries species declined from 10 to 3, and the percent composition of the catch declined from 63% to 9% between 1967 and 1995. A decline in the number of forage species is also apparent which indicates that the loss of fisheries species is probably not due simply to overfishing. The number of forage fish declined from 14 species to 6 species over the same 30 year period. Some small forage fish, such as sticklebacks and silversides, increased in abundance and forage fish now account for roughly 80% of the catch in eelgrass areas.

Direct links between loss of eelgrass and loss of commercial catch have been demonstrated in other areas of the world. Jenkins et al. (1993) demonstrated a clear connection between a 70% loss of seagrass and a 40% loss in total commercial fish catch in Western Port Bay, Australia. The strong, parallel decline in fish catch and seagrass loss occurred in species which were specifically adapted to life in a seagrass habitat. Species with a reduced ecological link did not show a clear parallel decline.

Along the east coast of the United States, the closest connection between eelgrass and a commercially important marine species is not with a fish but is with the bay scallop. The larvae of these bivalves settle on eelgrass blades prior to their transformation into adults (Pohle et al., 1991; Brousseau, Chapter 6). The harvest of bay scallops declined drastically during the wasting disease epidemic of the 1930s (Thayer et al., 1984). In addition to bay scallops, there is some indication that gadoids in coastal regions make use of eelgrass habitats at certain life stages. Chandler et al. (1996)

found that two to three year old pollock were caught more often within eelgrass beds then in neighboring, unvegetated habitats in Boston Harbor. Pollock also preferred vegetated substrate over sand in Great South Bay, Long Island (Briggs and O'Connor, 1971). Tupper and Butilier (1995) reported higher growth rates of age-0 Atlantic cod in eelgrass compared to sandy areas, cobble habitats, and underwater reefs in Nova Scotia. Survivorship in eelgrass was lower than in cobble and underwater reefs but higher than in sandy habitats. Studies of eelgrass beds along the Danish coastline in the early 1900s concluded that eelgrass was an important habitat for juvenile cod (Peterson and Boysen-Jensen 1911; Peterson, 1918) as have more recent studies in Nova Scotia (Tupper and Butilier, 1995).

Most of the fish that currently use eelgrass as a habitat in New England are not directly taken in commercial or recreational fisheries, but are forage fish or prey for species taken in fisheries (Deegan et al., 1997; Heck et al., 1989, 1995). Because much of the current eelgrass habitat is arguably degraded by pollution to some extent already, establishing a direct link between declines in commercial or recreational finfish fisheries and loss of eelgrass habitat in New England using existing areas will be difficult. Unfortunately, information on fish use of eelgrass areas in New England prior to the 1930's is lacking, making it difficult to establish if eelgrass was an important habitat for fisheries species prior to the onset of the wasting disease epidemic and habitat degradation.

FISHING ACTIVITIES

FISHING GEAR

The ongoing concern about the impacts of mobile fishing gear on benthic communities in fishing grounds has been reflected in a number of recent symposia, reviews, and edited volumes (e.g., Dorsey and Pederson, 1998; Auster and Langton, 1999; Watling and Norse, 1999; NRC, 2002). The issue is potentially very important since the level of disturbance represented by mobile gear is widespread and intense. The development of roller gear through the 1980s and 1990s has allowed trawlers access to rocky and cobble habitats that were formerly inaccessible. As a result, virtually all benthic habitats are now potentially susceptible to the effects of mobile gear. We summarize the major scientific issues in the next few paragraphs and suggest that the reader refer to one of the recent reviews for more details.

Mobile fishing gear such as otter trawls, scallop rakes, and clam dredges are used in a variety of nearshore and offshore habitats to harvest demersal and benthic species. These can change the physical habitat and biological structure of ecosystems and therefore have potentially wide ranging impacts on a number of ecological levels. Studies have already documented that mobile gear reduces benthic habitat complexity by removing or damaging the actual physical structure of the seafloor, and causes changes in species composition of infauna, i.e., smaller invertebrates that live in the upper layers of the sediment and are prey for groundfish (Dayton et al., 1995; Auster and Malatesta, 1995; Auster et al., 1996; Collie et al. 1997; Auster and Langton, 1999; Engel and Kvitek, 1999). Mobile gear may also change surficial sediments and sediment organic matter, thereby affecting the availability of organic matter to microbial food webs (Pilskaln et al., 1999; Schwinghamer et al., 1999). Of major direct concern to commercial fish interests is the potential impact that the loss of benthic structural complexity may have on the survival of juvenile groundfish. From an ecosystems perspective, the simplification of the physical structure in repeatedly trawled areas would likely result in lowered overall biodiversity. The level of trawling in New England waters is intense. Auster et al. (1996) estimated that since 1976, the annual areal extent of trawling on Georges Bank has been equivalent to two to three times its entire bottom area. Some specific locations are trawled as much as 40-50 times per year (Auster and Langton, 1999).

Trawling results in a loss of habitat complexity through the removal of both biogenic structures, such as sponges, bryozoans, and shell aggregates and sedimentary features. This is important to fisheries since physical structure may be critical to the survival and growth of different fish species. Many taxa, especially juvenile fish, exhibit facultative associations with microhabitat features such as biogenic depressions, shells, burrows, sand wave crests, and even patches of amphipod tubes in low topographic environments such as subtidal areas of Massachusetts Bay (Lough et al., 1989; Langton and Robinson 1990; Auster et al., 1991, 1994, 1995, and 1996; Malatesta et al., 1992; Walters and Juanes, 1993; Tupper and Boutilier 1995). Cobble-gravel over sand-mud, for example, is a primary habitat for juvenile lobsters. This habitat may be a bottleneck for the recruitment of early benthic phase lobsters, as well as other shelter seeking species such as Jonah (*Cancer borealis*) and rock (*Cancer irroratus*) crabs (Wahle and Steneck, 1991). Late juvenile silver hake, (*Merluccius bilinearis*), showed a positive association with amphipod tubes in flat sandy areas (Auster et al., 1994, 1995). Postlarval silver hake may occur in patches of dense amphipod tube cover to avoid predators and to be near preferred prey (i.e., amphipods and shrimp). Similar associations have been found for Atlantic cod (Gotceitas and Brown 1993) and yellowtail flounder (Walsh 1991, 1992). In laboratory studies, Lindholm et al. (1999) found that predation on age-0 cod was significantly lower when the bottom was covered by emergent epifauna, such as is present in an untrawled area, compared to bare sand. Destruction of benthic organisms, such as amphipods and worms, by trawling alters food availability and microtopography which could subsequently affect the growth and survival of juvenile fishes.

In nearshore habitats, vegetation, such as eelgrass and kelp, and boulders and cobbles provide the microtopography that provides a refuge and foraging area for juvenile fish and macroinvertebrates. In deeper habitats, the microtopography is created by worm and amphipod tubes, sponges, and other biogenic features along with the boulders and cobbles. Mobile gear flattens this relief in both areas.

Trawl fishing not only changes the physical character of the seafloor, but also increases turbidity and resuspension of bottom sediments (Auster and Langton, 1999; Pilskaln et al., 1999). Sidescan sonar shows that physical disturbance to surficial sediments by trawling, as evidenced by abundant and persistent trawl furrows, is extensive on the seafloor of heavily fished areas within the Gulf of Maine (Jenner et al., 1991; Valentine and Lough, 1991). Trawling generates a plume of suspended sediment which increases turbidity and may alter sediment composition if the finer particles are swept away on water currents.

Some habitats are more sensitive to the effects of fishing gear than other habitats. In a study of the physically stressed intertidal zone of Minas Basin, the impacts of otter trawling were found to be minor (Brylinsky et al., 1994). These conclusions, however, cannot be assumed to be true for subtidal habitats with more diverse assemblages of benthic organisms and lower levels of natural disturbance (Sainsbury et al., 1993). Daan (1991), on the basis of production/biomass ratios, suggested problems might be most severe in heavily fished areas, subtidal areas, or for long-lived organisms.

Auster and Langton (1999) presented a conceptual model in which the impact of mobile gear on habitat complexity increases with fishing effort, but the extent of increase depends on the habitat type. More complex habitats, such as piled boulders and cobbles with epifauna show the steepest decline in habitat complexity with increased fishing effort. Their model predicts that cobbles and gravel with no epifauna would show little decrease in habitat complexity with increased fishing effort, since the effect of a trawl there would be to turn over structures, but the pre-existing structures would still be present afterwards. Recently a comparative risk assessment that integrates the size, severity,sensitivity and uncertainty of the impact of trawling on the seafloor has been developed (NRC, 2002).

The impacts of mobile gear are not limited to offshore habitats. Clam harvesting by raking and mechanical harvesting (“clam kicking”) has a severe and long-lasting effect on seagrass ecosystems (Peterson et al., 1987). Seagrass biomass in mechanical harvesting treatments fell by ~65% below controls. Recovery did not begin until more than 2 years had passed and seagrass biomass was ~35% lower than controls 4 years later. This could have severe impacts on fish and shellfish that depend on seagrass as settling locations or for protection from predators.

The extent to which trawling, dredging, and other fishing activities have contributed to the decline in fisheries or would impede recovery of overfished species in New England is still an area of debate among fisheries managers, the industry, and scientists. The evidence at the moment is indirect, in that the losses of structural features of the benthic community that are known to be important to a number of commercial fish species at some life stages have definitively been observed as a consequence of mobile fishing gear. Although the evidence does not indicate that the current fisheries crisis has been caused in large measure by the habitat effects of mobile gear, the major concern is with how these habitat impacts may affect recovery. At the current low population levels of many commercial groundfish it is possible that increased predation on juvenile groundfish in habitat impacted by dragging could be hindering recovery. It is logical to assume that an activity carried out over such a wide area and that impacts juvenile survival will ultimately affect fish populations at low population levels. It is clearly an area where more research is needed, particularly on how trawling affects the survival of juvenile fish and on the impacts and recovery periods of different bottom types under different intensities of trawling.

BYCATCH

Discards of bycatch, i.e., non targeted species or undersized individuals, can also have profound effects on fisheries habitat. We cannot do justice to this complex topic here, however it needs to be mentioned. Many individuals discarded as bycatch do not survive after being released. In addition to the obvious direct effects on populations and marine food chains of the loss of a large number of individuals, the disposal of large quantities of dead bycatch may alter the organic matter loading and cause changes in dissolved oxygen profiles and nutrient cycling.

HYDROLOGICAL ALTERATIONS OF ESTUARIES

CHANGES IN FRESHWATER INPUTS

Fisheries yields of coastal species have repeatedly been correlated with freshwater inputs (Aleem, 1972; Sutcliffe, 1973; Deegan et al., 1986; Nixon, 1992). Freshwater flow diversion, regulation and alteration by control structures and changes in land use have caused serious damage to estuaries worldwide (Clark and Benson, 1981; Rozengurt and Hedgepeth, 1989; Hancock, 1993). Virtually every river flowing into Massachusetts and Cape Cod Bays has an altered hydrograph due to control structures or changes in land use (Rebeck and DiCarlo, 1972). Regulation of river runoff for the production of hydropower, domestic and industrial use, and agriculture reduces the volume and alters the timing of freshwater delivery to estuaries. The flow of the Ipswich River, for example, is reduced by about half due to water withdrawals for human uses (K. Mackin, Ips. Riv. Watershed Assoc., pers. comm.). The type of land use in a watershed is also a strong determinant of the quantity, timing, duration and chemical composition of freshwater inputs to estuaries (Hopkinson and Vallino, 1995). Urbanization, for example, affects the timing and magnitude of river discharge after a rainstorm (Figure 5.5). Urban areas have large expanses of impervious surface, such as roads or parking lots, which causes more water to flow off the land more quickly than if the land were forest or field.

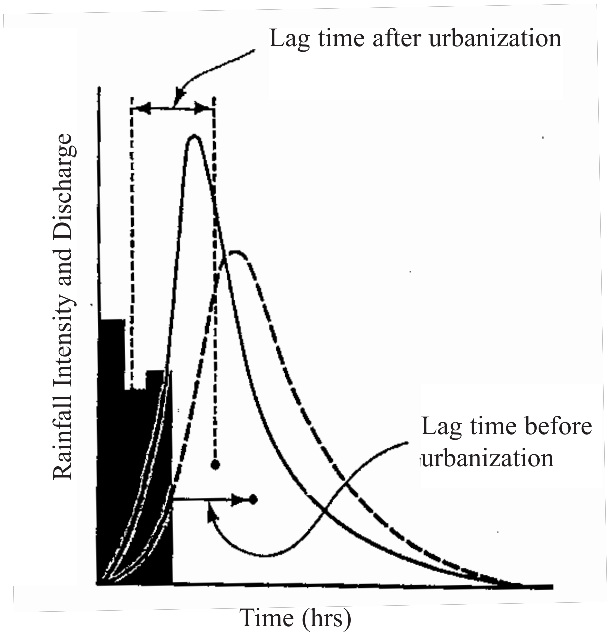


Figure 5.5. Alteration in volume, timing, duration and intensity of freshwater inputs with increased urbanization in the watershed (from Dunne and Leopold 1978).

Such reductions and alterations of freshwater inputs affect water circulation and the chemical properties of estuaries. Diversion of freshwater increases the salinity of coastal marine ecosystems and can diminish the supply of sediments and nutrients to coastal systems (Boesch et al., 1994). Habitats may change in response to altered hydrodynamics. Alteration of the natural hydroperiod can affect estuarine circulation on different time and magnitude scales, including short-term (diel) and longer term (seasonal or annual) changes. Salinity and sedimentation rates have a marked effect on the type and rate of wetland habitat present. In addition, organisms themselves often have specific salinity, temperature or habitat requirements for spawning or successful growth during juvenile stages.

Many fish species depend on the development of a counter current flow set up by freshwater discharge to enter estuaries as larvae or early juveniles (e.g., Pearcy, 1962; Townsend and Graham, 1981; Wipplehauser and McCleave, 1987; Day et al., 1989). Counter current flow is the input of water at the bottom from the coastal ocean to counterbalance the outflow of freshwater at the surface. A high freshwater discharge causes a strong saltwater influx which carries many species into estuaries (Kaartvedt and Svendson, 1990). Pearcy (1962) found that larval winter flounder changed their depth distribution between day and night and this, coupled with the counter-current flow, concentrated them in estuaries. As freshwater inputs to estuaries are lessened with increased freshwater withdrawals in the watershed, longitudinal and vertical estuarine habitat structure is altered and larval transport can be disrupted (Dadswell et al., 1987). Other examples of New England species likely to be affected by alterations in counter current flow are American eel, striped bass, white perch, Atlantic herring, blue crabs, lobsters, Atlantic menhaden, cunner, tomcod and rainbow smelt.

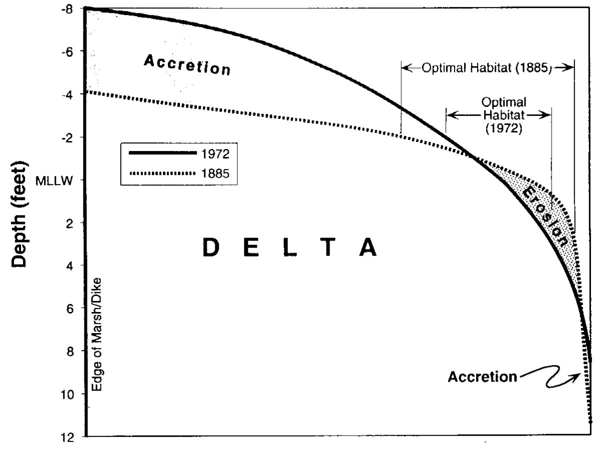


Figure 5.6. Loss of optimal fish habitat due to diversion of freshwater and resulting alteration of sediment load and hydrology (Jay and Simenstad, 1994). *Reprinted with permission, Estuarine Research Federation.*

Dams and Roadways

Dam construction on tidal rivers has caused habitat degradation within estuaries. Changes in flow, sediment delivery, salinity, and temperature result in changes in estuarine community structure, water chemical composition, food webs and loss of freshwater and estuarine habitats. Withdrawal or diversion of 40% of the annual runoff of the Skokomish River (Washington State) has resulted in a 6% loss of total unvegetated flats, more than 40% loss of low intertidal area, 18% loss of eelgrass area and a reduction in the size of the mesohaline mixing zone (Jay and Simenstad, 1994). One result was a 40% loss of optimal fish habitat between 1885 and 1972 due to changes in sediment load and distribution (Figure 5.6). In this case, sediment transport was the critical link between upstream alterations and the remote, downstream estuarine consequences. Dams also affect the migratory paths of fishes due directly to blocking (Moring, Chapter 3) and also to changes in the distribution, or local extinction, of prey species or alteration of temperature and salinity regimes.

Many marshes are fragmented and hydrologically isolated by roads, causeways, railroad beds, and dikes. Restricted water circulation results in declines in primary productivity and fish use of these habitats (Roman et al., 1984; Rozas et al., 1988).

Mosquito Control

Many salt marshes along the east coast of the United States are lined with mosquito control ditches. Their effect on fish that use salt marshes is not clear. By increasing the amount of water penetrating into the vegetated surface of the marsh, such channels may increase the use of marsh surfaces at high tide by foraging fish, such as mummichogs (Rozas et al., 1988). Conversely, negative effects on salt marsh fish would occur where ditches drained salt pannes and fish habitat dried out.

Power Plants

The impacts of power plants on estuaries and fisheries has been the subject of extensive reviews (e.g., Uziel, 1980; Larsen, 1981; Hall et al., 1982; Boynton et al., 1982; Summers 1989; Reeves and

Bunch, 1993). Power plants can affect fisheries by: 1) altering water circulation patterns by water withdrawal and changing water temperature, 2) altering estuarine production cycles through changes in water temperature and circulation patterns, 3) increasing death, decreasing growth and altering spawning because of elevated water temperatures, 4) increasing mortality by direct impingement of larvae and juveniles on intake screens, 5) increasing mortality and decreasing growth by releasing contaminants such as chlorine, bromine, copper and zinc, and 6) increasing mortality of fisheries species by direct impingement of their forage species. For example, Summers (1989) found that striped bass, bluefish and weakfish could experience significant losses (>25%) to total population production due to high levels of forage fish entrainment by power plants.

There has been interest in using the intense tides in the Bay of Fundy and other macrotidal estuaries throughout the world as a source of hydroelectric power. A major issue is the potential affect on fish attempting to pass through the turbines. Dadswell and Rulifson (1994) estimated a mortality of 20-80% of fish, depending on the species, passing through a low head tidal turbine on the Annapolis River estuary in the Bay of Fundy. Dadswell (1996) estimated that the annual shad spawning run on the Annapolis River has declined by over 50% in a fourteen year period since the installation of the hydroelectric plant despite the absence of local commercial fishing. The mean size, length of males and females, the mean and maximum age of individuals, and the percentage of repeat spawners have all declined in the shad run during the same time period.

Sea Level Rise

Global change and the rising sea levels could have major impacts on estuarine fish populations and coastal fisheries (Kennedy, 1990; Bigford, 1991). If sea level rises faster than the ability of salt marsh surfaces to accrete sediment and peat, then a greater amount of the surfaces of salt marshes will be regularly flooded during high tides in the future. This would provide increased habitat for estuarine fish if the marsh maintains its stability. The small change in sea level that has occurred in the last 50 years, however, is not sufficient to have caused the current dramatic declines in fisheries.

Exotics

The introduction of exotic species may also have profound effects on habitat quality by affecting predation and competition interactions. Carlton (1993) listed 13 different marine organisms that have been introduced into New England coastal waters since colonial times, including two crabs, a bryozoan, five mollusks, four sea squirts, and one red alga. Some of the exotics, such as the European periwinkle, *Littorina littorea*, and the green crab, *Carcinus maenus*, have been with us so long that few people realize they are not native. Many of these were carried to New England waters as fouling organisms on boats or in the ballast water.

This is a relatively new area of research, so there are little conclusive data on the ecological impacts of exotics and how they might effect fish and shellfish habitat. Some of the impacts are likely to be quite profound. Green crabs, are voracious predators on soft-shelled clams and newly settled winter flounder, and interfere with attempts to transplant eelgrass. The non-native haplotype of *Phragmites australis* has pushed out native species of salt and brackish marsh plants and may increase the rate of sedimentation in marshes, reducing the amount of intertidal habitat available to marsh fish (Able et al., 2003). Zebra mussels, which have colonized oligohaline as well as freshwaters in the Hudson River basin (Mills et al., 1996), have had strong effects on freshwater phytoplankton and zooplankton populations in that river (Caraco et al., 1997).

Two Case Studies that Consider both Fishing and Habitat Effects

Winter Flounder in New England

Winter flounder (*Pseudopleuronectes americanus*), is one of the most commercially and recreationally important fish species in the northeast and provides one of the best examples of the importance of habitat to fisheries yield (ASMFC, 1992). Many populations spawn in and use estuaries as nursery habitat (Howe and Coates, 1975; Howe et al., 1976). It has been the focus of a variety of environmental impact studies because of its economic value and because it shows clear responses to poor water quality (Murchelano and Briggs, 1985; Bejda et al., 1992). Early life stages (eggs, larva and juveniles) of inshore populations are susceptible to water withdrawal (Crecco and Howell, 1990), toxic substances (Nelson et al., 1991) and physical loss or degradation of habitat (Briggs and O'Conner, 1971). Habitat degradation has been shown to increase juvenile mortality (Briggs and O'Conner, 1971) and decrease growth (Bejda et al., 1992; Saucerman, 1990). Age-1 and older fish are also subjected to high levels of recreational and fishing mortality (Boreman et al., 1993).

The Fishery Management Plan for inshore stocks of winter flounder provides an analysis of the relative effects of habitat loss versus changes in fishing mortality on fish survival (ASMFC, 1992). Based on the work described below (Boreman et al., 1993), the plan concludes that a strategy of habitat improvements that would increase juvenile survivorship would result in long term benefits to population success and provide a firmer basis for increasing yields in the future than would a strategy based solely on a reduction in fishing mortality. Boreman et al. (1993) compared the relative value of increasing age-0 survival through habitat restoration or decreasing fishing pressure on adult stocks as ways to reverse the trends of decreasing stock declines. They used the eggs-per-recruit (EPR) method which is a way of equating mortality effects on early life stages of a fish species to subsequent loss of fishing opportunity. Thus the EPR method can be used to compare changes in potential egg production due to loss of juvenile fish because of habitat loss or degradation to losses in egg production due to harvesting of adults. As such, it gives managers a means to compare potential effects of habitat change with changes in fishing mortality. Their analysis based on population characteristics of Cape Cod Bay flounder populations indicates that doubling juvenile survival through habitat restoration yields the same egg production as reducing fishing mortality by 63%. This result suggests that growth in stock abundance is limited by a carrying capacity bottleneck that occurs sometime before the fish become susceptible to fishing pressure. They also found, however, that very few adult age classes (ages 1-3) contribute to egg production because fishing mortality was high. This means that loss of a single year-class because of disruption of juvenile habitat could lead to a serious population decline. The combination of habitat limitation and severe overfishing leaves the stock vulnerable to collapse. The best strategy for stock preservation is both habitat improvements and control of fishing pressure.

These recommendations apply to inshore stocks of winter flounder. A similar analysis for offshore populations, such as those on Georges Bank, has not been done and may or may not come to similar conclusions.

The North West Shelf, Australia

Although we currently lack the data to separate fishing mortality from habitat alteration effects in New England offshore fisheries, examining a similar situation for the North West Shelf region of Australia is instructive. This region faced problems similar to those of our offshore fisheries in New England. Fish species composition was changing from desirable to undesirable species, and total abundance was declining. The North West Shelf was under intense trawling fishing pressure (Sainsbury et al., 1993) and its benthic habitat altered. The catch of epibenthic fauna (mostly sponges, alcyonians and gorgonians) had declined from 500 kg/hr to only a few kg/hr. Little was known about the relationship between the fish stocks and the habitat provided by demersal epibenthic organisms other than that there was a strong correlation between the presence of these organisms and fish populations. Four research hypotheses were developed which either together or separately could explain the changes:

H1. environmentally induced changes independent of the fishery,
H2. multiple independent responses by fish species to exploitation,
H3. alteration of biological interactions due to the fishery, or
H4. indirect effects of fishing, such as habitat alterations.

Each explanation had different management implications. If the main cause for the decline was the loss of epibenthic habitat (hypothesis 4), or trawl-induced changes in competitive/predation interactions (hypothesis 3), then there might be scope for expansion of a trap fishery to replace trawling. On the other hand, if the historical declines were because these stocks had intrinsically low productivity (hypothesis 2), then the only solution was to cut back on all fishing.

To clarify these issues, an adaptive management approach was used. Broad areas of the NWS were regulated with 2 different management regimes (open to trawling and closed to trawling) for 5 years and the fish populations and epibenthos monitored by fishery-independent trawls. Catch rates in the area closed to trawling increased along with the epibenthos, while fish catches and epibenthos abundance continued to decline in the areas open to trawling. A second area which had initially been open to trawling was closed to trawling 2 years into the study and catch rates in this area also began to recover.

These results show a good correlation between the catch rate and the abundance of epibenthic organisms, however, it was possible that both the epibenthic organisms and the fish were responding separately to the effects of trawling. To test for this, alternate resource dynamic models were developed. Combining the historical catch and effort data with the experimental results indicated that abundance of the major fish species is limited by the amount of suitable habitat (epibenthic fauna). The analysis carried out on the NWS illustrates the scope for use of the "adaptive" management approach to evaluating seemingly contradictory explanations to large and complex problems.

WHICH NORTHEAST FISH SPECIES HAVE BEEN AFFECTED BY HABITAT LOSS OR DEGRADATION?

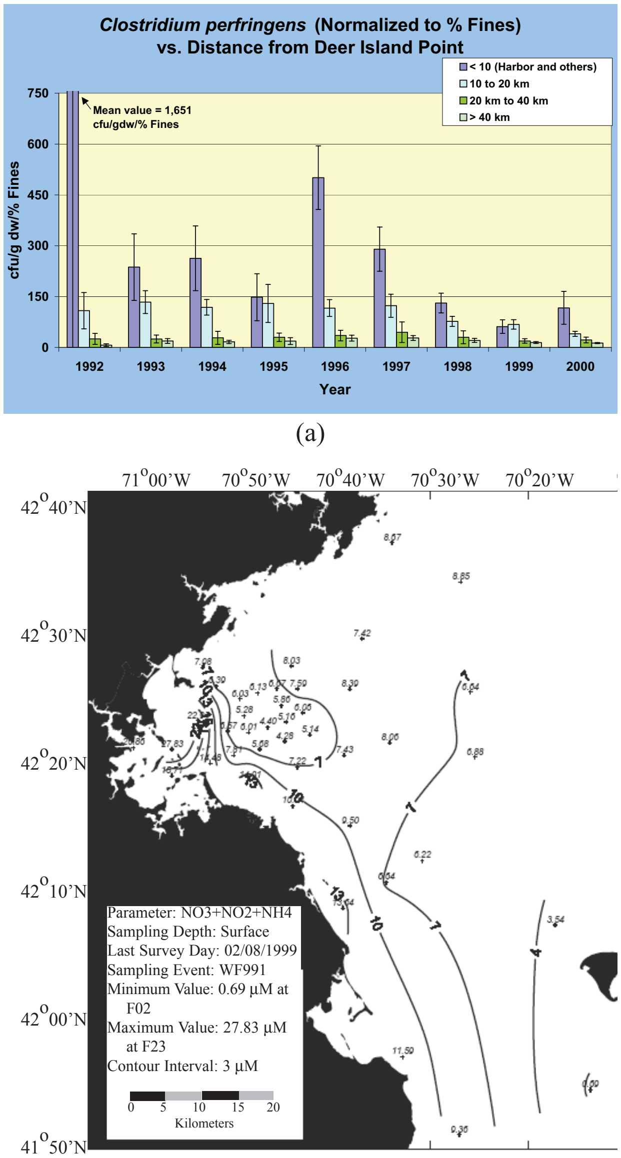
Because of the complex relationships of individual species to habitats and the myriad causes of habitat degradation it is difficult to unambiguously establish cause and effect between habitat declines and fisheries declines. The above example of nearshore populations of winter flounder provides one illustration of a particular population where habitat degradation has affected a fishery. Except for anadromous fish (see Moring, Chapter 3) and the decline in bay scallops during the eelgrass wasting disease epidemic in the 1930s (Thayer et al., 1984), there are little data allowing us to unambiguously link habitat changes with regional declines in fish populations, since most fish populations for which there are adequate data have also been heavily exploited. On an embayment level, we know, for example, that loss of eelgrass reduces the ability of the bay to support bay scallops and winter flounder. These populations are also under heavy fishing pressure that may have had an equal or greater impact on numbers. The combination of habitat loss and degradation, overfishing, and some natural environmental fluctuations may cause greater declines in a population than any of these factors alone. Some species can use alternate habitats, food sources and migration pathways, while others cannot. In addition, different populations of the same species may have different habitat requirements. For example, the Georges Bank population of winter flounder never comes into any estuary, while other populations are specific to certain estuaries (Howe et al., 1976).

With those caveats in mind, there are several predictions one might make if habitat losses and degradation were major factors in fisheries declines. Although the discussion of these predictions points to the difficulty of separating habitat from other factors as causing most fisheries declines, we think it is still instructive.

THE DECLINE SHOULD BE MOST OBVIOUS FOR THOSE SPECIES WHOSE HABITAT HAS BEEN THE MOST DEGRADED. VARIATIONS IN HABITAT QUALITY AND EXTENT SHOULD BE REFLECTED IN FLUCTUATIONS IN THE FISHERY.

Anadromous fish are the clearest example of a strong relationship between habitat degradation and population declines (Moring, Chapter 3). Dams, culverts, and cranberry bogs have all been impediments to their passage to spawning areas (Rebeck and DiCarlo, 1972) and erosion due to poor land use practices and eutrophication have degraded their spawning areas.

Inshore stocks of winter flounder also show a loss of fisheries yield because of habitat alteration, although the causes vary from estuary to estuary (ASMFC, 1992). Land derived pollutants tend to decrease in a gradient with distance from shore (Figure 5.7). In some estuaries, the loss has been attributed to eutrophication and anoxia; in others, toxic contaminants. The previously cited example of the relationship between bay scallops and eelgrass



(a)

(b)

Figure 5.7. Distribution of land derived (a) *Clostridium* (MWRA, 2003) and (b) Dissolved inorganic nitrogen (Libby et al., 2000) from Boston Harbor into Massachusetts Bay.

also provides an example of a response of a fisheries species to habitat degradation and loss. Recent declines in the bay scallop fishery on Long Island have been attributed to the loss of eelgrass as a result of smothering by the brown tide organism (Dennison, 1987). Species, such as bay scallops, that are not flexible and that have very specific requirements for spawning, feeding or migration pathways are most at risk from habitat change.

Recent surveys by Heck et al. (1989, 1995) and Deegan et al. (1997) on Cape Cod underscore the particular importance of eelgrass habitats in the Northeast to coastal fish and macroinvertebrates in terms of supporting a larger measurable diversity of species than other estuarine habitats. Nonetheless, the impact on finfish fisheries from the eelgrass decline of the 1930s and other, more recent declines has not been demonstrated. This is in part because we lack historical data both on the extent of these habitats and on abundance of fish within these habitats and because currently few fish species of commercial importance use these habitats (Heck et al., 1989; Deegan et al., 1997).

Since the present decline in commercial fish species in New England is occurring both near and offshore, nearshore habitat loss is obviously not the only explanation for the decline. The commercial fishery of New England is less dependent on species with a clear ecological dependence on coastal areas than other parts of the country (Nixon, 1980). For example, of the top five important commercial species of finfish in New England (Atlantic cod, haddock, yellowtail flounder, American plaice and winter flounder), only winter flounder uses estuaries extensively, and even within this species, not all populations use estuaries (ASMFC, 1992). As stressed earlier in this chapter, the most widespread habitat alteration in offshore fishery has been the use of mobile gear. Since all these offshore species are currently at low population levels due to intensive fishing (Murawski, Chapter 2), any effects of habitat changes from fishing gear have been strongly confounded by fishing mortality.

Broad scale environmental changes have occurred on Georges Bank and other offshore regions, but the evidence that they are the primary cause of recent fisheries declines is not compelling. Such environmental changes include recent increases in seawater temperature and the impacts of mobile fishing gear described earlier. If these factors were having a major impact on offshore fisheries populations, then one would, at the very least, expect a decline in the survivorship of juveniles relative to the size of the spawning stock. Such a decline in this ratio could also be caused by other factors, such as survivorship of eggs and 0 year fish prior to recruitment or increases in predator populations, so this is not a conclusive indication, but it should nonetheless occur. Based on the ratio of new recruits to the biomass of the spawning stock that spawned the new recruits, there is no evidence for reduced juvenile survivorship in cod, haddock, and yellowtail flounder (Murawski, Chapter 2). These species are at historic population lows in our region, and it is likely that the magnitude of overfishing in the offshore fisheries overwhelms any effect of habitat alteration in New England at this time.

BENTHIC FISH SHOULD SHOW GREATER DECLINES THAN WHOLLY PELAGIC SPECIES

Because most habitat alterations have their greatest impact either directly or indirectly on the benthos, another prediction is that benthic species are more likely to be affected by habitat degradation than pelagic species (Caddy, 1993; Deegan et al., 1997). Low dissolved oxygen in bottom waters, alteration of benthic substrata, and concentrations of pollutants all should disproportionately impact benthic species.

The recent declines in fisheries have, in fact, affected many demersal species dramatically. However some pelagic fish, Atlantic Bluefin Tuna, Atlantic Swordfish, and a number of shark species, are all overexploited and at low population levels along the east coast at present (NMFS, 2001). These are not known to have any benthic stages. Atlantic herring and mackerel, two pelagic species that are not heavily fished now, are presently relatively abundant and classified by NMFS as underexploited (NMFS, 2001). Atlantic herring have demersal eggs and may be subjected to impacts on the benthic community by mobile fishing gear (Valentine and Lough, 1991), yet they seem to be doing well.

THE TIMING OF THE DECLINE IN FISHERIES SHOULD BE RELATED TO THE TIMING OF HABITAT LOSSES AND DEGRADATION

It may be possible to test this hypothesis with nearshore fish that use coastal wetlands and estuaries because the periods of some habitat losses are well defined. As described earlier, the filling of coastal wetlands was most intense in the northeast between 1950 and 1970. One would predict that if the loss of coastal wetlands was a major factor in fish declines, then there should have been a larger decline in coastal fish populations during 1950-1970 than in recent years when wetlands protection efforts have been stepped up.

There are several reasons why relationships between wetlands loss and declines in fisheries have been difficult to document even in those regions of the country where the fisheries are comprised of species that have clear ecological links to coastal wetland habitats. First, we often do not have good documentation on the fishing effort or landing of inshore fishes most likely to have been affected by these coastal alterations during a relevant time period such as during the period of most intensive wetland filling (1950-1970). For example, we have only sporadic information on American shad, striped bass and winter flounder landings prior to 1965 (Figure 5.8). Another difficulty is that in the past most estuarine fisheries were not fully exploited, thus, any loss in the total population due to loss of habitat could be made up in the fishery by increasing effort (Houde and Rutherford, 1993; NOAA, 1995). On a national level, most (but not all) estuarine-dependent fisheries in the United States have declined sharply or collapsed, in contrast to relatively stable catches of estuarine-dependent species on a global scale (Houde and Rutherford, 1993). Houde and Rutherford (1993) attribute overfishing as the major cause of the decline of estuarine fish, with some impacts from habitat alteration and “the vaguely documented but probably real consequences of pollutants and contaminants.” In sum, the “timing hypothesis” is hard to test since there are too many confounding variables—in particular, estuarine fish stocks were overfished at the same time severe habitat alterations were occurring.

When the composition of fish communities in coastal areas are considered over long periods of time (> 20 yr) some changes become apparent. In Waquoit Bay, for example, some fish species were 10 to 60 times less abundant in 1967 compared to 1987 while some increased in abundance (Figure 5.9). The differences in the fish community, described earlier in this paper between existing moderate and low quality eelgrass habitats in Waquoit Bay, while consistent and significant, were small compared to such changes seen over a thirty year period in the entire bay (Table 5.3, Deegan et al., 1990). Species common in commercial and

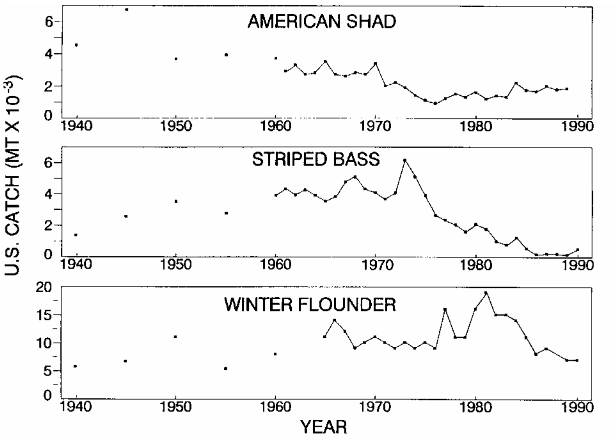


Figure 5.8. Landing statistics for three species, American shad, striped bass and winter flounder, that have a strong ecological connection to estuaries and that are also important in commercial and recreational fisheries (from Houde and Rutherford, 1993). Note that the landing information for these species prior to the mid-1960s is very sparse. *Reprinted with permission, Estuarine Research Federation.*

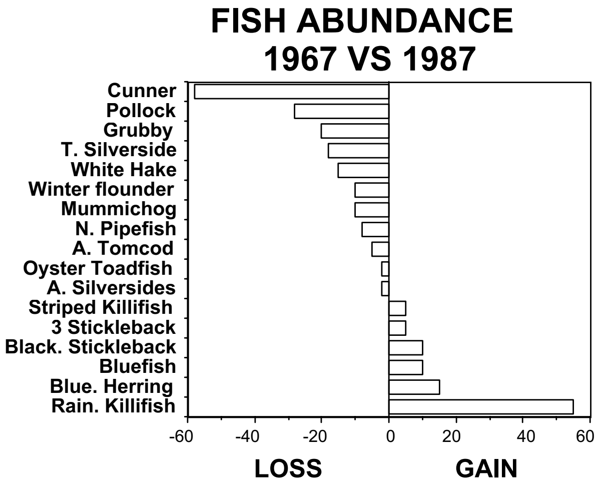


Figure 5.9. Changes in fish community composition and abundance in Waquoit Bay over a twenty year period (Deegan et al., 1990). Many important commercial and recreationally important species, such as winter flounder, declined in relative abundance, while small forage fish, such as rainwater killifish, have increased in relative abundance.

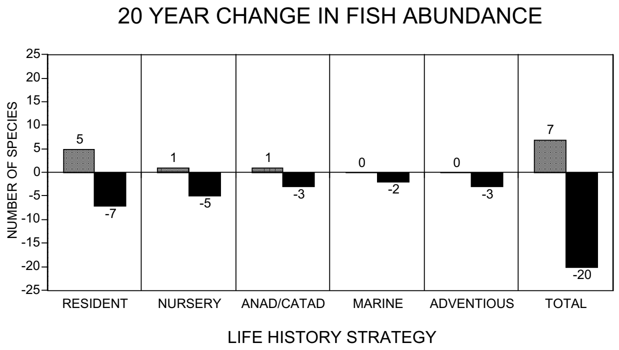


Figure 5.10. The number of species in each life history category which either declined or gained in abundance from 1967 to 1987 in Waquoit Bay, MA (Deegan, unpublished data). Negative numbers indicate the number of species that declined in abundance, while positive numbers indicate the number of species that gained in abundance.

recreational fisheries, such as winter flounder, white hake and pollock declined during this time period, while some small forage fish, such as sticklebacks and silversides, increased in abundance. The number of species that declined in abundance exceeded the number of species that increased in abundance across all life-history patterns over the twenty year period (Figure 5.10). Species that use the estuary as a nursery area were particularly affected with 84% of the species declining in abundance.

It is difficult to attribute changes over 20 years in Waquoit Bay to any single cause because many changes occurred simultaneously: land use in the watershed changed from natural to suburban, nutrient loading increased, the open bay was dredged, and its hydrology was altered by freshwater control structures and dredging. The decline of eelgrass area to less than 20% of historical levels and the reduced carrying capacity of the remaining habitat as a result of these alterations were probably important factors in the change in the fish community. In addition to these local factors, regional fishing pressure has changed the populations of predatory fish, such as bluefish, cod, striped bass, and summer flounder, that migrate into estuaries along the coast for part of their life history.

There are other reasons why there has been no umambiguous "signal" reflected in nearshore fish stocks from coastal wetland loss and degradation. It is possible that despite the past losses of habitat, enough suitable estuarine habitat still remains to sustain populations of estuarine species. In addition, certain essential components of the marsh community, such as the mummichog, *Fundulus heteroclitus*, are sufficiently flexible in their own habitat requirements and tolerant of habitat degradation that they can still provide the key links between estuaries and offshore fisheries despite human activities in coastal areas.

An unwitting "experiment" currently underway that might test these speculations is related to the tremendous loss of coastal wetlands in the Mississippi River delta due to land subsidence. In this area, containing roughly one quarter of the country's coastal wetlands, about 57,000 acres of coastal wetlands were replaced by open water from 1974-1983 along with an even greater amount of freshwater alluvial wetlands (Tiner, 1991). The land in the delta is undergoing submergence because the heavily channelized Mississippi River and its tributaries no longer provide the sediment that has historically enabled these wetlands to keep pace with rising sea level (Turner and Rao, 1990). The loss of wetland acreage has not had an immediate impact on such estuarine dependent commercial species as brown shrimp, but there is speculation that the long term consequences could be disastrous, once the submerged wetlands remnants completely break up (Boesch et al., 1994).

Because increased fishing pressure and habitat alterations often occur almost simultaneously, it will always be difficult to completely separate their effects both spatially and temporally on fisheries. This becomes even more problematic when one source of habitat alteration is the gear used in the fishery. Overfishing and habitat alteration may also act synergistically in contributing to the decline of fisheries. One likely interaction is that overfishing causes the initial collapse of the population and habitat alterations prevent the recovery of the fishery even after fishing pressure is reduced.

VARIATIONS IN LIFE AMONG DIFFERENT SPECIES HISTORIES (I.E., R- VERSUS K-SELECTED) SHOULD AFFECT THE DEGREE TO WHICH THESE SPECIES ARE IMPACTED BY HABITAT ALTERATIONS

One might look at life history characteristics of fish to see if any factors inherent in the fish make their populations more or less susceptible to the impacts of fishing or habitat alterations. Some traits that may be significant include number of young produced, age-specific survival, age at first reproduction, age-specific fecundity, longevity, and number of different habitats required over a lifespan. Some species, termed r-selected, produce an abundance of young to compensate for fluctuating and unpredictable environments. Juvenile survival is low and the fish typically mature at an early age. Populations levels of K-selected species are constrained by a relatively stable environmental carrying capacity. K-selected species produce fewer young, but higher juvenile survivorship than r-selected species and are slower to reach reproductive age.

The relative importance of juvenile versus adult mortality should be related to features of reproductive or life history strategy. Juvenile mortality is generally related to an aspect of the habitat while adult mortality is often controlled by fishing effort.

Schaaf et al. (1993) used life history information for 12 stocks of fish from the mid Atlantic region to compare the effects of destroying juvenile and adult habitat through pollution on stock size. They used estimates of age-specific mortality and fecundity in single species computer simulation models to compare the effects on stock size of an increase in mortality at the juvenile stage, such as might be caused by pollution, versus increased adult mortality. For example, they found that destroying 2% of the estuarine habitat of juvenile Atlantic menhaden could result in a 58% decline in population levels after 10 years. Destroying the same amount of oceanic adult habitat resulted in only an 8% decline. Variability in juvenile survivorship appears to be quite significant for some stocks, while other stocks were more susceptible to changes in adult mortality.

The analysis of Schaaf et al. (1993) which predicted a large decline in long-term harvest of menhaden from a small alteration in juvenile habitat also emphasizes the cumulative impact of small changes in habitat on populations. Their analysis also showed that a severe pollution event (and presumably other forms of habitat degradation) could have a devastating effect if it occurred when the fish were concentrated in their spawning areas. It would be interesting to expand this modeling to include fishing mortality and loss of juveniles to bycatch.

Most exploited fish species in the northeast are intermediate between r- and K- species and contain elements of both. Groundfish, for example are relatively uniform in their life histories, generally maturing between 2 to 3 (gadids) or 3 to 4 (flounders) years. Their reproductive and mortality terms vary from stock to stock. Compared to terrestrial vertebrates, almost all species of fish (and shellfish) in the northeast produce tremendous numbers of young and have extremely high juvenile mortality rates and large year to year variations in reproductive success. On the other hand age to first reproduction differs substantially (2-3 years in Atlantic cod, 7 years in lobsters, 8-10 years in Atlantic bluefin tuna), as does adult survivorship. We predict that populations of rapidly maturing and shorter-lived species are less likely to be influenced by anthropogenic habitat alterations than those that are more K-selected, since rapid growth and maturity should be advantageous in a changing, disturbed environment. In addition, the populations of r-selected species should recover more rapidly from habitat alterations and overfishing, at least in the short term. The analysis remains to be done.

IN THE FUTURE

It is clear from the above discussions that the links between habitat alteration and loss of fisheries production can be subtle, diverse and operate on many scales from site-specific to regional. The differences among species in the nature of the relationship between fish and their habitats underscores the difficulty of determining the cause and effect relationships between habitat loss and degradation and loss of fisheries. It should not be surprising that we can find no unambiguous correlation between a single type of habitat alteration and the loss of a fishery since many habitat alterations have occurred simultaneously. Isolating the impacts of habitat alterations is also confounded by fishing activity and other constraints on fish populations.

IMPORTANT RESEARCH QUESTIONS

- *What are the critical ecological processes and habitats that sustain fisheries?* For example, are upwellings important in the New England region? What environmental “cues” do fish use to determine migration pathways and spawning locations? How important is regional hydrology in distributing larvae from “sources” areas? In estuaries, how important are inputs from uplands compared to in situ production within the estuary? Are salt marshes and seagrasses critical nursery habitats for fish and shellfish in northern New England as they are in other parts of the country?

- *What are the spatial and temporal scales of the critical ecological processes?* Must we consider processes on the scale of a single salt marsh within an estuary, several salt marshes within a single estuary or a series of estuaries? Is it sufficient to understand year-to-year variation or are there longer-term, perhaps decadal trends in weather or river discharge, that we must account for in our management plans?

- *Are there life history bottlenecks?* Understanding the sequence of life-history stages that control populations in the absence of harvesting is critical to population management. Wahle and Steneck (1991) suggest that cobble habitat is essential for the settling of juvenile lobsters and may be fostering a demographic bottleneck. How many other species of fish have such a critical relationship with a particular habitat type that may be limiting their populations?

- *What factors influence the carrying capacity of habitats?* For example, what role do physical structure and the production of food play in determining how many individuals or species will be found in an area?

- *What landscape factors control the productivity and distribution of habitats?* Understanding why habitats are not uniformly distributed nor uniformly productive among locations would lead to a better understanding of the landscape features that must be preserved to maintain fisheries. For example, are seagrass beds adjacent to marshes more productive than those in open embayments or those occupying small coves between rocky headlands? Are offshore biogenic habitats of uniform density more productive for fish than those that are patchy and scattered?

- *What are the impacts of eutrophication, contaminants, and fishing methods on critical ecological processes and habitats (i.e. assess ecosystem and habitat integrity)?* We need to understand not only how natural ecosystems and populations function, but also how human intervention changes the way ecosystems function. Although changes in water quality as a result of anthropogenic eutrophication are well documented, we know much less about the indirect effects of eutrophication on biological communities. For example, what are the consequences to fish of the frequent disturbance of the benthic community by mobile fishing gear?

- *What are the separate effects of overfishing and habitat degradation on populations and ecosystems?* Removal of an animal population by fishing and loss of the ability of a habitat to support certain species may have very different implications for the structure and function of ecosystems and the future viability and management of fisheries. Fish that are the targets of fisheries may control important ecosystem processes by their predation on other animals, behavioral activities such as burrowing, or by providing physical structure as by-products of their life-history (see Witman and Sebens 1992). Removal of key species can cause the indirect decline of other species. Altering the habitat directly can cause a general decline in the abundance of organisms as well as cause species shifts.

IMPORTANT MANAGEMENT QUESTIONS RELATED TO THE RESEARCH QUESTIONS

- *What are the most important areas of habitat to protect?* The Magnuson-Stevens Fisheries Conservation and Management Act (MSFCMA) of 1996 requires that fisheries managers delineate, protect, and conserve essential fish habitat (EFH). Baseline ecological data must be collected on existing habitats and their associated fish communities so that managers can determine which habitats should be considered essential to fish, what threats these habitats face, and what level of protection they need. The MSFCMA provides the opportunity to designate a subset of EFH as habitat areas of particular concern (HAPC). These “HAPC’s” are especially critical to some species of fish, vulnerable to human impacts, and therefore in need of high levels of protection. This may be important if there are multiple choices for a proposed dredge and fill operation or where a specific type of fishing gear is used.

- *Will a system of marine reserves be a useful tool in sustaining fish populations? If so, what are the key characteristics of the habitat that should be protected?* A number of scientists have suggested setting up nonextractive marine reserves as a conservation tool to insure sustainable fisheries (see articles in Shackell and Willison 1995). Which species will benefit by having refugia from direct fishing mortality and from the indirect effects of fishing gear?

- *What are the habitat characteristics of greatest importance to managed fish and shellfish and to the integrity of the marine ecosystem?* How will such reserves be managed and integrated into fisheries management plans? How large do they need to be to be viable? How many different habitat types should each contain? How should each be linked to other reserves to make a viable system?

- *What are the trends in various habitats?* Is the habitat stable or is it degrading? What natural or anthropogenic factors are responsible for any changes in the habitat? What type of habitat monitoring program is needed to aid in maintaining sustainable fisheries resources?

- *What are the acceptable levels of change in habitat variables?* What are the boundaries of acceptable change? For example, how much water can be harvested for offstream use and how much can the seasonality of flow regimes be altered without major impacts on fish communities. How often can a benthic community be trawled before it begins to degrade?

- *What are the quantitative relationships between habitat change and watershed activity?* It is important to establish the linkages between watershed activities and fish habitat. Quantitative relationships will have to be established. For example, what land use practices contribute to sediment runoff and how can land use practices be modified to reduce sediment contribution to an acceptable limit?

- *What are the habitat rehabilitation options?* In certain circumstances management actions to rehabilitate degraded habitat may be both desirable and practical. Managers need data on specific rehabilitation requirements. For example, it may be possible to replant salt marshes, but managers need to know what are the hydrologic, geomorphological, and production characteristics that will make salt marshes productive fish habitats? Cost is another obvious consideration.

SUMMARY AND CONCLUSIONS

a. Over time, the problem for coastal habitats has changed from outright destruction to more subtle degradation, such as eutrophication. In offshore habitats, the recent expansion of mobile gear to habitats that were formerly immune to dragging and bottom dredging has exacerbated the impact of fishing activities on habitats and the fisheries they support. The impacts on fisheries will likely depend on the type and extent of the habitat alteration, the frequency of the disturbance compared to natural changes, the characteristics of the habitat, and the life history characteristics of the species involved.

b. Habitat degradation can drastically alter fish communities. Eutrophication and alteration of freshwater inflows are currently the two most prevalent problems in nearshore habitats. Offshore, substantial changes in the physical and biological structure of habitats as a result of the widespread use of mobile gear have been documented. How these perturbations impact fisheries and the marine food web is still under investigation.

c. Winter flounder provides one case where habitat degradation is believed to impact certain coastal populations measurably. Toxic pollution, water withdrawals and other forms of habitat degradation reduce the viability of eggs and juveniles. The extent of these impacts varies from place to place.

d. Loss and degradation of coastal habitats are probably not the major cause of recent declines in most commercial fish stocks in New England. This is because most of the currently important fisheries are based on offshore populations with no direct ecological connection to nearshore habitats. Habitat alterations in nearshore areas are probably responsible for past and ongoing declines in some nearshore fisheries.

e. The most critical habitat question relating to commercial fish in New England at the moment is how habitat alterations in offshore regions are affecting fish currently at low population numbers after years of overfishing. The impacts of widespread and intense disturbance of offshore benthic communities by mobile gear on fish survivorship, particularly juvenile groundfish and demersal eggs of pelagic species, are potentially significant. Its affect on fish population numbers has not been quantified yet. Such alterations on offshore habitats have not been as significant as overfishing in explaining the current fisheries decline, however, existing data on the impacts of mobile gear suggest that the rate of recovery will be impeded without habitat conservation and management.

f. A number of modeling studies suggest that even within the context of overfishing, habitat degradation can still have an impact on fish populations.

ACKNOWLEDGMENTS

We thank our colleagues for stimulating and challenging conversations on the relative importance of human versus natural factors controlling populations. Reviews by Peter Auster, Mark Chandler, and William Robinson contributed greatly to this manuscript. This work was supported in part by the Jessie B. Cox Charitable Trust, The National Science Foundation (NSF-OCE 9726921, NSF-DEB 9726862), the Mellon Foundation, the Environmental Protection Agency (R825757-01-1), and the Massachusetts Bays Program.

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Chapter VI

Effects of Natural Mortality and Harvesting on Inshore Bivalve Population Trends

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INTRODUCTION

> *The future of the quahaug industry of Massachusetts lies in the hands of her citizens, since only through public sentiment can suitable laws be obtained for its preservation.*
>
> *-David L. Belding, 1912*
> *(in Belding, 1930b)*

The inshore bivalve fishery of New England is focused on three commercial species: the soft-shell clam (= soft clam; *Mya arenaria*), the quahog (= quahaug, hard-shell clam, or hard clam; *Mercenaria mercenaria*) and the bay scallop (*Argopecten irradians*). The soft-shell clam inhabits intertidal mudflats throughout the region but the primary fishery for this species is centered along the coast of Maine, the North Shore of Massachusetts and in Boston Harbor. The quahog, which inhabits intertidal and shallow subtidal flats, and the bay scallop, which is associated with eelgrass beds (*Zostera marina*), are predominantly distributed in southern New England (South Shore of Massachusetts including Cape Cod and the Islands, Rhode Island and Connecticut).

Since all three species are harvested in Massachusetts, and since Massachusetts fisheries data, albeit limited, are available on each of them, this state will be highlighted throughout this chapter. Historically, Massachusetts has been a major shellfish producer. During the past fifteen years, (1982-1998) Massachusetts has been the leader nationwide in the production of bay scallops and in most years is second only to Maine in the production of soft-shell clams. Combined, the annual landings of quahogs, soft-shell clams and bay scallops in Massachusetts have ranged between 3-5 million pounds between 1982 and 1993, valued at between eleven and twenty-one million dollars, ex-vessel price (NOAA, 1999). Commercial landings and average value of the quahog, soft-shell clam and bay scallop fisheries in the U.S. from 1982-1998 are shown in Figure 6.1. Since ex-vessel prices do not reflect costs associated with managing the fishery, such as enforcement, operation of depuration facilities, etc., these values may overestimate the realized economic value of the resources.

Although no complete map of productive shellfish beds exists, the Massachusetts Geographic Information System (MASS GIS) program in collaboration with the Department of Marine Fisheries (DMF) is currently mapping locations of potentially productive beds and their public health classifications along the Massachusetts coastline. The mapping of shellfish management areas and sampling stations has been finished (data available from T. Hoopes, DMF). The completed project will provide needed baseline information against which future assessments of shellfish growing habitat can be compared. Such information will also be useful to managers interested in selecting seeding sites for juvenile shellfish (Parker et al., 1998).

There is a growing concern among scientists and managers that the shellfish resources in many

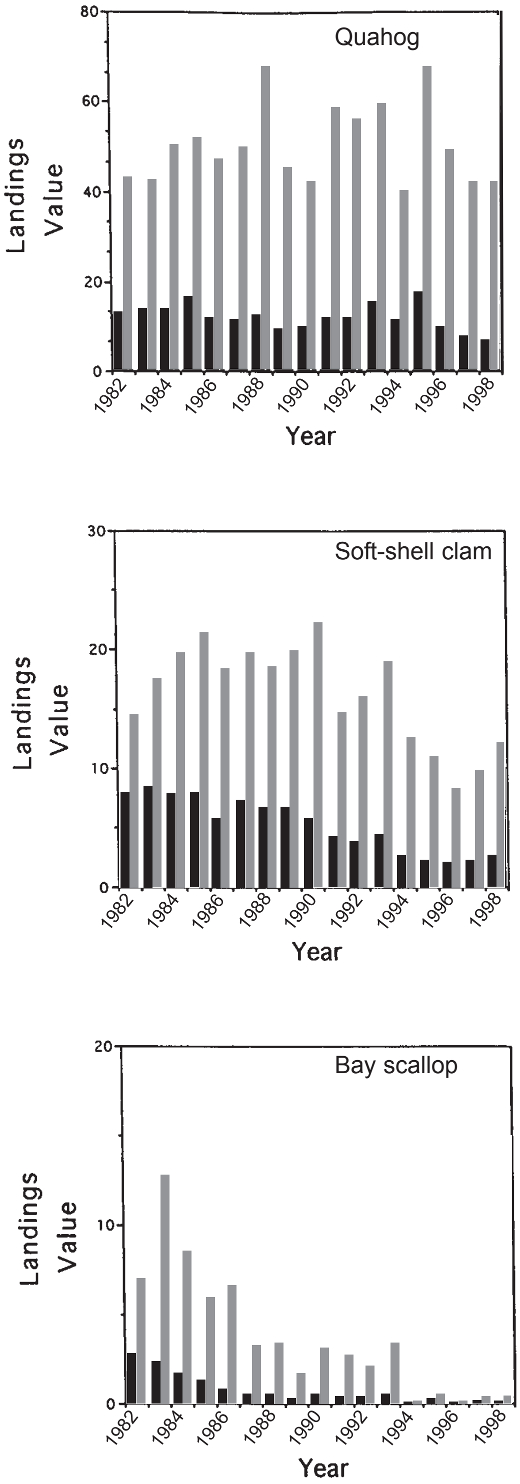

Figure 6.1. U.S. commercial quahog, soft-shell clam and bay scallop landings in millions of pounds of shucked meats and landed value (millions of dollars), 1982-1998, as reported by the U. S. Dept. of Commerce, NOAA, Fishery Statistics of the United States, http://www.st.nmfs.gov/pls/webpls/MF_ANNUAL_LANDINGS.RESULTS. (black bar = landings; grey bar = value).

areas of New England are declining (Rice, 1996). In fact, as early as 1905, reports were issued indicated that a decline in shellfish resources was already underway (Kellog, 1905; Belding, 1930a). This decline is reflected in the U.S. commercial landing statistics (Figure 6.1), especially for soft-shell clam and bay scallop resources, the bulk of which are harvested in New England. Annual catch statistics compiled by Matthiessen (1992) suggest that the landings of quahogs in Massachusetts have steadily declined from the 1950s to the 1990s (Figure 6.2a). Statewide landings of soft-shell clam have remained fairly stable (Figure 6.2b), but landings from Buzzards Bay (Alber, 1987) during the period 1955-1985 have steadily decreased. Bay scallop landings from the 1950s to the 1990s show considerable variability, with years of high production followed by several years of decline (Figure 6.2c), making it difficult to detect a definite trend. A downward trend in these stocks may simply be masked by natural variability.

There are a number of possible reasons for declines in landings of bivalve shellfish along the New England coast. One may simply be reduced fishing effort - fewer shellfishermen working coastal areas. This possibiity seems unlikely, however, since on Cape Cod alone, the number of recreational clam permits issued each year has roughly doubled between 1970 and 1990 (Matthiessen, 1992). More likely, the reduced landings are the result of reduced availability of the resource due to increased contamination, habitat degradation or loss, and 'overfishing'. The degree to which any or all of these factors contribute to shellfish decline, however, remains to be assessed. The potential role of contamination are addressed by McDowell (Chapter 7), whereas habitat issues are discussed by Deegan and Buchsbaum (Chapter 5). It is the purpose of this chapter to focus on the role of natural and fishing mortality in the apparent decline of inshore shellfisheries resources in New England, using Massachusetts as the primary example.

LIFE HISTORY INFORMATION - FACTORS AFFECTING NATURAL MORTALITY

The effective management of any fishery depends on availability of reliable biological information for the species in question. First, an

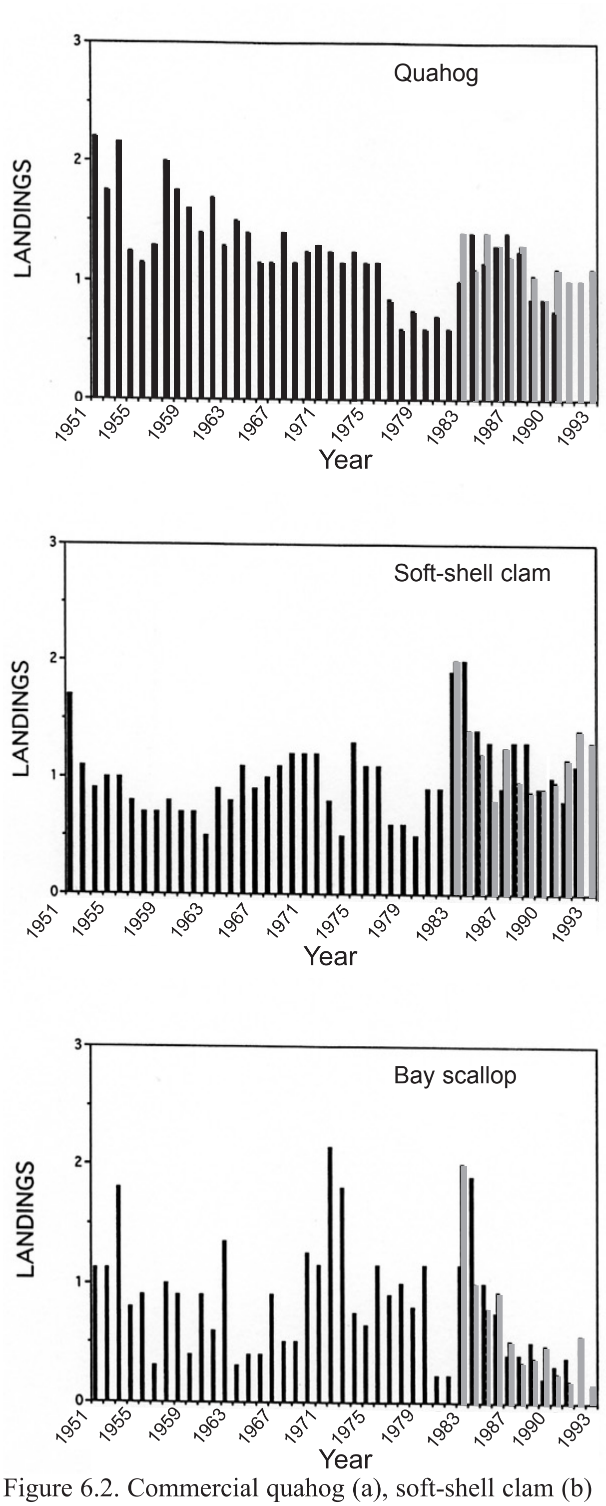

Figure 6.2. Commercial quahog (a), soft-shell clam (b) and bay scallop (c) landings (millions of pounds of shucked meats) in Massachusetts, 1951-1993. Solid bar = landings as modified from Matthiessen, 1992; gray bar = landings as reported by NMFS, Fisheries Statistics Division (http://www.st.nmfs.gov/ow-commercial/gc_runc_cgi.sh?SELECTION_STATE=Massachusetts&qyear1=1982&qyear2=1999).

understanding of the causes of natural mortality in populations is necessary in order to assess the degree to which overall mortality is due to harvesting. Secondly, knowledge of life history information is essential if harvesting strategies are to be developed and fishery impacts are to be assessed through the use of mathematical models.

Among the first accounts of the natural history of the commercially-important inshore bivalves of Massachusetts are the reports of Belding (1930a,b,c) published by the Massachusetts Division of Fisheries and Game in the early 1900s and reprinted in 1930 [and again in 2004]. These reports give detailed information on soft-shell clam, quahog and bay scallop life histories and fisheries. Since the publication of those early reports, a considerable body of literature has developed, much of which is of importance in assessing and managing these three species. In some areas, however, critical information is still lacking.

SOFT-SHELL CLAM (*Mya arenaria*)

Mya arenaria reaches sexual maturity in its second year of life (Coe and Turner, 1938; Porter, 1974; Brousseau, 1978, 1987). Gamete production rate varies from year to year and from population to population for reasons yet to be determined (Brousseau and Baglivo, 1988). The time and frequency of spawning varies widely in geographically separated populations (Table 6.1). The traditional view of fixed patterns of spawning based on latitudinal range is inadequate; habitat-specific exogenous factors such as local water temperature and food supply must be considered as well.

Settlement of recently-metamorphosed larvae from the plankton, approximately two weeks after fertilization, is the major source of recruitment into the population (since post-larval transport of spat is probably limited). Large fluctuations in yearly recruitment are characteristic of marine organisms with planktotrophic larvae. Larval recruitment, when it occurs, may represent a large proportion of the population, and that year-class may dominate the population for many years to come ("year-class phenomenon"). Recruitment fluctuations from year to year are largely the result of differential mortalities which can occur during three critical phases: 1) fertilization, 2) the free-swimming planktonic stage

Table 6.1. Duration of the spawning season of *Mya arenaria* along the Atlantic coast reported in the literature. (Modified from Brousseau, 1987).

STUDY SITE	MONTH		REFERENCE
	J F M A M J J A S O N D		
Malpeque Bay, Canada	-------------		Stafford, 1912
	----------------		Sullivan, 1948
St. Andrews, Canada	-------		Stafford, 1912
	--------		Battle, 1932
Eastern Maine	-----------		Ropes and Stickney, 1965
Boothbay Harbor, ME	---------		Ropes and Stickney, 1965
Robinhood Cove, MA	---------------	1951	Welch, 1953
	------------------	1952	Welch, 1953
Gloucester, MA	------- -----------	1973	Brousseau, 1978
	------- ----------	1974	Brousseau, 1978
	------- ----------	1975	Brousseau, 1978
Plum Island Sound, MA	-----------		Ropes and Stickney, 1965
N. of Cape Cod	---------		Belding, 1907
N. of Boston	--------		Belding, 1930a
Ipswich, MA	-------		Stevenson, 1907
Plymouth, MA	----------------		Stevenson, 1907
Southern Cape Cod	---------		Belding, 1907
	-------		Belding, 1907
Chatham, MA	------------		Stevenson, 1907
Martha's Vineyard	-------------------------		Deevey, 1948
Woods Hole, MA	---		Bumpus, 1898
Rhode Island	------		Mead and Barnes, 1904
Wickford, RI	-------- --- --	1950	Landers, 1954
	------- -----------------	1951	Landers, 1954
	----- ------	1952	Landers, 1954
Stonington, CT	---------- ------	1983	Brousseau, 1987
	---------- -----	1984	Brousseau, 1987
	----------- -----	1985	Brousseau, 1987
New Haven, CT	------------		Coe and Turner, 1938
Westport, CT	----------	1984	Brousseau, 1987
	----------	1985	Brousseau, 1987
New Jersey	-----		Belding, 1930a
New Jersey	----		Nelson and Perkins, 1931
Chesapeake Bay	---------		Rogers, 1959
Chesapeake Bay	--- ----	1956	Pfitzenmeyer, 1962
	-- -------	1957	Pfitzenmeyer, 1962
	----- -----	1958	Pfitzenmeyer, 1962
	---- ------	1959	Pfitzenmeyer, 1962

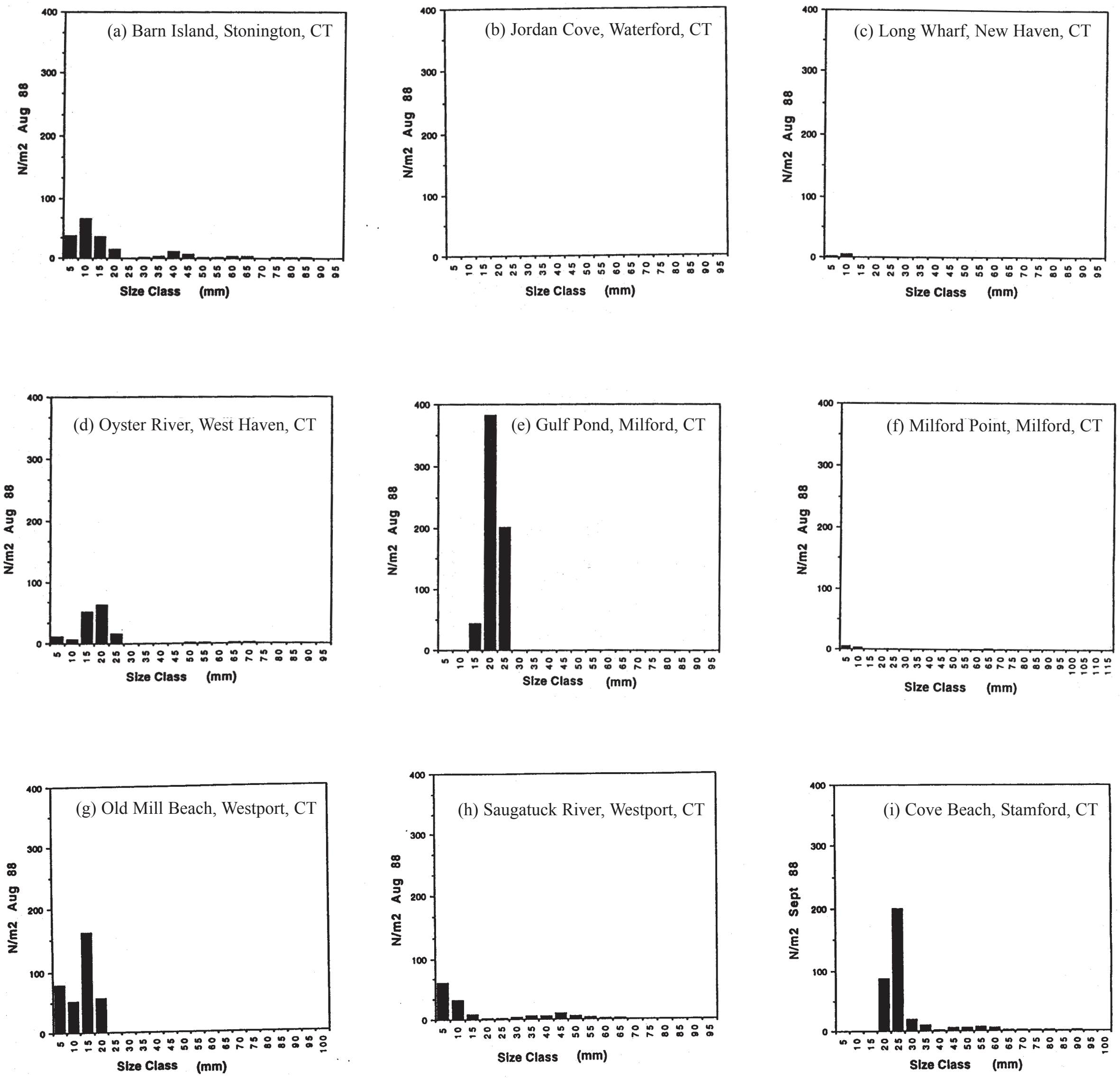

Figure 6.3. Size frequency distributions of *Mya arenaria* (soft-shell clam) during the late summer, 1988 for nine populations in Long Island Sound, (a) Barn Island, Stonington, CT; (b) Jordan Cove, Waterford, CT; (c) Long Wharf, New Haven, CT; (d) Oyster River, West Haven, CT; (e) Gulf Pond, Milford, CT; (f) Milford Point, Milford, CT; (g) Old Mill Beach, Westport, CT; (h) Saugatuck River, Westport, CT; and (i) Cove Beach, Stamford, CT.

and 3) the early post-settlement larval attachment phase.

The extent of this early life stage mortality can range from 40 to 100% (Muus, 1973; Gledhill, 1980; Brousseau et al., 1982; Brousseau and Baglivo, 1988; MacKenzie, 1994). As with other newly settled invertebrates (Hunt and Scheibling, 1997), catastrophic post-settlement mortalities of newly-recruited *M. arenaria* are not unusual (Brousseau, unpubl.) and appear to be characteristic of some environments. High mortality of spat may result from a number of abiotic factors including anoxic conditions, unfavorable temperatures, low salinities and the effects of contamination (see McDowell, Chapter 7). Biotic factors such as interspecific competition (Bradley and Cooke, 1959; Sanders et al., 1962; Moller and Rosenberg, 1983; Andre and Rosenberg, 1991), predation (Kelso, 1979; Wiltse, 1980; Ambrose, 1984; Smith, 1952; Guenther, 1992) and biological disturbance (Dunn et al., 1999) may also contribute significantly to post-settlement mortalities in *M. arenaria*. The degree to which these various causes of mortality are responsible for recruitment fluctuations is still largely unknown.

In addition to temporal variations, newly-settled *M. arenaria* also show marked spatial variation in recruitment, both within and between populations or population subunits (Snelgrove et al., 1999; Vassiliev et al., 1999). Patterns of recruitment in nine populations/subpopulations of *M. arenaria* from Long Island Sound during the 1988 late summer spawning period are shown in Figure 6.3. Spatial variation in spatfall may occur in a wide range of circumstances and depend on such factors as hydrodynamics (Emerson and Grant, 1991), suitability of settling substrate and differential survival of post-settlement juveniles.

Natural mortality rates, although high for larvae and spat, tend to decrease at close to an exponential rate with increasing size and age in *M. arenaria*. (Brousseau, 1978; Brousseau and Baglivo, 1988). Survivorship schedules follow the type III survivorship curve of Deevey (1947) — extremely heavy mortality early in life followed by low, roughly constant, mortality rates thereafter. Quantitative inter-populational differences in age-specific survival rates are measurable (Brousseau and Baglivo, 1988), suggesting that within the framework of a general life history strategy, a response to the biotic and abiotic components of the immediate environment is possible.

The environmental factors with the greatest effect on the survival of estuarine bivalves, especially the surface-dwelling juveniles, are temperature, salinity, dissolved oxygen, substrate, water movement, sediment transport and food availability. Adult *M. arenaria*, however, typically inhabit the intertidal zone and are adapted to a wide range of fluctuations in water temperature and salinity (Belding, 1930a; Chanley, 1957; Pfitzenmeyer and Drobeck, 1963; Castagna and Chanley, 1973; Shaw and Hammons, 1974). In addition, sediment depth tends to buffer temperature and salinity variations (Sanders et al., 1965; Johnson, 1965, 1967), probably minimizing the effects of these factors on survival of adult clams. The physical disturbance of clams brought about by activities such as harvesting is also a possible factor causing mortalities. Studies have shown that commercial baitworm digging negatively affects the survival of *M. arenaria* by directly damaging shells and by exposing clams to increased risk of predation (Ambrose et al., 1998).

Biotic factors such as competition, predation, disease and parasitism are probably more significant contributors to natural mortality than abiotic ones, but again the severity of the effect may vary depending on the size (age) of the individual. The failure of the soft-shell clam fishery of New England during the early 1950s was attributed to predation from the green crab (Glude, 1954), but this effect was most likely operating on juvenile clams. *Mya arenaria* reach a refuge from predation once they attain a certain size or have the ability to burrow to depths beyond the range of the predator (Edwards and Huebner, 1977; Commito, 1983; Smith et al., 1999). Nonetheless, field experiments have demonstrated that one deep-burrowing nemertean, *Cerebratulus lacteus*, is an important predator of adult soft-shell clams (Rowell and Woo, 1990). Competition for food or space is most likely a substantial cause of mortality only in the small, surface-dwelling clams. On the other hand, parasites have been reported in both juvenile and adult *M. arenaria* (Uzmann, 1952; McLaughlin and Faisel, 1997) and two types of neoplasms have been identified. It has been shown that gonadal neoplasms inhibit normal oogenesis and spawning (Barber, 1996), whereas hematopoietic neoplasia, a proliferative disorder characterized by increased numbers of "leukemia-like" cells in tissues and organs (Farley, 1969) has been shown to be a major source of mortality in field populations (Brousseau and Baglivo, 1991; Weinberg et al., 1997).

The lifespan of *M. arenaria* has been estimated at 10-12 years in Massachusetts populations (Belding, 1930a) and the oldest individual found in a study of age/growth in Long Island Sound was 11 years of age (Brousseau and Baglivo, 1987). An inverse relationship between growth rate and age has been described for *M. arenaria* and a general trend of increased growth rates with decreasing latitude exists (Table 6.2), however, geographical considerations alone are poor predictors of growth patterns (Belding, 1930a; Newcombe, 1935; Swan, 1952; Smith et al., 1955; Newell, 1982; Brousseau and Baglivo, 1987) or mean life expectancies.

QUAHOG (*Mercenaria mercenaria*)

The youngest age at sexual maturity reported

Table 6.2. The time needed for *Mya arenaria* to reach harvestable size (51 mm) as reported in the literature (Adapted from Brousseau and Baglivo, 1987).

Site	Latitude	Age at 51 mm (yrs)	Reference
Prince William Sound, AL	60°34'N	6-7	Feder and Paul, 1974
Roskilde Fjord, Denmark	55°34'N	6-7	Munch-Petersen, 1973
Lynher River, England	50°23'N	3-4	Warwick and Price, 1975
Economy Pt., Nova Scotia (8 ft. above chart datum)	45°20'N	5-6	Newcombe, 1935
St. Andrews, New Brunswick (8 ft. above chart datum)	45°10'N	5	Newcombe, 1935
Clam Cove, New Brunswick (16 ft. above chart datum)	44°45'N	7	Newcombe, 1935
Clam Cove, New Brunswick (8 ft. above chart datum)	44°45'N	5-6	Newcombe, 1935
Sissiboo River, Nova Scotia (8 ft. above chart datum)	44°30'N	5-6	Newcombe, 1935
Bedroom Cove (Georgetown Is.), ME	43°35'N	5-6	Spear and Glude, 1957
Sagadahoc Bay (Georgetown Is.), ME	43°35'N	3-4	Spear and Glude, 1957
Rowley, MA	42°26'N	2-3	Belding, 1930a
Quincy, MA	42°09'N	2-3	Turner, 1949
Gloucester, MA	41°39'N	2-3	Brousseau, 1979
Monomoy Pt., MA	41°30'N	2	Belding, 1930a
West Falmouth, MA	41°30'N	2	Kellogg, 1905
Narragansett Bay, RI	41°24'N	1-2	Mead and Barnes, 1903
Stonington, CT	41°20'N	1.5	Brousseau, 1987
Old Mill Bch., Westport, CT	41°07'N	1.5	Brousseau, 1987
Saugatuck R., Westport, CT	41°06'N	3	Brousseau, 1987

for *M. mercenaria* is one year of age (Loosanoff, 1937a; Eversole et al., 1980; Bricelj and Malouf, 1980). The spawning time of quahog populations varies with latitude, and the length of the spawning period increases with decreasing latitude (Table 6.3). Local conditions play an important role in the reproduction of this species. Appropriate temperature and food supply is necessary to condition quahogs to spawn in the laboratory (Loosanoff and Davis, 1950, 1963; Castagna and Kraeuter, 1981). Gametogenesis coincides well with phytoplankton abundance (Loosanoff, 1937b; Ansell and Loosmore, 1963) and Kassner and Malouf (1982) have suggested that food availability influences the timing of spawning. Whether or not food or certain chemical constitutents within their food act as a stimulus to trigger spawning in natural populations remains to be determined.

The major source of *M. mercenaria* recruits into the population is from the settlement of planktotrophic larvae, and like the soft-shell clam, recruitment of the quahog is sporadic. Unlike the soft-shell clam (Belding, 1930b; Moller and Rosenberg, 1983), however, there are no reports in the literature of the settlement of extremely large concentrations of spat. Quahogs are seldom found in high enough densities to allow commercial seed harvesting (Kraeuter and Castagna, 1989). The larvae lead a precarious existence at the mercy of both natural enemies and adverse physical conditions.

Work to date suggests that predation by organisms such as crabs, carnivorous snails, demersal fish and birds is the dominant factor controlling quahog abundance in naturally-occurring sets (Hibbert, 1977; Kraeuter and Castagna, 1985; MacKenzie, 1977; Virnstein, 1977; Bricelj, 1993; Micheli, 1997), but the role of established infauna in limiting *M. mercenaria* recruitment remains unclear. In spite of a growing number of studies which have demonstrated that established benthos

Table 6.3. Spawning period for populations of *Mercenaria mercenaria* along the east coast of North America based on evidence of gamete maturity and release. (Modified from Eversole, 1989).

STUDY SITE	TEMP (°C)	MONTH J F M A M J J A S O N D	REFERENCE
Wellfleet, MA	24	---------	Belding, 1930b
Milford, CT	23-25	--------------	Loosanoff, 1937b
Long Island, NY	20	------------	Kassner and Malouf, 1982
Long Island, NY	20	--------------------	
Delaware Bay, DE	25-27	------------------	Keck et al., 1975
Core Sound, NC	27-30	----------------------------	Porter, 1964
N. Santee Bay, SC	20	--------------------------------	Manzi et al., 1985
Clark Sound, SC	20-23	--------------------------------	Eversole et al., 1980
Wassaw Sound, GA	22-26	------------------------------------	Pline, 1984
Alligator Hbr., FL	16-20	--	Dalton and Menzel, 1983
Indian R., FL	<30	--	Hesselman et al., 1989

can adversely affect the early recruitment of benthic animals (Williams, 1980; Luckenbach, 1984; Andre and Rosenberg, 1991), a study by Ahn et al. (1993) has shown that dense *Gemma gemma* populations do not reduce the survival of newly-settled quahogs in a sandy substrate even when food is limited. Similarly, laboratory studies by Zobrist and Coull (1994) have shown that growth and survivorship of juvenile clams is not significantly reduced by the presence of meiofauna. Rice et al. (1989) have shown that intensive shellfishing enhances settlement and/or survival of juvenile quahogs, but whether this is due to removal of competing adults or to the disturbance of the sediment itself is not known. There is surprisingly little in the way of empirical data available, however, to assess the magnitude of the impact of natural sources of mortality on larval/juvenile survival. The only published account is a study of post-settlement survival in a New Jersey population in which a natural mortality rate of 75% was reported during the first six months of life (Connel et al., 1981).

Life expectancy of adult quahog is markedly higher than that of juveniles (Hibbert, 1977; Connel et al., 1981), and survivorship probably also follows the type III survivorship curve of Deevey (1947). The adult quahog has few natural enemies, few parasites and few pathogens that cause catastrophic mortalities. The occurrence of gonadal neoplasia in *M. mercenaria* has been documented but is rare (Bert et al., 1993), and, in fact, the quahog has been reported to possess an antitumor substance called “mercenene” which may protect the species from cancer (Schmeer, 1964). The recent appearance of QPX, a protistan disease reported to occur in quahogs from Prince Edward Island (Whyte et al., 1994) and Massachusetts (Smolowitz et al., 1998), may represent a significant threat to survival in juvenile and young adult clams.

Mortality within natural populations of *M. mercenaria* has been attributed to low salinity by Haven et al. (1975). A minimum salinity tolerance of 10 to 13 PSU was suggested by Castagna and Chanley (1973) and salinity tolerance tests indicate that salinities below 10 PSU would likely result in death during a 10-day exposure period (Winn and Knott, 1992). On the other hand, quahogs appear to be quite tolerant of low temperatures and low levels of dissolved oxygen (Winn and Knott, 1992).

Mercenaria mercenaria is one of the longest lived inshore bivalves of New England. Belding (1930b) estimated that quahogs live at least 20 to 25 years but Jones et al. (1989) reported two specimens from Narragansett Bay that were 40 years of age upon capture. This long lifespan is probably due in part to the clam's hard shell and its ability to close up completely for extended periods of time, excluding all but the most persistent of predators.

BAY SCALLOP (*Argopecten irradians*)

Unlike the two other bivalves discussed above, *Argopecten irradians* is a hermaphroditic bivalve (i.e. possessing both a testes and an ovary when sexually mature). However, only one type of sex product is usually given off at any one time (Belding, 1930c). It is hypothesized that this non-simultaneous release of gametes helps prevent self-fertilization by individuals within the population. Self fertilization, however, could play a role in the persistence of populations at very low densities.

Most scallops only spawn once, during their first year of a two-year lifespan. Such a short life span is unusual among marine bivalves. A life expectancy of 20-30 months has been reported for bay scallops from Massachusetts (Belding, 1930c). The maximum life expectancy of Long Island bay scallops is 22-23 months (Bricelj et al., 1987). In North Carolina, most scallops live only 14 to 18 months (Gutsell, 1930), while in Florida, they live 12 to 18 months (Barber and Blake, 1983). Adult bay scallops experience a period of mass mortality during their second winter and before the start of the second spawning cycle. Belding (1930c) estimated that under natural conditions only 20% of the *A. irradians* reach the two-year mark. The cause of the mortality has been attributed to senescence (Belding, 1930c; Bricelj et al., 1987), but the adult bay scallop also has natural enemies. Sea stars and the oyster drill (*Urosalpinx cinerea*) both prey on adult bay scallops, but their damage is believed to be minimal.

Gamete maturation in *A. irradians* is dependent upon food supply and a certain minimum temperature (Sastry, 1968), but spawning is not restricted to a particular period in the year or to a critical temperature (Sastry, 1963). As with soft-shell clams and quahogs, there is considerable geographic differences in spawning season, with spawning occurring later in the year in more southerly populations (Belding, 1930c; Sastry, 1966; Barber and Blake, 1983; Bricelj et al., 1987; Peterson et al., 1989; Tammi et al., 1997; Tettelback et al., 1999). Bay scallops in Massachusetts commence spawning with increasing temperatures (Belding, 1930c) while those further south spawn with decreasing fall temperatures (Gutsell, 1930; Sastry, 1963).

Bay scallop recruitment clearly shows a high degree of variability from year to year (Peterson and Summerson, 1992) which may in large measure be due to variable larval mortality. The initial free-swimming stage is followed by settlement onto elevated surfaces, primarily eelgrass blades (*Zostera marina*), to which they attach by means of byssal threads. Once settlement occurs, bay scallops are vulnerable to predation due to their thin shells, epifaunal habit and inability to maintain prolonged valve closure. In spite of the fact that eelgrass has been shown to be an effective spatial refuge from some crustacean predators (Pohle et al., 1991), high predatory risk still exists for unattached scallops prior to attainment of a partial size refuge (*ca*. 40 mm) from most predators. Periodic losses of eelgrass, such as that due to a “wasting disease” in the 1930s, have been disastrous for the bay scallop industry (Thayer et al., 1984). The occurrence of unusual algal blooms (*Aureococcus anophagefferens*) has been linked to recruitment failure of bay scallops in Long Island waters (Siddall and Nelson, 1986; Cosper et al., 1987; Tettelbach and Wenczel, 1993). The larvae either starved to death (Gallagher et al., 1989) or encountered suboptimal temperatures for survival due to delayed spawning of the adults brought about by the presence of the algae (Tettelbach and Rhodes, 1981). An outbreak of the red tide dinoflagellate, *Ptychodiscus brevis*, has also been linked to the recruitment failure of the bay scallop in North Carolina waters (Summerson and Peterson, 1990).

STOCK ASSESSMENT

A large part of the difficulty in assessing the role of overfishing on inshore bivalve stocks is the lack of dependable stock assessment data. In Massachusetts, a statewide survey of marine resources, including shellfish, was conducted about 30 years ago by the Division of Marine Fisheries and published between 1965 and 1973 as a monograph series (Jerome et al., 1965, 1966, 1967, 1968, 1969; Fiske et al., 1966, 1967,1968; Curley et al., 1970, 1972, 1974, 1975; Chesmore et al., 1971, 1972, 1973; Iwanowicz et al., 1973, 1974). No follow-up survey was ever done, however, so those studies are not useful in assessing trends.

The commercial landings statistics cited in the Introduction (Figure 6.1) are simply the annual compilation of the landings statistics reported to the U. S. Department of Commerce by the states. They are of limited use in assessing trends in abundance since they are biased by the level of fishing effort and, in the case of sedentary bivalves, the acreage of shellfish beds open to harvest, both of which can vary from year to year. Measures of landings per unit effort (LPUE) are more instructive than landings statistics alone for assessing abundance, and to some degree fishing pressure, since decreases in LPUE with increased fishing effort suggest a population in decline from overfishing (Gulland, 1974).

In order to calculate LPUE, annual estimates of landings as well as a measure of fishing effort are needed. In Massachusetts, landings records (both reports from individual shellfishermen, and constable reports) and licensing information from each town are compiled by the State's Division of Marine Fisheries. These data are currently the only means available to monitor annual changes in shellfish abundance. Reports of yearly catch by individual fishermen, however, may be underestimated. A comparison of the shellfish landings reported by shellfishermen and those reported by constables for seven Massachusetts towns selected at random for

Table 6.4. Comparison of the shellfish landings (quahog and soft-shell clam) as reported for the years 1990 - 1992 to the Massachusetts Division of Marine Fisheries by shellfishermen and constables. Landings statistics are reported as number of bushels landed.

	Landings	
	Shellfishermen	Constables
Soft-shell clams		
Town: Gloucester		
1990	203	2,000
1991	953	3,000
1992	1,231	4,000
Town: Rowley		
1990	285	---
1991	281	---
1992	462	3,800
Town: Newbury		
1990	2,857	6,000
1991	3,584	6,772
1992	5,180	7,879
Town: Essex		
1990	1,352	5,000
1991	1,742	5,000
1992	1,603	---
Quahogs		
Town: Dartmouth		
1990	7,610	13,564
1991	6,393	18,951
1992	2,461	21,884
Town: New Bedford		
1990	1,822	2,235
1991	85	940
1992	225	465
Town: Fairhaven		
1990	153	16,400
1991	440	8,100
1992	234	44,200

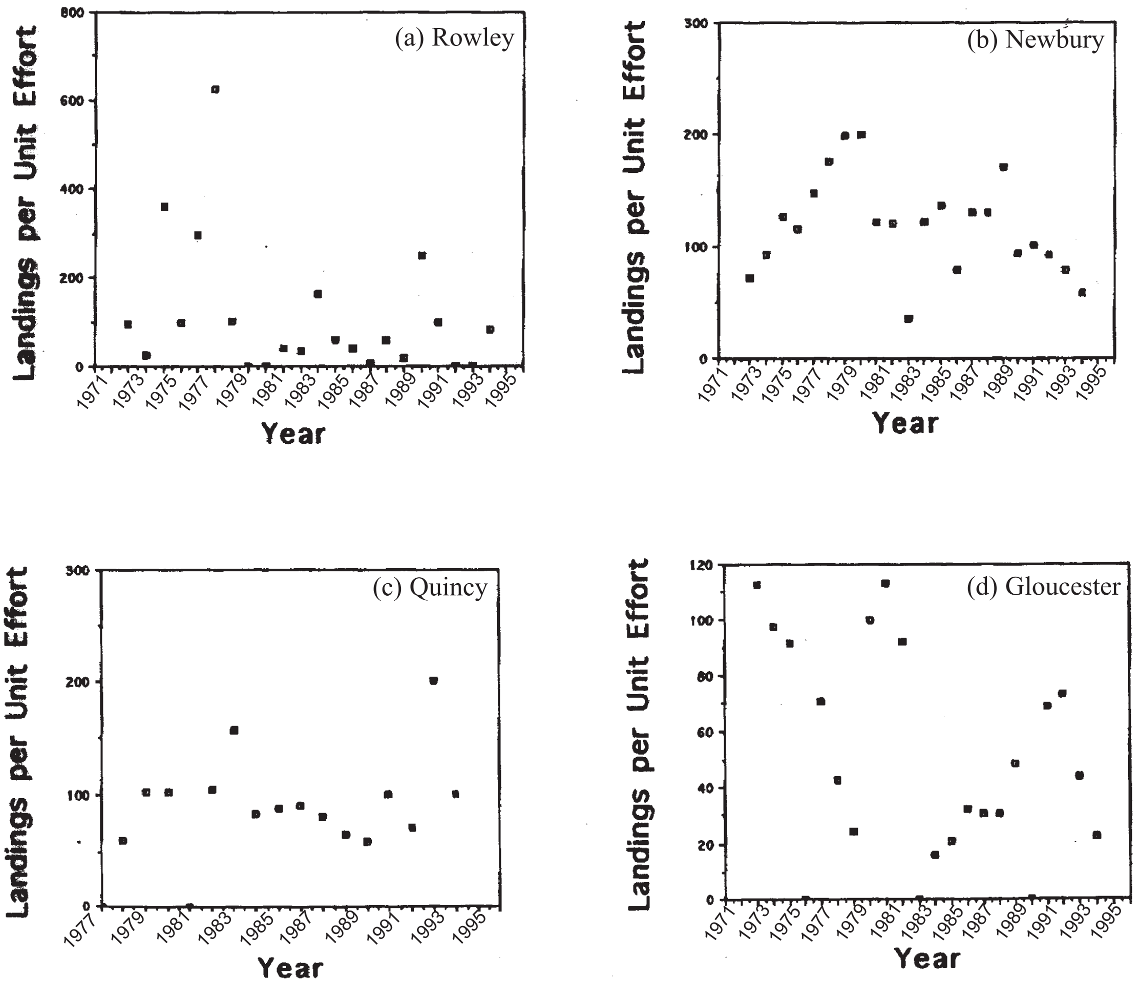

Figure 6.4. Landings per unit effort estimates calculated from soft-shell clam landings for four municipalities. a) Rowley, MA, 1972-1993. No data available for the years 1979, 1980, 1986, 1991 and 1992; b) Newbury, MA, 1972-1993; c) Quincy, MA, 1978-1993. No data available for 1981; d) Gloucester, MA, 1972-1993. No data available for 1975, 1982, and 1989.

the years 1990 - 1992 supports this contention. In each town, the size of the catch reported by the fishermen was consistently lower than that of the warden, in most cases by at least 50% (Table 6.4).

As a result, shellfish landings estimates submitted by town constables are generally considered more reliable, but the reliability of these estimates too, can vary from town to town. First, the method of estimation is not standardized among municipalities. In some cases, constable reports are based on the number of diggers, ability of the digger, and a production rate estimate for each flat dug. Such an assessment requires that the constable have an intimate knowledge of the harvesters and the resource harvested and has time allocated to monitor both. In others, the towns rely on a written report from the commercial fisherman coupled with spot checks of catch by the local warden. There are still other towns where the constable simply bases landings estimates on the number of permits issued and the quota allowed, assuming that every fisherman has caught his quota on every day in which fishing can take place.

Another difficulty in determining LPUE from Massachusetts landings reports lies with the method of reporting the number of licenses issued. The state requires that only the total number of shellfish harvesting permits issued by the town be reported. In some towns more than one species may be fished commercially. For example, many of the towns in Buzzards Bay harvest both quahogs and bay scallops. Therefore, it is impossible in most cases to determine the actual number of shellfishermen harvesting each resource.

Given the shortcomings outlined above, the usefulness of LPUE estimates calculated from Massachusetts DMF shellfish landings reports is limited. However, in the absence of independent stock assessment data, it provides the only source of information currently available to assess trends in abundance of nearshore bivalve stocks. With these serious limitations in mind, the following assessment was made for the soft-shell clam resource in Massachusetts.

Estimates of landings per unit effort based on constable "catch" statistics (bushels per year) and the number of commercial shellfish permits issued were calculated for soft-shell clam stocks from the following cities/towns: Rowley, Newbury, Quincy and Gloucester (Figure 6.4). These towns were chosen for three reasons. First, soft-shell clams represent the only commercial bivalve resource in these areas and hence it could be assumed that all reported fishing effort was on this resource. Secondly, it could be assumed that the acreage open to shellfish harvesting has remained unchanged. In fact, increases in the amount of acreage closed to shellfishing on the North Shore were negligible during the 1980s, probably because this area of the coast has not experienced the rapid increase in development and population compared to other parts of Massachusetts during this time period (Buchsbaum, 1992). Thirdly, the number of commercial licenses issued provides a fair estimate of the actual fishing effort applied in any year. Even though the number of recreational permits issued far exceeds commercial ones in many towns, "mess" diggers account for less than 20% of the total catch reported.

The absence of any clear trend in the annual landings per unit effort values suggests that soft-shell clam stocks have fluctuated dramatically in abundance during the period of analysis, especially in the towns of Gloucester, Newbury and Rowley (Figures 6.4 a,b,d). In Gloucester, peak periods of abundance occurred during the early 1970s and around 1980, with periods of rapid decline following. These periods of clam abundance were probably due to an unusually good spatfall which sustained the fishery for a few years (R. Knowles, pers. comm.). In Newbury peaks of abundance occurred in the late 1970s and late 1980s whereas in Rowley soft-shell clam abundance peaked in the mid to late 1970s.

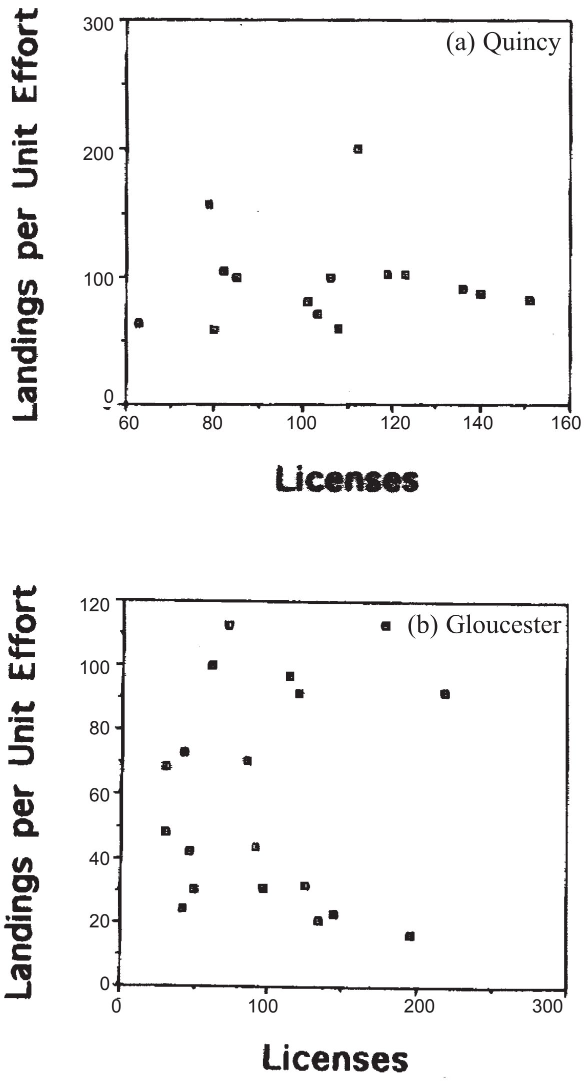

Figure 6.5. Relationship between landings per unit effort for soft-shell clam landings and the number of commercial licenses issued for two municipalities. a) Quincy, MA, 1978-1993; b) Gloucester, MA, 1972-1993.

Of the four data sets, however, the information from Quincy is probably the most reliable (Figure 6.4c). All clams taken legally from Quincy flats must be depurated before being sold. The landings data reported by Quincy are the actual number of bushels taken from Quincy flats which passed through the depuration plant in Newburyport. Unless some clams were reaching market illegally, these production figures should account for all the clams taken from the flats in those years. Except for two years of high abundance (1983 and 1992) the reported soft-shell clam production per digger hovered around 75 bushels per year. Again, there is no clear indication from these data that LPUE has shown a decline over the years studied.

The relationship between LPUE and the number of commercial licenses issued for the cities of Gloucester and Quincy is shown in Figure 6.5. While any plot of landings per unit effort against effort must be treated with some reserve, decline in LPUE as fishing effort increases is usually regarded as an indication that a stock is overfished (Gulland, 1974). The scatter of points for Gloucester data and the steady clam production from Quincy flats does not support the contention that resources in these areas are overfished. To some extent this is a self-regulating fishery. Unlike offshore fishermen who must maintain a boat and crew, the capital investment to dig inshore bivalves is small and the diggers will probably choose other employment in years when the resource is not abundant enough to justify the effort.

In the absence of truly reliable stock assessment data it can only be said that the available information does not support the belief that soft-shell clam resources of the North Shore are overexploited. The inability to make even these crude assessments for either the quahog or the bay scallop, however, only emphasizes the critical need for more reliable stock assessment information for all commercially fished inshore bivalves. Most importantly, there is a need for accurate landings information. A system that allows management personnel to determine the volume of shellfish landed on a daily basis should be the goal. Secondly, better ways of estimating fishing effort are needed. A measure of effort such as "hours dug" would provide a more instructive measure of the actual digging pressure on a population. In a study done by Creaser and Packard (1993) information on catch/effort and landings was recorded during all tides fished from a population of soft-shell clam in Machiasport, Maine that had recently been opened for depuration digging. This study could serve as a model to managers from other locales interested in generating reliable catch statistics for their fishery.

The city of Gloucester is presently attempting to generate more accurate landings information for clamflats within the city (R. Knowles and D. Sargent, pers. comm.). Lack of adequate personnel, however, has limited this effort to flats designated as “management areas”. In nearby Salem, a citizen volunteer group (Salem Sound Coastwatch) conducted transect surveys to determine soft-shell clam densities in areas of Salem Sound which have been closed to shellfishing for over 30 years (B. Chase, pers. comm.). Sites in Salem Sound, Massachusetts, sampled as part of the DMF's marine resources study in the 1960s (Jerome et al., 1967) have been resampled and may provide some interesting comparative data. However, these efforts are largely local and small in scale. More resources should be made available to municipalities so that this type of approach can be extended to all areas in the State which support shellfishing activity. A program similar to the Massachusetts Coastal Commercial Lobster Trap Sampling Program (Estrella and Cadrin, 1992), is needed for inshore bivalve resources so that more accurate trends assessment is possible.

FISHERIES ASSESSMENT

> *We have left undone those things which*
> *we ought to have done; And we have*
> *done those things which we ought not to*
> *have done...*
> *-Book of Common Prayer*
> *(Anon, 1928)*

“Overfishing”, as it relates to marine resources, can be defined in three ways: (1) as the removal of so many animals from a biological community that ecologically related or dependent species are negatively affected; (2) as the removal of so many animals from the population that over time the average size of harvested individuals is reduced dramatically (growth overfishing); or (3) as a fishing effort so intense that the number of animals harvested over time declines as a result of lowered reproductive output in the harvested population (recruitment overfishing). The latter definition is operative in much of the finfisheries literature (Gulland, 1974; Beverton and Holt, 1957; Ricker, 1975) and is probably relevant to invertebrate stocks as well. Nevertheless, efforts to predict fishing intensities at which this effect will be felt in shellfish stocks have lagged far behind similar efforts for finfish populations. Over the past few decades, however, there has been some progress in the knowledge of the dynamics of invertebrate stocks and a resultant interest in the development of methods of assessing and managing these resources. As these approaches have become more refined, an understanding of the life history, environment and ecological interactions of the organism has taken on increasing importance.

Application of Fisheries Models

Mathematical models have been used by fishery scientists in attempts to understand the mechanisms driving population dynamics in species. The value of such models to the manager is that they provide a framework in which to study the consequences of possible management actions. The three types of models most commonly used in fisheries assessment are: (1) surplus production and yield-per-recruit models (Beverton and Holt, 1957), (2) stock-recruitment models (Ricker, 1975) and (3) matrix population models (Leslie, 1945; 1948). Structural models such as surplus production and yield-per-recruit (Y/R) models can be used to predict changes in yield or Y/R with variations in fishing intensity. Stock-recruitment models may be used to predict recruitment levels for a given spawning stock size. Matrix populations models are often used to predict changes in population size based on fixed schedules of vital rates (age or size-specific birth and death rates), i.e. life tables, for the population under study. The success of these models as predictors, however, lies in the degree to which their assumptions are met. While all three types of models have been applied to bivalve populations, the use of more holistic models that integrate factors such as natural environmental variability, contamination, habitat impacts, and fishing pressure have yet to be attempted.

Matrix population models have been used extensively for analyzing life history tactics in a variety of species (Hartshorn, 1975; Longstaff, 1977; Caswell and Werner, 1978; Enright and Ogden, 1979; Pinero et al., 1984; Levin et al., 1987), and the Leslie matrix model (Leslie, 1945; 1948) in particular, has long been used to estimate population size. These models, however, rely on the availability of age (size) - specific schedules of births and deaths (life tables), information that is not yet available for all the species of bivalves discussed here. Additionally, in their usual form, these models are deterministic. Consequently, they are not entirely appropriate either for shellfish or other marine species in which recruitment of juveniles is highly variable from year to year and controlled largely by environmental parameters not yet fully understood. Many extensions to early models of population growth have been developed, including viewing some of the life history parameters as random variables and incorporating the effects of harvesting into the model (Beddington and Taylor, 1973; Rorres and Fair, 1975). The next step is to develop generalized optimal harvesting strategy models for the stochastic case.

Limited attempts to incorporate more realistic treatments of recruitment variability into modelling efforts aimed at assessing management strategies have been made. In a study of yield sustainability under constant-catch policy and stochastic recruitment for the Atlantic surf clam, *Spisula solidissima*, Murawski and Idoine (1989) assumed a binary pattern of recruitment in which year-class strength is uniformly poor except during relatively infrequent years when exceptionally strong cohorts are recruited. Ripley and Caswell (1996) introduced stochastic recruitment (log normally distributed) to a stage-structured matrix model of clam populations. In a preliminary study of the soft-shell clam, *Mya arenaria*, using computer simulations to estimate the mean and range of population size projected over many decades, yearly larval settlement rates were varied randomly while all other vital rates were held fixed (Brousseau et al., unpubl.). The study found that an adaptive harvesting strategy (harvesting intensity is adjusted according to the settlement rates during the recent past) gave reasonable yields while protecting the standing stock. The major limitation of all these studies, however, is the inability to verify the range of input parameters used. Consequently, it is uncertain whether or not the conclusions reached are directly applicable to populations in the field.

Better empirical information is needed concerning the distribution pattern of settlement rates in shellfish populations. Sensitivity analysis of population growth rate to changes in the life history parameters of several species of commercially important shellfish has shown that population growth rate is more sensitive to changes in larval survival/early recruitment than to changes in other life history parameters such as fecundity and adult survivorship (Brousseau and Baglivo, 1984; Malinowski and Whitlatch, 1988). The importance of this early fluctuating stage in the life history of these species predicates the need to focus more attention on understanding the relationship between stock density and recruitment rates, the long-term pattern of recruitment events and the role of hydrodynamics in the settlement process in order to improve the usefulness of population models in applications to marine species. Intensive field work focusing on bivalve larval biology in natural systems must be done in spite of its high cost and labor-intensive nature.

SUSTAINABILITY OF THE BIVALVE FISHERIES

Defining 'overfishing' as that activity which directly leads to declining stocks over time is related to an important finfisheries concept known as maximum sustainable yield (MSY). As fishing pressure increases more individuals of progressively smaller size are harvested until the decreasing size of the animals results in decreases in total catch size (weight) despite the increased numbers harvested (growth overfishing). The MSY for a commercial finfish species is estimated based on records of commercial catch, size and age of harvested species and annual recruitment variability. Structural models, such as surplus production and yield-per-recruit models developed initially for finfish, have been applied to invertebrates in isolated cases (Caddy, 1980), but in general have been of limited usefulness in estimating yields in exploited molluscan stocks. One problem with such models is their dependence on adequate information on the intensity of the fishing effort (see discussssion above).

Another problem central to the difficulty in maintaining sustainable shellfisheries is the inability to define the management unit. The term biological "stock" has often been used to describe a discrete, self-perpetuating population of organisms that share a common gene pool and can be managed (Larkin, 1972). The biological stock is now viewed by many in the fisheries community as the management unit. Understanding the genetic structure of an exploited species is the first step in devising management strategies that ensure the long-term survival of a fishery.

Recent developments in molecular techniques have made several types of genetic markers (mitochondrial DNA, nuclear DNA and allozymes) available for assessing population-level structuring on local and regional geographic scales. These techniques have been widely used in finfisheries research to estimate intraspecific genetic variation as well as population allocation to mixed-stock fisheries for a number of commercial species including cod (Pogson et al., 1995), bluefin tuna (Grewe et al., 1997), salmon (Scribner et al., 1998) and red mullet (Mamuris et al., 1998). Such stock identification has become a major focus of research efforts aimed at assisting in the formulation of marine finfishery management decisions.

Molecular techniques have been used less widely in efforts to delineate shellfish stocks. Information is beginning to emerge for such exploited molluscan species as abalone (Shepherd and Brown, 1993), deep-sea scallops (Wilding et al., 1998) and limpets (Weber et al., 1998) as well as for soft-shell clams (Morgan et al., 1978; Caporale et al., 1997), quahogs (Dillon and Manzi, 1992; Juste, 1992) and bay scallops (Bricelj and Krause, 1992; Wilbur et al., 1999). Large-scale research efforts aimed at defining the stock boundaries for such widely-distributed species such as *M. mercenaria* and *M. arenaria* are needed, however, if effective management of the resource is the goal.

For sedentary species, such as bivalves, the difficulty associated with defining the population unit greatly complicates stock assessment calculations. The concept of the metapopulation has been used by population biologists to describe the dynamics of spatially fragmented subunits of species which are linked together by dispersal stages (Hanski and Gilpin, 1991). Such analysis could have application in the management of economically important species such as scallops and clams whose stock units may occupy as large an area as a sea, gulf or estuary or as small an area as a single shellfish bed. Decisions regarding management of such resources depend in large part on an understanding of the relative importance of local (demographic) versus regional (recruitment and/or emigration) processes in the overall maintenance of the population unit. In order to assess the relative importance of these factors, however, a clear understanding of stock structure is needed.

Related to the difficulty of defining the "unit stock" is the difficulty in establishing an overall stock-recruit relationship for sedentary molluscan populations. Stock-recruitment models have been used to predict recruitment levels for a given spawning stock size in various finfish populations (Ricker, 1975). Hancock (1973) reported, however, that there is little evidence to indicate a direct relationship between spawning stock size and recruitment in an exploited population of cockle (*Cardium edule*). He concluded that "heavy spatfall may occur in a whole range of circumstances, including (1) when adult stocks are high or low, (2) when predation has been reduced or (3) when conditions for larval survival and settlement are especially good, or any combination of the three." Data on stock and recruitment for most species he discusses are so limited, however, that to generalize for species other than the cockle is unwise. (This difficulty is also described for lobsters by Steneck, Chapter 8). It may simply be that stock-recruitment relationships are masked by the difficulties associated with defining the stock "unit" as discussed above. It is too early to conclude that the size and/or demographics of the parent stock has little influence on reproductive success or failure in invertebrate stocks.

CONTAMINANTS AND HABITAT DEGRADATION

Increased population pressures during the past twenty-five years leading to overdeveloped shoreline areas and increased threat of bacterial contamination have been well documented, especially for the South Shore of Massachusetts (Buchsbaum, 1992; MBP, 1996). This has led to the largest rate of increase in regional shellfish closures in the State due to contamination by fecal coliform bacteria. The North Shore experienced lower closure rates than other regions during this period only because of its long history of shellfish closures due to fecal contamination. As the number of shellfish closures rise, harvesting pressure on remaining beds becomes more intense, increasing chances that overfishing will occur.

Increased development brings other changes that influence shellfish habitat and productivity, such as the addition of nutrients and toxicants to estuaries and embayments, alteration or restriction of tidal flow due to roads, bridges, piers and shoreline armament, increased siltation caused by altered land-use practices, and conflicts brought about by the increased use of the nearshore environment for recreational activities (Deegan and Buchsbaum, Chapter 5; McDowell, Chapter 7). Eutrophication of estuaries may accompany land-based development (Menzie-Cura, 1996) and may impact shellfish beds by altering the food supply (sometimes increasing shellfish productivity). More often than not however, increased primary production results in more frequent periods of low dissolved oxygen through increased respiration of the primary producers and associated community, particularly during warmer weather and overcast days. Sediments may become hypoxic or anoxic, and even shellfish that can survive prolonged periods of low dissolved oxygen (*M. arenaria*, *M. mercenaria*) can become stressed (Newell and Hidu, 1986). Juveniles are particularly susceptible and may die if dissolved oxygen levels persist for prolonged periods. The presence of increased algal mats or degraded sediments may also interfere with spat settlement.

The majority of the contaminant and habitat degradation impacts listed above (and described in more detail by Deegan and Buchsbaum in Chapter 5 and by McDowell in Chapter 7) are rather localized, and do not impact the entire metapopulations of soft-shell clams, quahogs and bay scallops uniformly. Thus the impacts of contaminants, habitat degradation, and overfishing are difficult to assess not only at the metapopulation level, but also at the local population level because of the broad dispersive abilities of the planktonic larvae. The analysis presented in this chapter indicates that we are unable to measure and document any significant impacts from the combined stresses of fishing, contamination and habitat destruction. There is no clear evidence of overfishing, at least not for the Massachusetts soft-shell clam fishery. While it is likely that both contaminants and habitat alterations contribute to larval mortality and reduced recruitment, the importance of these additional stresses cannot be measured at the present time, especially against the backdrop of extreme interannual variability in recruitment that would be observed in the absence of these anthropogenic stressors.

CONCLUSIONS

In view of its importance both to the public health and to an industry of such magnitude I earnestly recommend that a commission be appointed and an appropriation made to cover a thorough investigation of the entire subject of the pollution of our clam flats...

-Z.A. Howes, 1930
(in Belding, 1930a)

This chapter assessed the impact of natural mortality and harvesting pressure on inshore bivalve resources, using data from Massachusetts as the primary example. Based on this review, the only fair conclusion to be drawn is that information currently available for assessment is inadequate to determine whether or not a statewide decline in these commercial stocks has occurred. The LPUE statistics calculated for the soft-shell clam do not support the view that stocks have declined over the past 25 years in the four North Shore communities studied, but that conclusion is based on very limited data. Lack of appropriate data to make even the crudest assessments for quahogs and bay scallops makes it nearly impossible to comment on the status of those species in the state. The other Northeast states have equally poor data sets or lack critical data entirely on these three species, making defensible stock assessments all but impossible.

Added to the difficulty of documenting stock trends is the inherent problem of identifying and assessing natural versus man-made sources of mortality. In addition to the various anthropogenic causes of mortality (pollution, habitat destruction, and overharvesting), natural mortality from an adverse environment, predation, competition and disease also contributes to fluctuations in species abundance. The relative importance of each of these factors in the overall picture is far from understood, but almost certainly, no one cause is responsible, nor are the same cause/causes responsible for all the species discussed.

In clams and scallops, which have vulnerable planktotrophic larval stages, natural mortality is extremely high early in life, resulting in complete recruitment failure during some breeding cycles. It has been suggested (Hancock, 1973) that the relationship between stock and recruitment in such species is so tenuous that the occurrence of a heavy spatfall is equally likely whether adult stocks are high or low, raising a question concerning the benefits of managing or protecting exploited stocks. The perceived absence of a stock-recruitment relationship for invertebrates, however, is more likely the result of a failure to view local clam flats or scallop beds as spatially fragmented subpopulations of a larger unit, the dynamics of which can be understood only if all of the subpopulations are considered together. Viewing resource units as part of a series of local subunits which in part, owe their persistence to the dynamics of other local subpopulations, may provide important insights into the management and protection of these areas.

It seems undeniable that active harvesting, which removes adults from a population, will ultimately affect reproductive output and overall mortality levels in a population. Whether or not such alterations lead to reduced productivity over time, however, is less easily determined. No data are available to assess the impact of harvesting for either the quahog or the bay scallop. Analysis of LPUE versus fishing effort for soft-shell clams indicates that this species is not overfished, but the data on which this analysis is based are questionable at best. The absence of reliable statistics for the assessment of long-term population trends makes it impossible to determine the extent to which reported declines in these resources are the result of overfishing and not simply the result of natural fluctuations in species abundance.

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Chapter VII

Biological Effects of Contaminants on Marine Shellfish and Implications for Monitoring Population Impacts

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INTRODUCTION

> *The integrity of the world's coastal waters is jeopardized by the deliberate and inadvertent entries of society's discards. Many substances introduced by mankind are toxic to marine organisms, thus impinging upon the health of ocean communities or restricting the human consumption of fish and shellfish.*
> *-Edward D. Goldberg, 1980*

The use of a sentinel species as an indicator of chemical contamination has been widely used in monitoring programs in the marine environment (Bayne et al., 1988; Jones et al., 1995). This approach has led to greater insights on the spatial and temporal distribution of contaminants and associated effects on sentinel species (Butler, 1973; NRC, 1980; Farrington et al., 1983; Bayne et al., 1988). Bivalve molluscs, including several species of mussels, oysters and clams, have been the most commonly used sentinel species in Mussel Watch monitoring programs.

Although our knowledge of the distribution of specific compounds and groups of compounds continues to increase, our understanding of cause and effect relationships between classes of contaminants and specific biological effects in bivalve molluscs is still lacking. Natural biogeochemical processes that ultimately control contaminant bioavailability and uptake by bivalve molluscs must also be examined when evaluating the potential of bivalve molluscs as indicators of chemical contamination. These biogeochemical processes influence contaminant persistence and bioavailability, ultimately controlling the fate and effects of these contaminants in coastal marine environments.

The purpose of this chapter is to evaluate the effects of contaminants on molluscan shellfish from the New England area, and to place these effects in context against the impacts caused by habitat degradation and overfishing. Much of the bivalve monitoring data that has been collected in the region can be used to augment findings from laboratory and field studies, and help us to assess the impacts of contaminants on populations and individuals.

CONTAMINANT DISTRIBUTIONS IN SEDIMENTS AND SHELLFISH

Regional studies in the Gulf of Maine have documented the spatial distribution of several classes of contaminants including trace metals, chlorinated pesticides, polychlorinated biphenyls (PCBs), and polycyclic aromatic hydrocarbons (PAHs) in sediments and biota (Larsen, 1992; Kennicutt et al., 1994). These studies have been reviewed in detail elsewhere (McDowell, 1995, 1997). The relationship between contaminant inputs and the distribution of contaminants in sediments and biota largely reflect a gradient, with nearshore areas, especially urban and industrialized areas, having the highest levels of contamination, and offshore areas having significantly lower concentrations.

The first U.S. Mussel Watch program (1976-1978) provided a regional assessment of contaminant distribution in bivalve samples from New England waters (Farrington et al., 1983; Goldberg et al., 1983). Data collected in this program documented the strong urban influence on contaminant distribution in mussel samples for both trace metal and organic contaminants. A recent review of a decade of data collected in the Mussel Watch component of National Oceanic and Atmospheric Administration's (NOAA) National Status and Trends Program (mid-1980s to mid-1990s) concludes that the concentrations of contaminants in bivalve samples are declining for many classes of contaminants (O'Connor, 1998). Exceptions to this general conclusion are reflected in the data for organic contaminants and lead, particularly at stations in urban areas such as Boston Harbor.

To a large extent contaminant distribution in sediments and biota reflect not only contemporary inputs but also a history of industrial activity. For example, chromium contamination in certain locations within the Gulf of Maine ecosystem - Great Bay Estuary (NH), Saco River (ME), and Salem Harbor (MA) - reflect a history of inputs from the once thriving tanning industry (Capuzzo and Anderson, 1973; Armstrong et al., 1976; Mayer and Fink, 1980; NOAA, 1991). Concentrations of other trace metals are elevated at other locations - Boothbay Harbor, Boston Harbor and Quincy Bay - and reflect a pattern of wastewater input and other industrial sources of contaminants to shallow water embayments (NOAA, 1989, 1991; Sowles et al., 1992; Jones et al., 1995). Hydrocarbon inputs may also vary spatially and temporally as a result of chronic municipal discharges, agricultural practices, oil spills and other point and non-point sources. In the Gulf of Maine there are numerous locations that have received inputs of petroleum hydrocarbons from both chronic discharges and accidental spills [Boston Harbor (MA), Casco Bay (ME), and Penobscot Bay (ME)] (Johnson et al., 1985; Larsen et al., 1986; MacDonald, 1991; Menzie-Cura & Associates, 1991; NOAA, 1991). The use of chlorinated pesticides in agricultural practices has declined since the early 1970s but traces of pesticide residues have been reported at locations within the Gulf of Maine subjected to inputs from agricultural runoff (Hauge, 1988; Larsen, 1992; Kennicutt et al., 1994). NOAA's National Status and Trends Mussel Watch Program also noted elevated concentrations of aromatic hydrocarbons, chlorinated pesticides and other chlorinated hydrocarbons in bivalve samples, especially in urban harbors and industrialized areas (NOAA, 1989; Jones et al., 1995; O'Connor, 1998).

Trophic transfer of contaminants to higher level predators and the human consumer are generally most significant for lipophilic contaminants such as chlorinated hydrocarbons, and other persistent organic pollutants (POPS). Shellfish closures and advisories based on chemical contamination are relatively few but include some examples from the New England coast, notably PCB contamination in New Bedford Harbor and dioxin contamination in Maine (McDowell, 1997).

TOXICOLOGICAL EFFECTS OF CONTAMINANTS ON SHELLFISH

> *The relationship of disease and environmental stress is becoming increasingly well established with time. Human activities - particularly those that result in chemical additions to the coastal/estuarine environment - have increased the potential stresses on fish and shellfish inhabiting those areas. Circumstantial evidence for associations of pollutants with certain fish and shellfish diseases and abnormalities is accumulating.*
>
> *-Carl J. Sindermann, 1979*

The effects of chemical contaminants on marine bivalve molluscs have been examined extensively during the past two decades. The majority of the studies have been conducted on the blue mussel *Mytilus edulis* (e.g., Bayne et al., 1985, 1988) with an effort to integrate responses over several levels of biological hierarchy (Table 7.1) and to examine responses linked to specific classes

Table 7.1. Response levels of marine organisms to chemical contaminants; adapted from Capuzzo (1981).

Level	Types of Responses	Effects at Next Level
Biochemical-Cellular	Toxication Metabolic impairment Cellular damage Detoxication	Toxic metabolites Disruption in energetics and cellular processes Adaptation
Organismal	Physiological changes Behavioral changes Susceptibility to disease Reproductive effort Larval viability Adjustment in rate functions Immune responses	Reduction in population performance Regulation and adaptation of populations
Population	Age/Size structure Recruitment Mortality Biomass Adjustment of reproductive output and other demographic characteristics	Effects on species productivity and coexisting species and community Adaptation of population
Community	Species abundance Species distribution Biomass Trophic interactions Ecosystem adaptation	Replacement by more adaptive competitors Reduced secondary production No change in community structure and function

of contaminants. Recent work has extended this approach to other species of bivalve molluscs and to assessment of population level responses (Widdows et al., 1990; Leavitt et al., 1990; Weinberg et al., 1997; McDowell and Shea, 1997).

UPTAKE AND ACCUMULATION

Understanding the relationship between sediment contamination and potential for uptake and accumulation of contaminants by benthic organisms is a challenging problem. Recent advances in sediment geochemistry research have increased our understanding of processes controlling bioavailability and uptake by benthic organisms. The accumulation of trace metal and organic contaminants by aquatic organisms is a complicated function of physical, chemical, and biological processes that influence exposure concentrations, bioavailability, and uptake, elimination and storage of contaminants by an organism (Fisher, 1995). In the benthic environment, nonpolar organic contaminants will partition among all accessible phases according to the capacity of each phase to accumulate the contaminant. Usually, these partitioning processes are described using equilibrium models, where equilibrium among all phases is assumed. This forms the basis for Sediment Quality Criteria based on equilibrium partitioning (Shea, 1988; Di Toro et al. 1991). Recent studies, however, suggest that bioavailability of lipophilic contaminants is not based on equilibrium theory alone (McGroddy and Farrington, 1995; McDowell and Shea, 1997). The hydrophobicity of specific contaminants, the source of contaminants, and the sorption of contaminants (especially pyrogenically derived PAH) on organic carbon particles can greatly influence the rates at which equilibrium may (or may not) be obtained, and hence the availability and uptake of contaminants by benthic species.

Similar concerns exist when considering the bioavailability of trace metals to benthic organisms. Characterizing the bioavailability of trace metals based on the acid volatile sulfide fraction of sediment led to some greater predictability of bioavailability potential (DiToro et al., 1990) but other sediment features and processes (e.g., POC, DOC, metal hydroxides, redox, bioturbation) may also influence bioavailability (Valette-Silver, 1999). Luoma et al. (1997) suggested a combined approach utilizing field observations of sediment concentrations and geochemical properties and laboratory observations of uptake and elimination of specific trace metals from both dissolved and particulate phases (Luoma and Fisher, 1997; Wang et al., 1997). Bioenergetic-based kinetic models are being developed that can better predict the relationship between field and laboratory observations of uptake and accumulation of metals in benthic organisms (Wang et al., 1996; Wang and Fisher, 1997).

BIOTRANSFORMATION AND DISEASE RESPONSES

Research on biotransformation mechanisms in marine bivalve molluscs has paralleled efforts on vertebrate species for over two decades. In comparison, bivalve molluscs have been considered to have a relatively low capacity for detoxifying organic contaminants through cytochrome P-450 monooxygenase reactions (Anderson, 1978; Livingstone and Farrar, 1984; Stegeman, 1985). The dominant metabolites of benzo(a)pyrene detected in molluscs have been primarily quinone derivatives, rather than the diol derivatives observed in fish (Stegeman, 1985; Stegeman and Lech, 1991), although Anderson (1985) did observe relatively high concentrations of diol derivatives as well. Stegeman (1985) suggested that PAH metabolism in bivalve molluscs may proceed through several catalytic mechanisms including peroxidative mechanisms in addition to cytochrome P-450 monooxygenase. The formation of oxyradicals and binding of these reactive compounds to DNA and other macromolecules (Livingstone et al., 1990; Garcia-Martinez and Livingstone, 1995) pose a link with observations of cell damage noted by other investigators. Metabolism of other compounds such as aromatic amines yields metabolites with mutagenic properties (Anderson and Doos, 1983; Kurelec et al., 1985; Kurelec and Krca, 1987; Knezovich et al., 1988) and DNA adducts (Kurelec et al., 1988). The relationship between biotransformation and disease processes in bivalve molluscs has been suggested by several investigators (Moore et al., l980; Stegeman and Lech, 1991). The reactive compounds formed during biotransformation could result in histopathological damage of molluscan tissues.

Mix (1986, 1988) reviewed the relationship among contaminant tissue burdens, biotransformation and histopathology/disease in marine bivalve molluscs. Although no conclusions could be made, he suggested that our limited understanding at that time of specific contaminant effects on cellular and physiological processes and mechanisms of biotransformation hindered our ability to explore the relationship between contaminant exposure and disease progression. Gardner et al. (1991) reported promising evidence on the relationship between contaminant distributions and metabolism and the prevalence of specific tissue neoplasias in the oyster *Crassostrea virginica* with exposure to sediments from Black Rock Harbor (Long Island Sound, USA; Table 7.2). As information continues to be gathered on the relationship between shellfish diseases and contaminant accumulation and transformation, the role of contaminants in disease processes should be elucidated.

Table 7.2. Concentration of organic contaminants in oysters, *Crassostrea virginica*, exposed to sediments from Black Rock Harbor[a].

Compound	Oysters ng/g dry wt.	Sediments ng/g dry wt.	Ratio Oyster/Sediment
Sufficient evidence as carcinogens			
Benz(a)anthracene	695	3450	0.20
Benzo(a)pyrene	88	3160	0.02
Benzfluoranthene	364	5970	0.06
Indeno(1,2,3-cd)pyrene	22	--	--
Dibenz(a,h)anthracene	9	--	--
Hexachlorobenzene	0.2	--	--
Chlordanes	100	--	--
Limited evidence as carcinogens			
Chrysene	1260	4450	0.28
Inadequate evidence as carcinogens			
Benzo(e)pyrene	264	2880	0.09
Fluorene	42	635	0.07
Phenanthrene	553	4020	0.14
Perylene	17	504	0.03
Benz(g,h,i)perylene	37	--	--
Coronene	1.3	--	--
No evidence as carcinogens			
Anthracene	191	1330	0.14
Fluoranthene	1777	5800	0.31
Promoters			
DDT and metabolites	1183	--	--
PCBs (Aroclor 1254)	1143	7170	0.16
Pyrene	2950	7250	0.41

[a] Data from Gardner et al. (1991)

CELLULAR AND PHYSIOLOGICAL RESPONSES

Cellular and physiological responses of bivalve molluscs to contaminants provide the basis of linking observations of contaminant chemistry with observed disruption in physiological function. Numerous indicators of cell function have been proposed as biomarkers of cell damage in response to contaminant exposure. Alterations in lysosomal structure and function are consistent with observations of degeneration of digestive gland epithelium, atrophy of digestive tubules, and degeneration of reproductive tissues (Lowe et al., 1981; Moore and Clarke, 1982; Couch, 1984; Pipe and Moore, 1985; Lowe and Pipe, 1985, 1986, 1987; Moore et al., 1989). These observations have been linked in bivalve molluscs with exposure to high levels of lipophilic contaminants in the mussel *Mytilus edulis* (Lowe, 1988; McDowell et al., 1999). Lowe and Pipe (1987) suggested that the reallocation of energy reserves from resorbed oocytes to storage cells might serve as a resistance strategy to survive the effects of hydrocarbon exposure.

Ringwood et al. (1999) observed alterations in lysosomal function and glutathione concentrations in juvenile oysters (*Crassostrea virginica*) exposed to contaminated sediments with a mixture of trace metals. Responses of cell function varied significantly with contaminant loading and the data agreed well with other estimates of sediment toxicity (Long et al., 1995).

Other indicators of contaminant effects in bivalve molluscs show promise as monitoring tools or biomarkers of exposure to chemical contaminants and biochemical or cellular damage. These include the presence of single-strand breaks or alkaline labile areas in the DNA complement of individual cells (Shugart et al., 1989) and the presence of stress proteins within the cell (Hightower, 1993). Many compounds have been shown to have genotoxic effects in marine organisms including methyl methane sulfonate (Nacci and Jackim, 1989) and N-methyl-N'-nitro-N-nitrosoguanidine (Nacci et al., 1992). In each case, significant increases in DNA breakage occurred in a dose dependent fashion. Cells have the capability of repairing damage to the DNA molecule (Martinelli et al., 1989), thus, providing a potential means of timing the exposure event and subsequent recovery.

Stress proteins are a group of proteins that are routinely synthesized within cells in response to exposure of the cell to a wide variety of physical and chemical conditions (Hightower, 1993). Some stress proteins are produced in general response to a wide range of stressors, whereas other stress proteins are unique to a specific chemical or physical stress (Bradley, 1993). A stress protein “fingerprint” can be measured and used as a marker of contaminant effects in the environment (Randall et al., 1989). At the present time, heat shock protein 60 (Sanders et al., 1991) and heat shock protein 70 (Steinert and Pickwell, 1993) appear to be appropriate as biomarkers of environmental stressors in marine bivalves. Recent studies by Clayton (1996), however, caution that the site of collection, season, and tissues sampled need to be carefully considered in order for heat shock proteins to be used as biomarkers of contaminant effects.

In addition to acting as indicators of contaminant exposure, the presence of these biomarkers may be indicative of sublethal damage to the organism that may have consequences for individual survival, reproduction, and population processes. The implications of genotoxic agents in terms of damaged DNA are obvious with respect to the overall impact on the transcription and replication of the DNA molecule. Sanders et al. (1991) observed the accumulation of heat shock protein 60 (hsp 60) in conjunction with a decrease in scope for growth (SFG) measurements in *Mytilus edulis* exposed to sublethal concentrations of copper. Accumulation of hsp 60 was a more sensitive indicator of copper exposure than reductions in bioenergetics as measured by scope for growth.

Alterations in growth rates of bivalve molluscs occur as a result of reductions in feeding rates, higher respiratory metabolism, and reduced digestive efficiencies. Reductions in physiological measurements (e.g., respiration rates, carbon turnover, and scope for growth) have correlated with reduced growth rates measured for bivalve populations from contaminated habitats (Gilfillan et al., 1976; Gilfillan and Vandermeulen, 1978; Capuzzo and Sasner, 1977). Alterations in bioenergetics and growth of bivalve molluscs following exposure to petroleum hydrocarbons appear to be related to tissue burdens of specific aromatic compounds (Gilfillan et al., 1977; Widdows et al., 1982, 1987; Donkin et al., 1990). Widdows et al. (1982) demonstrated a negative correlation between cellular and physiological stress indices (lysosomal properties and scope for growth) and tissue concentrations of aromatic hydrocarbons with long-term exposure of *Mytilus edulis* to low concentrations of North Sea crude oil. Recovery of mussels following long-term exposure to low concentrations of diesel oil coincided with depuration of aromatic hydrocarbons (Widdows et al., 1987). Donkin et al. (1990) suggested that reductions in scope for growth in *M. edulis* were related to the accumulation of two- and three-ring aromatic hydrocarbons, as these compounds induced a narcotizing effect on ciliary feeding mechanisms.

Diminished scope for growth, alterations in lysosomal function, and decreased reproductive effort appear to be general responses to contaminant exposure and may be indicative of general reduction in physiological conditions. The response of the turkey wing mussel (*Arca zebra*) to contaminants along a gradient in the waters surrounding Bermuda included reduced feeding rates and increases in metabolic expenditures associated with significant accumulation of lead, tri- and dibutyltin, petroleum hydrocarbons and their polar oxygenated derivatives, and PCBs (Widdows et al., 1990). Mussels collected along the same gradient showed changes in biochemical composition, especially in the ratio of neutral to polar lipids and carbohydrate content (Leavitt et al., 1990).

Mytilus edulis transplanted to New Bedford Harbor (Buzzards Bay, MA) showed reduced reproductive effort and increased degeneration and premature resorption of oocytes, coincident with high body burdens of PCBs and PAHs (Table 7.3, McDowell et al., 1999). The greatest differences in condition index and lipid reserves of mussels were observed during the pre-spawning period, consistent with the accumulation and utilization of lipid reserves for reproductive development. Following spawning no differences in condition index were evident and lipid reserves were diminished to minimum levels. Resident populations of mussels from New Bedford Harbor also showed extensive signs of gonad degeneration.

Table 7.3. Condition indices and reproductive effort of mussels, *Mytilus edulis*, transplanted to New Bedford Harbor, and reference sites.

Station	Maximum Condition Index Pre-spawning mg Dry Wt./Shell Volume[a]	Reproductive Effort % RE/Total Energy[b]
Nantucket Sound	335+30	0.88
Cleveland Ledge	340+20	0.71
New Bedford Harbor	230+15	0.34

[a] Data from McDowell Capuzzo (1996); Mean + 1 S.E.
[b] Calculated from mean values from eight individual animals at each site.

POPULATION LEVEL RESPONSES

> *The root problem is that we - and this includes ecotoxicologists and ecologists - still do not know enough about ecological systems to be able to identify what it is we want to protect about them and hence, what we should be measuring. Clearly, this is most acute at community and ecosystem levels.* *-Peter Calow, 1994*

Chronic exposure to chemical contaminants can cause alterations in reproductive and developmental potential of populations of marine organisms, resulting in possible changes in population structure and dynamics. It is difficult to ascertain, however, the relationship between chronic responses of organisms to contaminants and large-scale alterations in the functioning of marine ecosystems or the sustainable yield of harvestable species. Cairns (1983) argued that our ability to detect toxic effects at higher levels of biological organization is limited by the lack of reliable predictive tests at population, community, and ecosystem levels. Much research effort is needed in these areas before environmental hazards as a result of contaminant inputs can be adequately addressed.

Koojiman and Metz (1984) suggested that the sublethal effects of contaminant exposure should be interpreted in light of the survival probabilities and reproductive success of populations, thus bridging the gap between individual and population responses.

Although many indices have been proposed for evaluation of chronic responses of organisms to contaminants, few have been linked to the survival potential of the individual organism or the reproductive potential of the population (McIntyre and Pearce, 1980). Experimental studies directed at determining effects on energy metabolism or effects that influence growth and reproduction would be most appropriate for linking effects at higher levels of organization. When investigating biological effects of contaminants, many variables must be recognized and assessed. Differential sensitivity of different species of organisms, various life history stages, and species from different habitats may be related to contaminant bioavailability, capacity for contaminant biotransformation, and the metabolic consequences of contaminant exposure. The increased sensitivity of early developmental stages and the seasonal differences in the responses of adult animals may be related to stage-specific or seasonal dependency on particular metabolic processes (e.g., storage and mobilization of energy reserves, hormonal processes), with the result of altering developmental and reproductive success (Capuzzo, 1987).

Reproductive success and development of an organism may be affected by contaminant exposure by:

1. deposition of contaminants in gametes and developing embryos;
2. lysosomal dysfunction associated with oocyte resorption;
3. interference with feeding mechanisms, such that exposure mimics starvation responses;
4. failure to incorporate sufficient yolk in oocytes;
5. morphological abnormalities during embryogenesis resulting from failure of morphological systems to develop properly;
6. limited capacity of developmental stages to metabolize or depurate contaminants; and
7. limited capacity of early developmental stages and reproducing adults to draw on excess energy reserves (Capuzzo et al., 1988).

Thus, responses can be categorized as interfering with energetic processes (3, 4, and 7), biosynthetic processes (4), and structural development (2, 5) in addition to contaminant accumulation and depuration (1, 6). Alterations in bioenergetics linked with observations of reduced fecundity and viability of larvae, abnormalities in gamete and embryological development, and reduced reproductive success provide a strong empirical basis for examination of population responses. Incorporation of these responses in demographic models may lead to new insights on adaptations of specific life history stages to contaminant perturbations and the population consequences of stage- or age-specific effects of contaminants. Reduced scope for growth and decreased fecundity in mussels exposed to high levels of PCBs in New Bedford suggest population consequences (McDowell et al., 1999). For species that have planktonic life history stages, resettlement in highly contaminated areas may obscure demographic changes due to impaired bioenergetics.

The population dynamics of bivalve species have received considerable scientific attention due to the importance of many bivalves as commercially harvested fisheries. Demographic models have been developed to examine the importance of specific life history characteristics on population processes. Such models include: (1) analysis of the sensitivity of population growth rate to life cycle perturbation, (2) life table response experiments, and (3) population projection and prediction (Caswell, 1989a,b). In addition to quantifying the impact of fishing pressure on bivalve populations, demographic models have been used to assess the importance of environmental perturbations (e.g., disease, contaminant effects, etc.) on bivalve physiology and population dynamics (Weinberg et al., 1997).

Ayers (1956) suggested that larval mortality was one of the most important considerations in monitoring the population dynamics of the soft-shell clam, *Mya arenaria*, an observation consistent with numerous studies of bivalve species (Brousseau, 1978; Brousseau et al., 1982; Weinberg et al., 1986). Brousseau et al. (1982) suggested that larval mortality could be further separated into mortality that occurred during (a) fertilization, (b) the free-swimming larval phase, or (c) early post-larval attachment (see also Brousseau, Chapter 6). Using sensitivity analysis, Brousseau and Baglivo (1984) addressed changes in the population growth rate attributable to changes in settlement rates of larvae and in age-specific fecundity and survivorship rates of the soft-shell clam. They concluded that population growth rate was insensitive to absolute values in egg production and most sensitive to changes in egg and larval viability which contribute to the success of larval settlement. Malinowski and Whitlatch (1988) further documented that population growth rates were two to three orders of magnitude more sensitive to changes in survivorship in larval and juvenile stages of the life cycle than proportional changes in either survivorship or fecundity in adult size classes.

Since sensitivity analysis has identified that the larval stage is the most critical life history stage controlling population growth rate, experiments and field collections designed to quantify the vital rates associated with larval survival and viability are needed. Processes related to the allocation of energy to developing eggs and larvae that influence not just numbers of developing eggs but size and quality of energy reserves for larval development are especially important. These data can then be applied to a demographic model to ascertain how perturbations in larval viability may affect population growth and development. Any factor (e.g., disease, contaminant exposure, etc.) that alters the allocation of energy reserves to developing eggs and larvae may result in a reduction in larval viability and post-settlement success. Among the classes of contaminants that are prevalent in Boston Harbor and Massachusetts and Cape Cod Bays that may specifically alter energetic and reproductive processes in bivalve molluscs are the chlorinated hydrocarbons (including PCBs and pesticides) and polycyclic aromatic hydrocarbons. Population models could be used to differentiate the effects of contaminants, fishing pressure and habitat alteration on population structure of bivalve molluscs.

Studies have recently been completed in Massachusetts Bay to examine the effects of polycyclic aromatic hydrocarbons and chlorinated hydrocarbons on population processes in the soft-shell clam, *M. arenaria* (McDowell and Shea, 1997). Contaminants were detected in clam tissues and sediments collected along a sediment gradient of polycyclic aromatic hydrocarbon contamination in Boston Harbor and Massachusetts and Cape Cod Bays (300 to 66,000 ng per g dry weight), but the bioavailability of specific compounds varied at different sites. Estimates of the fraction of contaminants available in porewater and sediments for equilibrium partitioning (AEP) provided the best predictor of relative bioavailability.

The reproductive cycle of clam populations from the five sites varied with respect to the timing and extent of the spawning season but not with respect to the number of developing oocytes during a spawning event. Both female and male clams from the reference sites had advanced stages of gamete development during the late spring and spawning continued through the early fall. The large relative size of the digestive gland-gonad complex and accumulated lipid provided sufficient energy for this extended reproductive season. Populations from the upper Massachusetts Bay sites (Fort Point Channel, Saugus River and Neponset River) did not spawn until mid-summer and spawning occurred for only a short period of time. Asynchrony in gamete development between males and females was not observed at any of the five sites. In addition to an abbreviated spawning season, clam populations from the contaminated sites also showed a high prevalence of gonadal inflammation (cell proliferation) that was significantly different ($p<0.001$) from reference populations especially during the late fall to early winter (September to December). At the most contaminated site (Fort Point Channel), levels of hematopoietic neoplasia also reached 100% in December 1995 (McDowell and Shea, 1997).

Population growth rates were determined for all populations using a deterministic matrix model. Trends in population growth rates were not directly related to contaminant concentrations at each site, as other site features such as predator abundance and hydrographic features had strong influences on recruitment success (McDowell and Shea, 1997). The deterministic model was relatively insensitive to the differences in reproductive physiology related to contaminant exposure. High inter-annual and inter-site variability in recruitment patterns may mask contaminant effects on population processes. Stochastic models such as those developed by Ripley and Caswell (1996) may add more insights on variability in population structure as a result of the interactive effect of contaminants and other habitat features.

SUMMARY AND CONCLUSIONS

This chapter addresses the effects of contaminants on shellfish populations in coastal habitats. Accumulation of contaminants in shallow-water benthic habitats has led to contamination of shellfish resources at many locations along the New England coastline, especially in habitats adjacent to urban areas. Contaminated sediments have contributed to habitat degradation and have resulted in restricted access to shellfish resources. In spite of these problems, shellfish populations in contaminated habitats may be quite abundant even though reduced reproductive effort and high disease prevalence are also observed. Areas closed to fishing such as those studied in the more urban sections of Massachusetts Bay show populations with a wider distribution of size and age classes than those sites that are routinely harvested (Brousseau, Chapter 6). When moderately contaminated areas are open to periodic harvesting for relaying contaminated stocks, a discontinuity in size classes is observed (Brousseau, Chapter 6). The sporadic recruitment success of bivalve populations at shallow-water benthic sites appears to be the dominant feature influencing population size and age structure. Recruitment of newly settled bivalves to contaminated sites even when reproductive effort of adult shellfish at those sites is reduced will balance the losses related to disease and loss of reproductive potential. The comparison of contaminant effects and overexploitation of shellfish populations is difficult to make as fishing mortality may represent one of the major losses to population abundance in uncontaminated habitats. For shellfish stocks that are overexploited even in uncontaminated habitats sporadic recruitment patterns of many bivalve molluscs may require long periods between harvesting to compensate for the discontinuity in size classes. In contrast, the restriction of shellfish harvests in contaminated habitats may allow high population abundance of bivalve molluscs even though chronic sublethal toxic effects are common. Thus, the relative importance of contamination and overfishing on bivalve populations is difficult to assess, even at the local level.

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Chapter VIII

Are We Overfishing the American Lobster? Some Biological Perspectives

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Note: The following contribution was written in 1996 before the 2000 assessment was completed by the Atlantic States Marine Fisheries Commission. My article reflected management positions and approaches prior to the new assessment, and it considered the ongoing difficulty in determining whether lobster stocks are overfished. It is possible that several specific concerns identified in my paper will be addressed in that assessment. However, this paper predates the ASMFC assessment process and thus contains no information derived from that assessment. Two papers that had been "in prep" have since been published and are referenced as such in this chapter.

> *Unfortunately, for many years past we have watched...[the American lobster] decline until some have even thought that commercial extinction...awaited the entire fishery. What is the matter with the lobster?* *-Herrick, 1909*

> *Why are there so many American lobsters?* *-Miller, 1994*

INTRODUCTION

Because the American lobster, *Homarus americanus*, is the single most valuable species to the fisheries of New England, it is understandable that there is widespread concern for its health. For nearly a century this concern has centered on overfishing. However a growing number of scientists have questioned whether overfishing is its most serious threat, or if it is, how will we know?

When harvesting exceeds the ability of an exploited species to replace itself, it is overfished. There are myriad examples of overfished stocks that have collapsed such as cod, haddock and right whales. Then there is the unusual case of the American lobster. Despite repeated warnings for nearly a century that lobster stocks are overfished and collapse is imminent, stocks throughout the western North Atlantic remained stable and in recent years surged dramatically. In fact, landings in Canada and the United States in the 1990s have exceeded record highs and most fisheries scientists agree that this increase is primarily driven by high abundance rather than simply increased fishing effort (Elner and Campbell, 1991; Pezzack, 1992; Anon. 1993a; Miller, 1994). Were determinations of overfishing wrong? Should other factors not currently in the spotlight be considered more seriously? If the primary concern is for the health of lobster stocks, how might we best monitor it? In other words, how do we best put our fingers on the pulse of this marine resource? In this chapter, I will describe how overfishing on lobsters is currently determined, what strengths and weaknesses exist in this approach, and whether other factors such as environmental variability, habitat, and pollution have been sufficiently considered.

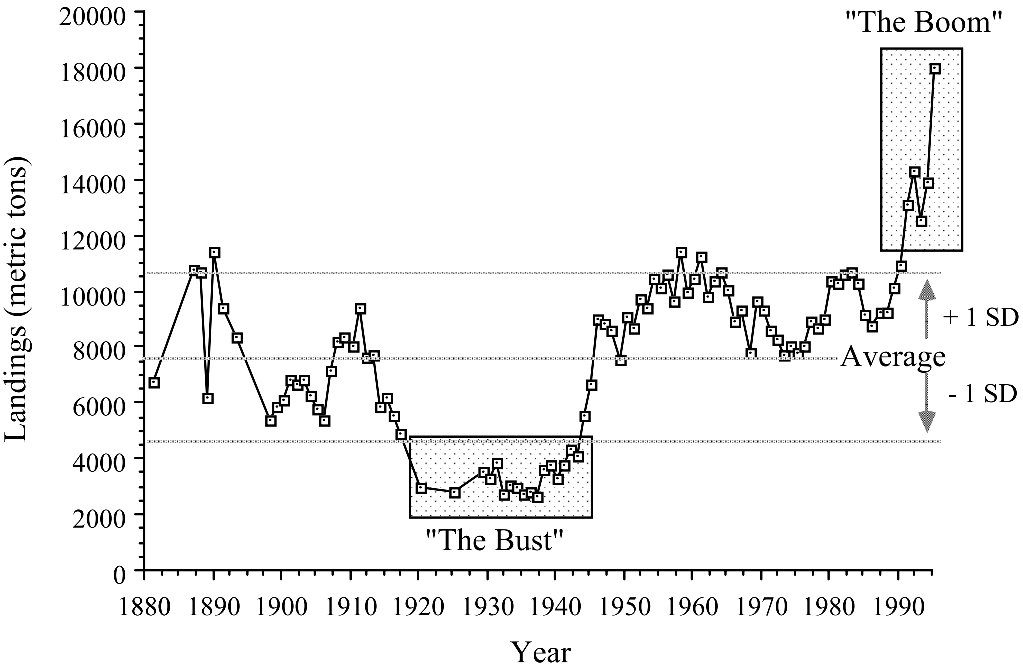

Figure 8.1. Lobster landings in Maine from 1880 to 1994 with average and variance (+ 1 standard deviation) over the period indicated by the three horizontal lines. The period below one standard deviation below the mean is called “the bust” and above one standard deviation above is called “the boom”. Data from Maine Department of Marine Resources.

TEMPORAL TRENDS IN AMERICAN LOBSTER AND FISHING EFFORT

My review focuses heavily on Maine because it has the largest harvest of lobsters in the United States and because good records have existed for more than a century. As others have pointed out (e.g., Elner and Campbell, 1991) some of the most striking patterns in Maine have been paralleled in most areas throughout the western North Atlantic.

One of the strongest patterns observed in the western North Atlantic is the decline in landings observed from the turn of the century to about 1925 (Figure 8.1). This pattern was evident in all major lobster producing regions of Canada (Nova Scotia, Newfoundland, and Gulf of St. Lawrence; Elner and Campbell, 1991; Pezzack, 1992; Anon., 1995) and the United States (Harding et al., 1983; Miller, 1994). In Maine, this resulted in the all-time population low that occurred between the World Wars and ended in the mid-1940s (labeled “The Bust” in Figure 8.1; Acheson and Steneck, 1997).

Equally striking as the bust is the “boom” period which began between the 1970s and 1980s and continues in some regions today (e.g., Elner and Campbell, 1991; Miller, 1994; Acheson and Steneck, 1997). In between the bust and boom periods, landings varied in different regions but in Maine and most of the Gulf of Maine there was a significant increase in landings during the 1940s. Although for the next 40 years stocks were remarkably stable, there were repeated concerns that they were overfished (discussed below).

One reason often cited for the general increase in landings from the 1930s into the 1990s is increased fishing pressure or effort (e.g., Fogarty, 1995). Fishing effort on lobsters, expressed as the number of traps fished per year, shows a strong increase since World War II especially during the 1970s, after which (until very recently) it has largely stabilized (Figure 8.2A; see Thomas 1980 for a more complete discussion of effort). Many argue that this is only part of the story and that effective effort has continuously increased due to longer soak time, improved trapping, hauling and navigating capabilities (Anon., 1996a).

Traditionally landings increase with increasing effort until the maximum sustainable yield is

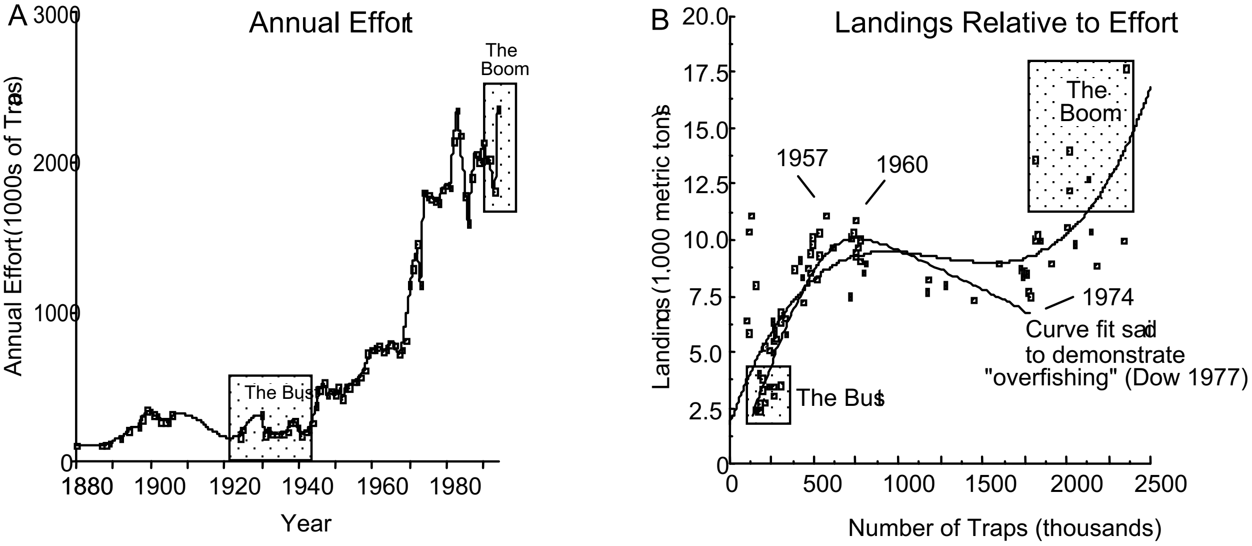

Figure 8.2. Lobster landings in Maine and fishing effort (i.e., number of traps fished) since 1880. A. Temporal trends in fishing effort, B. Landings relative to fishing effort. Effort is approximated by number of traps fished. Two best-fit polynomial curves yielded different results. Dow (1977) calculated a curve in 1974 showing a distinct decrease in landings with effort after landings highs recorded in 1957 and 1960. A reanalysis with data through 1994 shows a distinct increase during the boom of the 1990s. Data from Maine's Department of Marine Resources.

achieved. After that point, landings will decline with continued increases in effort. It was believed that the fishing effort on lobsters reached in the late 1950s and early 1960s attained the maximum sustainable yield (Dow, 1977; Figure 8.2B). The dramatic increase in effort in the 1970s (Figure 8.2A) was coincident with a decline in landings (Figure 8.1). By 1974 the trend of declining catch with increasing effort was interpreted as clear evidence of overfishing (see curve labeled "1974" in Figure 8.2B; Dow, 1977). In subsequent years, however, the declining trend reversed to the record levels of the recent boom. This resilience despite enormous effort has surprised managers and has contributed to a lack of confidence in fisheries science held by some in the industry.

FLUCTUATIONS IN LOBSTER STOCK: OVERFISHING AND/OR THE ENVIRONMENT?

EVIDENCE FOR OVERFISHING: ARE STOCKS REPRODUCTIVELY LIMITED? IS BROODSTOCK DECLINING?

> *The causes of the decline of the fishery are plainly evident. More lobsters have been taken from the sea than nature has been able to replace by the slow process of reproduction and growth.*
>
> *-Herrick, 1909*

Lobster stocks have long been assumed to be overfished. As indicated in Herrick's (1909) quotation above, the principal overfishing concern relates to the reproductive capacity of the stocks. Stocks that are reproductively limited have insufficient fertilized eggs to maintain population densities and are thus “recruitment overfished”. This is the major biological and management concern (Anon., 1996a). Economic concerns such as improving yield (e.g., “growth overfishing”) are usually considered a separate matter especially if the broodstock and reproductive potential remain strong. It is possible that “growth overfishing” or harvesting lobsters before they can provide the maximum sustainable yield per recruit could also have ecological consequences to the stock (e.g., by shifting the size structure of the population toward smaller individuals) but to date, there are no indications that this affects reproduction or sustainability of the resource.

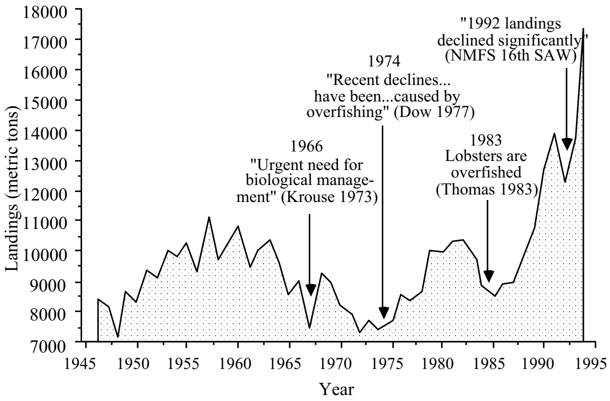

Figure 8.3. Overfishing and management concerns voiced by federal and Maine state fisheries managers.

Since 1945 most warnings of overfishing have been coincident with periods of declines in landings (Figure 8.3; Miller, 1994). As Anthony and Caddy (1980) stated, “Landing declines are often interpreted to be stock collapse due to overfishing”. However in every case, lobster stocks rebounded without a reduction in fishing effort. The fact that lobster population densities throughout their range have increased significantly in recent years indicates that they are not currently reproductively limited and thus not literally recruitment overfished. This point was amplified by Pezzack (1992) who pointed out:

> *A low level of broodstock (recruit overfishing) was a popular explanation for the depressed catch rates during the 1970s...; however the very large year classes which occurred in the 1980s, some of which were produced by these low population levels, weakens this argument.*

A decline in the reproductive potential of lobster stocks can only result from a decline in the abundance of broodstock lobsters. The reproductive potential is the broodstock sufficient to maintain the population assuming that adequate conditions for larval survival, growth and settlement prevail. To date, there is no direct indication that broodstock is declining in abundance. For example, there is no significant trend in abundance among lobsters at or above harvestable size in the National Marine Fisheries Service's groundfish trawl surveys (Figure 8.4.; Anon., 1993a) over the 12 year period from 1980 to 1992. These surveys provide the only direct estimate of abundance used for population data on which overfishing determinations are based (discussed below).

Despite the long history of suspecting stocks were overfished, there is little hard evidence to support that thesis. As pointed out by Elner and Campbell (1991), “...the events with lobsters in the

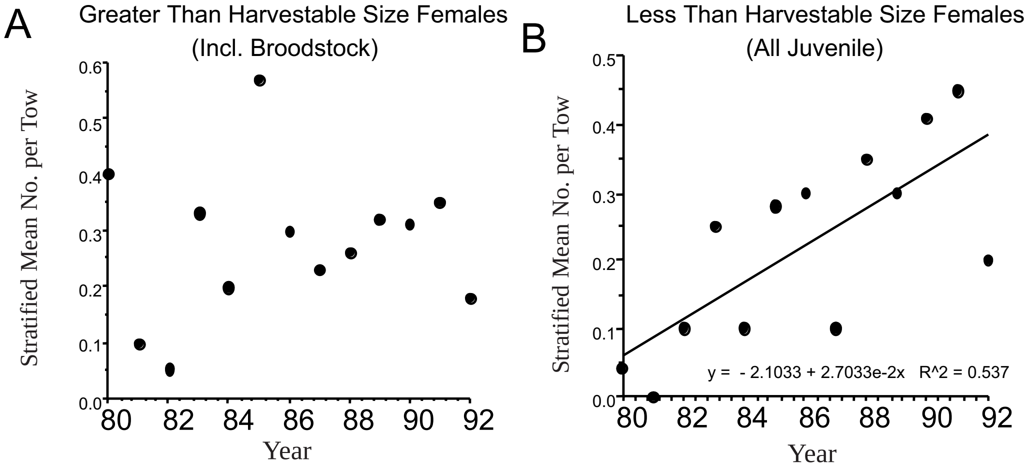

Figure 8.4. Trawl survey data on lobster abundance (Anon., 1993a). A) Fully recruited lobsters (i.e., > 83 mm CL) include all potential broodstock. There is no temporal trend in the abundanceof this component of the population. B) Prerecruits (< 83 mm CL) all of which are juveniles. There is a significant increase in this component of the population. Data from NMFS Northeast Fisheries Science Center Autumn trawl survey (Anon., 1993a). Recently revised data through 1995 continue the above trend (Anon., 1996b).

northwestern Atlantic over the past 10 years suggest [that] recruitment can be independent of fishing pressures....” In a similar vein, Miller (1994) questioned the efficacy of management by pointing out that, “The very large area over which landings increased argues against favorable management regimes or changes in regimes as causes. Seasons, minimum sizes, fishing effort, etc., vary a great deal over the area considered ... and changes in management during the 1970's and 1980's were small.” Could the demographic signal from the environment be stronger than that from fishing pressure or management measures? Given the undoubtedly great effort exerted by the fishery, the environmental control would have to be extraordinarily large.

ENVIRONMENTAL CONTROL: A CASE FOR WATER TEMPERATURE ON EARLY LIFE HISTORY PHASES

Temperature has long been considered a key environmental variable for lobsters (Huntsman, 1924; McLeese and Wilder, 1958; Flowers and Saila, 1971; Dow, 1977; Aiken and Waddy, 1986; Fogarty, 1988; Campbell et al., 1991). Since some temporal trends in both lobster landings and sea surface temperatures correspond broadly throughout the western North Atlantic (Elner and Campbell, 1991), temperature is a likely candidate. However, there is no consensus on how temperature affects landings.

It is also important to know which phase of the lobster life history is likely to be most impacted by temperature. Many species including lobsters have a “critical period” (*sensu* Hjort, 1914; Frank and Leggett, 1994) or “critical phase” (*sensu* Langton et al., 1996) in their life history. The critical phase is the period in the life history of an organism when cohort size and ultimately population size is determined (Langton et al., 1996). If temperature controls the success of a critical phase, then it will control the abundance of the species.

Three ways temperature may control landings are by regulating trapability, growth rates, and/or settlement success (Figure 8.5; Fogarty, 1988, 1995; Campbell et al., 1991; Addison and Fogarty, 1992; but see Aiken and Waddy, 1986 for other temperature effects). Because these temperature effects occur at different times in the life cycle of the lobster, by analyzing landings relative to the years since the thermal event, it should be possible to evaluate the importance of temperature and to identify the phase in its life cycle where the impact is greatest. For example, factors affecting larval settlement may have an immediate impact on population density but will only be evident at the time of recruitment to the fishery after they reach minimum harvestable size (83 mm carapace length, CL) about seven years after settlement. In contrast, changes in growth may be evident a year or two after a thermal event whereas trapability will be evident the year of the temperature record (i.e., no time lag, McLeese and Wilder, 1958). The particular season or month of thermal influence may also be important. For example, temperature effects on larval biology and behavior are confined to the summer months when they are in the water column.

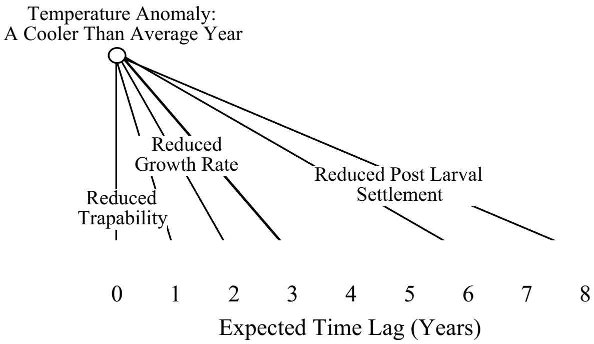

Figure 8.5. Expected time lags after a sea surface temperature anomaly (e.g., a cooler than average year).

To determine the life cycle phase when the temperature effects on landings are greatest, a progressive time-lagged regression analysis was performed (landings and Boothbay Harbor, ME sea surface temperature data from Maine's Department of Marine Resources). Regressions were run both with average and with August sea surface temperatures for the year of the reported catch (i.e., 0 year lag) and for each year prior to the landings for 20 years (i.e., a 20 year lag).

Sea water temperatures around the time of post-larval settlement have a significant impact on stock size and future landings. The best fit between landings from 1946 to 1986 and sea surface temperature was determined by the proportion of variance explained (i.e., the r^2 value) for each time-lagged regression (Figure 8.6). The strongest significant relationship between landings and average

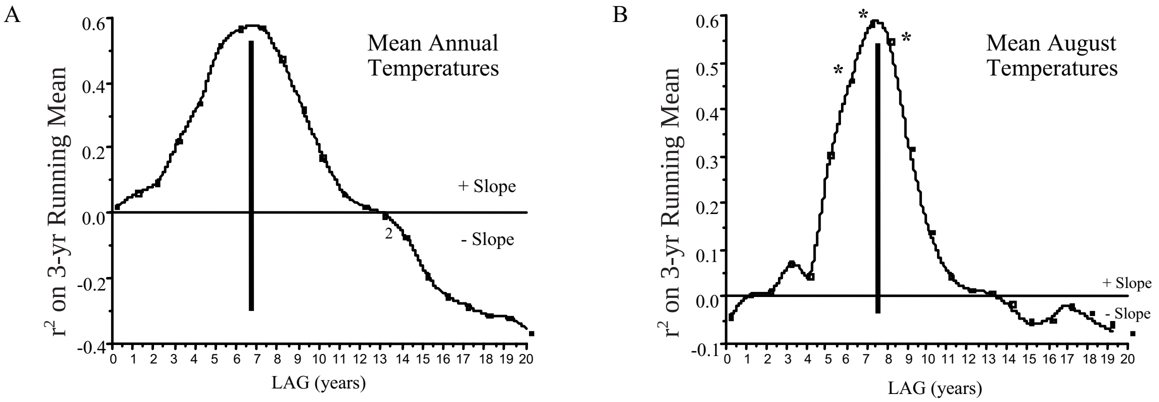

Figure 8.6. The proportion of variance explained by temperature with time lags ranging from 0 to 20 years. Variance is represented as r^2 based on regression analyses of mean annual temperature (A) and mean August temperatures (B). Asterisks represent significant inclusions to a multiple regression model at the 0.05 level. So that landings reflect stock size, rather than interannual variability, I used a three year running mean on landings (method of Ennis, 1986). This is necessary because at the time of harvest a warmer than average year may result in an extra molt and an early recruitment into the fishery and a cooler than average year may result in a delayed molt and a later than average recruitment into the fishery. Such noise is of little long-term consequence to the fishery. The progressive improving of the proportion of variance explained toward and away from the 6 - 8 year lag is probably in part the result of variance in cohort growth rates.

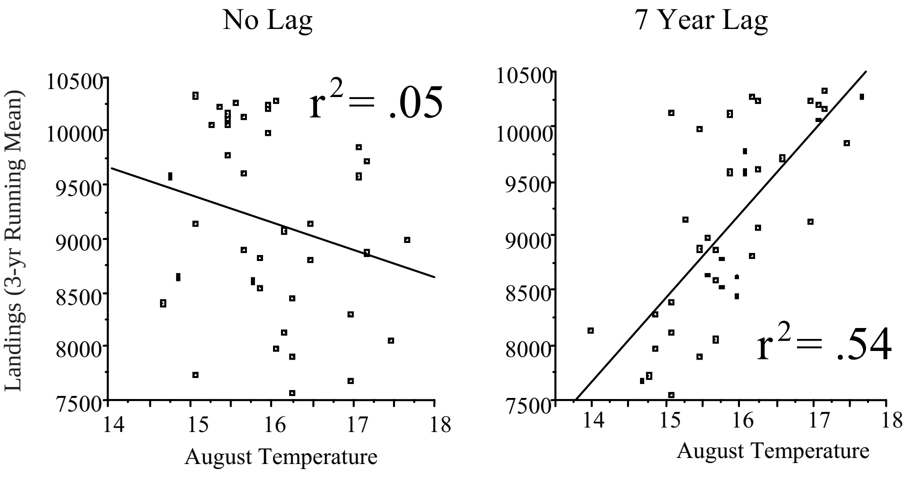

Figure 8.7. Annual landings and mean August temperatures from 1946 - 1986 for the year of the harvest (A, no lag), and seven years after the recorded temperatures (B, 7 y lag).

temperatures was found around a mode of six to seven years (Figure 8.6A). When August temperatures were analyzed in this way (Figure 8.6B), a similar but more pronounced spike occurred at the period between 6 and 8 years. Both analyses explain approximately the same proportion of variance in landings and both show the strongest signal is around 7 years later which is consistent with the idea that thermal control of population density is at or around the time of post-larval (PL) settlement.

Since settlement occurs primarily during August in Maine, thermal patterns then may be the single best environmental determinant of future landings. The wider curve resulting from average temperatures may reflect other more behavioral (i.e., trapability in year one, Addison and Fogarty, 1992) and growth-related impacts in subsequent years, but these are minor relative to the very strong signal evident at seven years or around the time of PL settlement.

To better visually interpret this analysis, Figure 8.7A shows the regression between landings and August temperature in the same year (i.e., no lag). The nonsignificant ($p = 0.12$) slope is negative and r^2 is only 0.05. In contrast, the relationship between landings and temperatures seven years prior shows a strong positive and significant ($p < 0.001$) relationship with an r^2 of 0.535 (Figure 8.7B). (Note that the r^2 values from each of these regressions is in Figure 8.6B).

I focused on environmental influences for the 40 years between 1946 and 1986 (Figures 8.6 and 8.7) since declines during that period, were interpreted by fisheries managers to have been the result of overfishing (Figure 8.3). Although temperature may have contributed to some fluctuations, this analysis does not indicate that stocks were not threatened over that period or that they are not threatened today by overfishing. However, patterns of decline were evidently more attributable to environmental factors than to fisheries-related impacts over that period. Fogarty (1995) showed similar results over this period using a transfer function technique. Since these fluctuations in landings have no apparent relationship with the size of the broodstock they are not evidence of overfishing.

Different analyses of the recent population increase have reached different conclusions. Whereas Fogarty (1995) found the boom of the 1990s (Figure 8.1) was attributable to temperature, my analysis did not (e.g., Figure 8.6; Acheson and Steneck, 1997). Several theories have been advanced to explain this change but beyond the thermal explanation (e.g., Fogarty, 1995), most relate to changes in the biotic component of the ecosystem. Prime examples are that effective nursery grounds or juvenile habitats have expanded due to recent increases in kelp and decreases in predator abundance in the coastal ecosystem (see Habitat and Ecosystem Considerations below).

The point is, it is essential for those interested in detecting recruitment overfishing to be able to filter out environmental "noise" from the fisheries signal. Clearly, many of the declines in lobster landings in Maine that were interpreted to indicate overfishing (Figure 8.3), in fact turned out to best correspond with time-lagged temperatures and the processes they affect at the time of larval settlement. Other environmental variables influencing lobster larval success have been suggested in other systems (e.g., river discharge in the Gulf of St. Lawrence; Sutcliffe, 1973, lagged 9 yrs). There are no studies I know of that indicate a shortage of broodstock, i.e., recruitment overfishing over the last half century.

LOBSTER MANAGEMENT: THE EGG PER RECRUIT DEFINITION OF OVERFISHING

THE EGG-PRODUCTION-PER-RECRUIT DEFINITION: SET AT A PRECAUTIONARY LEVEL

Federal law requires "an objective and measurable definition of overfishing for each managed stock or stock complex with an analysis of how the definition was determined and how it relates to the biological potential," (Anon., 1989). A recent report of the Stock Assessment Workshop (Anon., 1993a) concluded that the lobster fishery, as a whole (including Gulf of Maine), is overfished by the current definition of overfishing. The applicable definition of overfishing published in Amendment 5 to the Federal American Lobster Fisheries Management Plan (1994) is:

> *The resource is recruitment overfished when, throughout its range, the fishing mortality rate (F), given the regulations in place at that time under the suite of regional management measures, results in a reduction in estimated egg production per recruit to 10 percent or less of a non-fished population [F10%].*

This definition was adopted as a precautionary measure. That is, it is set at a level that should not allow reproductive or stock collapse since such an event would be a disaster to the industry for decades at least (Anon., 1996a). The conundrum is that only by experiencing stock collapse can the estimated egg production per recruit be calibrated or the efficacy of the overfishing definition be demonstrated. This definition and the egg-per-recruit (EPR) approach is the guiding light for fisheries management in all lobster producing states and recently for Canada (Anon., 1995).

Assumptions and Concerns of the Eggs per Recruit Overfishing Definition

The current EPR definition is based on the idea that fisheries models and statistics are sufficient to estimate the proportion of the lobster population that is harvested or dies each year and sets at a precautionary level the proportion of the population that must survive and reproduce to sustain the stocks. There are two primary concerns related to this definition: 1) it is based on many fundamental estimates or assumptions that are either untested or untestable, and 2) it assumes that the principal threats facing the resource relate to its reproductive health as estimated by the production of eggs per recruit.

There are six key assumptions necessary to determine overfishing:

1) The stock-recruitment relationship is known;
2) Stocks can be commensurably quantified throughout their range;
3) Mortality (both natural and fishing or F) can be estimated;
4) There is no large-scale geographic segregation of the population and net loss from a management zone due to migration is negligible;
5) The necessary proportion of ovigerous lobsters relative to an unfished population to sustain the stock is known (i.e., is it 2%, 5% 10% or 20%?);
6) Ecosystem change is inconsequential or does not affect the fundamental eggs per recruit relationship necessary to sustain the population.

Arguably the question should be, do we know enough to manage lobster stocks this way? Is there sufficient confidence in this approach to have it be the sole basis for management? There are disturbing voices from fisheries scientists who suggest the answer may be "no." Below I outline some of the serious questions that have been raised over each of the six assumptions above.

1) The Stock-recruitment Relationship Is Known. Knowing or estimating the relationship between the abundance of parent stock (i.e., broodstock) and the resulting yield of recruits to the fishery is central to fisheries management (Frank and Leggett, 1994). Pezzack (1992) reiterated this assumption by stating that the "stock recruitment relationship is the basis of much of ... lobster management." However he went on to point out a troubling problem-to date, "no clear stock recruitment relationship has been found in lobsters."

The estimated stock-recruitment relationship on which lobster stocks are managed in the US and Canada was published by Fogarty and Idoine (1986; 1988). Since there are no estimates of the size of parent stock populations, these investigators used data provided by Scarrett (1964, 1973) on larval abundance to represent broodstock and subsequent (time-lagged) landings for the

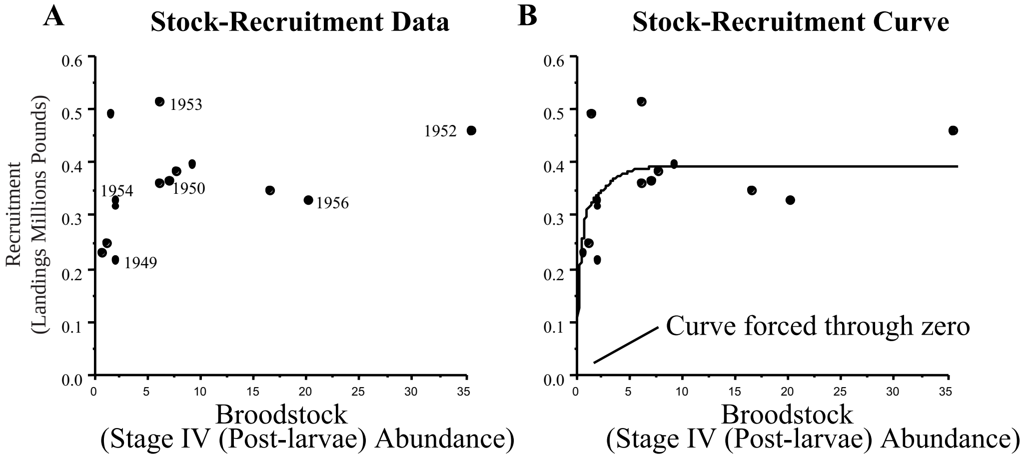

Figure 8.8. The stock-recruitment relationship on which lobsters stocks in the western North Atlantic are managed (Anon., 1995; 1996a,b). Broodstock (i.e., parental stock) and its resulting egg production is assumed to be represented by the abundance of Stage IV post-larvae. Recruitment of lobsters to the fishery is assumed to be represented by landings. A: Data from Scarrett (1973), B: Analysis and curve-fit from Fogarty and Idoine (1986).

Northumberland Strait in the Gulf of St. Lawrence in their analyses. The resulting plot (Figure 8.8A) shows no relationship between Stage IV post-larvae abundance and stock size. However, based on the assumption that no larvae will result in no landings, the published curve was forced through the origin (Figure 8.8B). This curve became the stock-recruitment curve on which lobsters throughout the western North Atlantic are now managed (Anon., 1995, 1996a,b).

A consequence of this stock-recruitment curve which steeply plunges to the origin, is that there will be virtually no warning of stock-collapse. That is, over a wide range of broodstock abundance there is no change in the harvested stock until the broodstock reaches very low abundances, at which time stocks are predicted to crash. This interpretation of this curve also suggests that landings alone will not be good indicators of the risk of stock collapse. The shape of this curve, especially its slope near the origin, is critical for the management strategy applied to lobsters (Anon., 1996a,b).

The most serious flaw in this logic is that there is no evidence that larval abundances relate to the size of broodstock. In fact, the data used for the lobster stock-recruitment curve (Figure 8.8A) shows very rapid interannual fluctuations in larval abundance. Extremely low larval abundances recorded in 1949 were followed by the highest value in 1952 which was followed by low values again in 1953 and 1954. Broodstock abundances do not and cannot fluctuate at that rate. We must conclude that larval abundance is a poor indicator of broodstock abundance and thus the curve (Figure 8.8B) is not a stock-recruitment curve. That larval abundances are variable and could result in low landings, is not the same as a broodstock-collapse resulting from recruitment overfishing.

Variability in local larval abundance can result from variation in reproductive success or environmental influences. For example, physical oceanography and meteorology affects ocean current and wind delivery patterns of competent post-larvae (Incze and Wahle, 1991). Water temperatures influence post-larval growth rates (Harding, 1992) or sounding behavior (Boudreau et al., 1991, 1993). All of these factors can account for significant differences in larval settlement without any change in broodstock abundance. Further, larval settlement may not always correspond with larval abundance. For example, if postlarvae choose not to settle, or if they settle in habitats where early post-settlement mortality is high (Wahle and Steneck, 1992), the resulting landings would be low or absent but not necessarily the result of low post-larval abundance. Given this, it may be inappropriate to interpret the post-larval -recruitment data (Figure 8.8A) as a curve forced through zero (Figure 8.8B).

The application of the stock-recruitment curve generated for Northumberland Strait was interpreted by Fogarty (1995) to show “a generally declining trend in larval production” with the evident result being “the population declined markedly shortly after cessation of ... sampling....” in 1968. This is contrary to published landings data for that region (Harding et al., 1983; Pezzack, 1992) which shows a steady decline from 1960 to mid 1970s. The same pattern was recorded in Maine over the same period (Figure 8.3) but clearly larvae came from different parent stocks. Had larval declines been the result of declines in broodstock, a much longer period for recovery would have been expected (Fogarty, 1995). However in both cases, without any significant management action, and continued increases in fishing effort, this decline was followed by a steady population increase for well over a decade.

The stock-recruitment relationship for lobsters remains elusive because it is a problem of scale. Broodstock abundance would more likely relate to landings if the population is “closed”. But it is now widely recognized that larvae come from reproductive lobsters that live elsewhere and thus represent an open system or a metapopulation (Cobb and Wahle, 1994).

The stock-recruitment relationship for lobsters is a window into a bigger question. In a recent review by Frank and Leggett (1994) they point out that stock-recruitment models in general have fallen under “increasing criticism”. They cite papers that argue that “fisheries scientists have shown excessive willingness to impose theory on data rather than testing the null hypothesis that there is no relationship between stock and recruitment.” I submit it would be difficult to reject that null hypothesis using published data for the American lobster.

2) Stocks must Be Commensurably Quantified Throughout Their Range. It is difficult to quantify

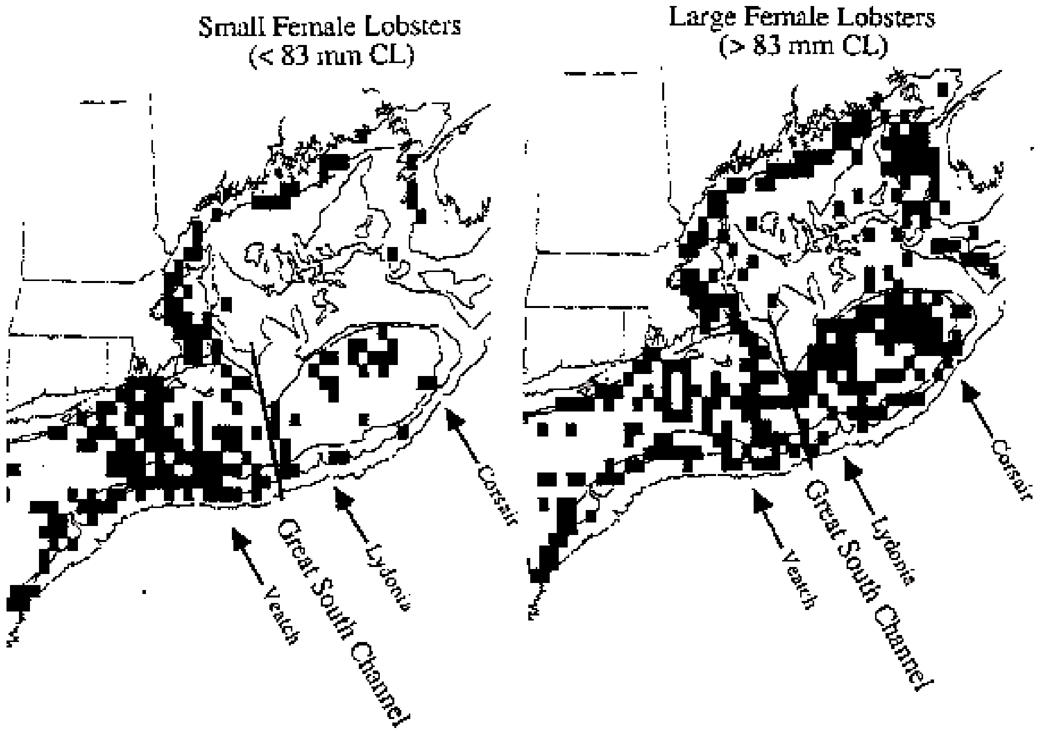

Figure 8.9. Results from randomized stratified trawl surveys conducted between 1982 and 1991 during Autumn for the Gulf of Maine and southern New England (modified from NMFS Northeast Fisheries Science Center). Black blocks had one or more lobsters, unshaded blocks had none. Note that coastal regions in the central Gulf of Maine that have the highest landings have no trawled prerecruit lobsters (i.e., < 83 mm CL) whereas sand regions of the western Gulf of Maine that have much lower landings have much higher trawled densities of prerecruits. Note that Georges Bank (east of the Great South Channel) is dominated by larger lobsters (i.e., > 83 mm CL). Offshore canyons and the Great South Channel labeled for reference.

lobster abundance in regionally commensurable ways using current techniques. For example, NMFS biannual groundfish trawl surveys randomly go to different trawl locations each year (i.e., stratified random sampling, Figure 8.9). If in one year the trawls sample some hot spots but in other years they do not, interannual variability will be great. Trawl samples contain very few lobsters per tow. Figure 8.4A shows only about one tow in three will contain a female lobster and the resulting annual abundance estimates have high interannual variability. Some of the variability relates to the substratum type being trawled. Trawl-capture efficiency will be high in sand where lobster population densities are naturally low and very low in boulder fields where lobster population densities are high. Other variability relates to regions where trawl sampling can be conducted. To avoid damage of fixed gear (lobster traps), trawl sampling targets regions where lobstering effort is low or nonexistent. For example, coastal regions in central Maine where most lobsters are harvested and prerecruit densities (in #/m^2) are greatest are not sampled (Figure 8.9). Coastal regions in sand-dominated Massachusetts where fewer lobsters are harvested and prerecruit abundances are less, are sampled. As a result, population patterns derived from trawl sampling are opposite that of landings and demographic studies (Figure 8.9). Addison and Fogarty (1992) stated that: “the only reliable measure of true changes in abundance... would be direct census estimates.” This currently does not exist.

3) Mortality (Both Natural and Fishing Mortality, or F) Can Be Estimated. Measurements of natural mortality are lacking (Thomas, 1980; Conser and Idoine, 1992). An assumed 10% natural mortality per year is usually used but this may be far from the mark especially if applied to the entire lobster population. Rates of predation are very high at the time of post-larval settlement and rapidly decreases as they grow (Wahle and Steneck, 1992; Wahle, 1992). When lobsters in coastal Maine grow to near harvestable size (greater than 60 mm CL) they are virtually immune to predators (Steneck, 1989, 1995a) because most coastal predators today are small in size (Malpass, 1992; Witman and Sebens, 1992). In contrast, on Georges Bank where large lobsters predominate (Campbell and Pezzack, 1986), annual rates of natural mortality are likely to be exceedingly low. In all likelihood for any given size there will be a significant habitat-related (e.g., sand vs. boulder) and region-related (e.g., coastal vs. offshore) difference in natural mortality. Ontogenetic differences in mortality rates are likely to vary by orders of magnitude.

The means of estimating fishing mortality for lobster is the DeLury method (Conser and Idoine, 1992; Anon., 1996a). This method assesses the relative abundance of prerecruit lobsters (assumption #2) and based on estimated rates of growth, assuming there is no migration (assumption #4) and natural mortality is 10% (assumption #3), the expected harvestable biomass of an unfished population can be approximated. The amount observed less than that expected of an unfished population is assumed to reflect fishing mortality. Most of the assumptions used to estimate mortality will be difficult to test. However, the assumption of no migration (assumption #4) creates a particular problem.

4) There is No Large-scale Geographic Segregation of the Population and Net Loss from a Management Zone Due to Migration is Negligible. This assumption was reiterated by Fogarty (1995) for the population dynamics models for lobster: “a closed population is assumed in which immigration and emigration are negligible....” It is well known that the range of lobster movement increases with body size (Krouse, 1980). Large, reproductive lobsters, may migrate hundreds of kilometers (Campbell and Stasko, 1985; Campbell, 1986, 1989) whereas young of the year lobsters may not move more than a meter (R. A. Wahle, personal communication). As a result, there are off-shore and deep water regions dominated only by large reproductive lobsters (Figure 8.9; Skud and Perkins, 1969; Campbell and Pezzack, 1986; Steneck, in prep) and shallow coastal regions dominated by juvenile lobsters (Steneck, 1989; Steneck and Wilson, 2001).

There are serious management implications if regional losses due to migration cannot be distinguished from mortality. For example, in the 16th Stock Assessment Workshop (Anon., 1993a), the stock assessment area of Southern Cape Cod-Long Island Sound Inshore had the highest estimated rate of mortality (exploitation rate of 81%). This zone also happens to be elongate, largely coastal with an abundance of prerecruit lobsters (< 83 mm CL, Figure 8.9). One would expect the greatest net movement of larger lobsters out of this zone to regions offshore. In contrast, the Georges Bank and South Offshore stock easement area had the lowest estimated rate of mortality (exploitation rate of 35%). This zone is dominated by large lobsters that apparently have migrated in from other regions (see Figure 8.9). Recognition of this problem with lobsters was voiced by a fisheries manager who stated: "...we all could be overestimating 'F' [Fishing mortality] values due to migration." (Anthony and Caddy, 1980). It was also identified as a general fisheries management problem by Frank and Leggett (1994) who point out that: “population dynamic models applied to fish populations frequently ignore dispersive processes such as immigration and emigration....” Especially since, “emigration... has commonly been viewed as the equivalent of mortality....” They go on to point out that this has been well known among ecologists (e.g., Wynne-Edwards, 1962) who “believed that dispersal acted as a safety valve providing immediate relief to the potentially negative effects of overpopulation.” Frank and Leggett (1994) go on to conclude that “recruitment dynamics may be seriously misinterpreted if such dispersive processes are ignored.”

5) The Necessary Proportion of Ovigerous Lobsters to Sustain the Stock Is Known (i.e., is it 2, 5, 10, or 20%?). If we assume the egg per recruit management approach works, what proportion of the population must be reproductive to sustain stocks? The target estimated egg production per recruit of 10 percent of an unfished population was proposed as a precautionary level which was acknowledged to be a rather rough educated guess based on estimates from fin fish and other exploited crustaceans. Since lobster stocks were determined to have suffered overall mortality rates that result in less than 10% egg production per recruit for 8 of the 10 years analyzed (Anon., 1993a), yet lobster populations increased over this time, this value cannot indicate the limit of the stock's reproductive potential. According to NMFS scientists (Anon., 1993b):

> *The fishing mortality rate which would result in a recruitment failure is not known for lobster populations in the United States. It is true that there is uncertainty about whether the 10% level is 'correct'. However, the only way to know for sure is to reduce the population to the point where it collapses and to observe the level of egg production where the collapse occurred. For obvious reasons, we do not want to see this happen....*

The certainty expressed in “the only way to know for sure....” may be unwarranted. If environmental conditions can impact lobster stocks by causing several years of failure in larval settlement, then stocks could collapse but not due to insufficient broodstock or a reduced reproductive potential. As shown above, stock declines or even a short-term collapse does not necessarily indicate recruitment overfishing, *sensu strictu*. Thus the question remains, how should this biological reference point be determined and tested? Uncertainty regarding the 10% reference point was reflected in the Canadian decision to apply the egg per recruit approach but recommended a 5% biological reference point (Anon., 1995).

6) Ecosystem Change Is Inconsequential or Does Not Affect the Fundamental Eggs per Recruit Relationship Necessary to Sustain the Population. The final assumption relates to changes in the ecosystem. The egg per recruit approach simplifies many of the complex problems described above into a few simple elements. In essence it assumes that the population dynamics of the resource depends primarily on the broodstock. While in the extreme case of reproductive collapse this is undeniably correct, as the singular guiding principle for recruitment overfishing, it may be an oversimplification. This approach tacitly assumes that other changes in natural mortality or larval settlement success will be negligible. There are reasons to believe that these and other important components of the ecosystem have changed over the past century. Obviously some abiotic changes such as water temperature described above may change ecosystem function, but there are also some relatively recent biotic changes that may be important to lobster stock abundance.

Predator-prey interactions are undoubtedly different since groundfish were depleted from coastal locations. Whereas cod and other groundfish were abundant in coastal Maine during the 1920s (Rich, 1930), they have been depleted from coastal zones since the 1950s (Witman and Sebens, 1992; Steneck, 1995a, 1997; Conkling and Ames, 1996). Thus natural mortality has probably declined over the past several decades as fewer and smaller size classes of lobsters remained vulnerable to predators.

It is also possible that nursery grounds or sites for successful settlement have increased recently. Harvesting of the green sea urchin, *Strongylocentrotus droebachiensis*, over the past decade depleted populations of this major macroalgal herbivore from coastal zones. As a result kelp beds have expanded throughout the Gulf of Maine (Steneck et al., 1995) and this may provide more sites for lobster settlement. Wahle and Steneck (1991) found that besides cobble rock some newly settled lobsters can be found in kelp as a nursery habitat. It was also suggested that a relatively small change in predation rates early in life may significantly change the population size (R. A. Wahle, personal communication).

Kelp and other macroalgae provide shelter for larger lobsters and can significantly change the local carrying capacity of some habitats, such as relatively featureless ledge or bedrock, for larger adolescent phase lobsters (> 40 mm CL; Breen and Mann, 1976; Bologna and Steneck, 1993). In effect, this could retain harvestable lobsters in coastal zones that would otherwise migrate offshore or to deeper water.

Thus, the changes to the ecosystem are likely to detract from the efficacy of the egg per recruit determination of overfishing. In other words, it is entirely possible that the proportion of reproductive lobsters necessary to maintain lobster populations are likely to vary with some of these changes in the abiotic (e.g., temperature) and biotic (e.g., predators, nursery grounds, habitable space) components of the ecosystem.

ARE LOBSTERS OVERFISHED? WHO KNOWS?

> *Considering the imprecision associated with stock definition, the extent to which factors other than spawning stock size seem to cause variability in recruitment, and the lack of understanding of larval recruitment processes, determination of a stock-recruitment relationship for the American lobster is unlikely in the near future.* *-Ennis, 1986*

Because there may be serious problems with the stock-recruitment relationship and the egg per recruit approach for lobster management does not mean stocks are not nearly overfished - they may be. The central concern should be, if they are, how will we know? Because of the nature of the current best estimate of the egg per recruit relationship for lobsters (Fogarty and Idoine, 1986, 1988), the precautionary level for the allowable F is applied so that a surplus supply of eggs is always available. However fisheries scientists have openly questioned whether this is the best goal for management. For example, Elner and Campbell (1991) wonder if environmental controls make “recruitment... independent of fishing pressure...”. It follows then that “traditional fisheries models based on concepts such as surplus production and stable recruitment would be largely redundant.” Further, it could be argued that simplifying the biological concerns for lobsters to only a recruitment overfishing argument distracts attention from other stock-threatening activities or events which could have impacts as great as reproductive collapse.

Concerns about recruitment overfishing are really concerns about the reproductive potential necessary to sustain current lobster stocks. A critical gap in our knowledge of what sustains the stocks is not knowing the location or size of their effective broodstock. Effective broodstock are reproductive lobsters that contribute to landed lobster stocks. Gravid females that release their larvae into ocean currents that take them away from nursery grounds will not be contributing their offspring to the fishery and thus are not part of the stock's effective broodstock.

Different regions with different oceanographic characteristics will have different effective broodstocks. Therefore, not only will the effective broodstock for Long Island Sound be different from that of the Gulf of Maine, but also sizable populations of reproductive lobsters within the Gulf of Maine may have equally little impact to sustain their local stocks. As an additional management goal, it would be very useful to locate, monitor and protect the effective broodstock. Because reproductive female lobsters produce large larvae after a long period of parental care, the per-egg survival rates are likely to be much greater than they are for most marine organisms. Also, since lobsters are long-lived (maximum age may be as high as 100 years, Cooper and Uzmann, 1980) and have a long reproductive life, larval supply to nursery grounds may remain high even after several years of settlement failure. All of this suggests that a more “surgical approach” to fisheries management is possible for this species than is or has traditionally been used (Steneck, 1996). If effective broodstock persists in deepwater refugia, then steps should be taken to protect that component of the population. For example, it would be prudent and more risk-averse to prohibit the harvest of oversized and v-notch lobsters in other state and federal waters. Recent interest in metapopulation models for managing the American lobster (Anon., 1996a) tacitly recognizes the significance of self-segregating broodstock persisting in a refuge from highly vulnerable juvenile stocks. Until there is consensus on the best approach for conserving broodstock, multiple independent approaches should be less risky.

CONCERNS OTHER THAN OVERFISHING

If lobster stocks crashed due to factors other than broodstock abundance or egg production, the economic impact would be just as severe. For this reason, other key factors should be considered in assessing the health of lobster stocks. Two important issues that relate more to the lobster's environment than to its reproductive health, are habitat degradation and pollution.

HABITAT: A DEMOGRAPHIC BOTTLENECK IN THE EARLY BENTHIC PHASE?

If lobster population densities are regulated by settlement success as has been shown for reef fish (Doherty and Fowler, 1994), barnacles (Connell, 1985; Gaines and Roughgarden, 1985) and for benthic assemblages in general (Underwood and Fairweather, 1989), then factors contributing to successful settlement may well play a larger role in their demographic success than will broodstock size *per se*. In any location, successful settlement requires (1) available competent larvae (which requires sufficient larval production and oceanographic dispersal; see Underwood and Fairweather, 1989), (2) the propensity to settle (e.g., sounding behavior, Boudreau et al., 1991; and appropriate tactile, visual or chemical cues, Scheltema, 1974) and (3) available nursery grounds (Wahle and Steneck, 1991).

Successful settlement requires each of the conditions be met. For example, a demonstration of oceanographic control on lobster larval availability is evident in the larval shadow created by the lee side of islands where settlement is significantly reduced (Incze and Wahle, 1991). The propensity of lobster post-larvae to settle may be controlled by water temperature (Boudreau et al., 1991 and discussed above). Assuming those first two conditions are met, available nursery grounds may control recruitment of lobsters to the benthos (Steneck, 1989; Cobb and Wahle, 1994) and thereby control the ecosystem's carrying capacity.

Newly settled lobsters have very specific habitat requirements for small shelter-providing habitats such as peat reefs or cobble beds (Able et al., 1988; Cobb and Wahle, 1994). Experiments have shown that settling lobsters suffer extraordinarily high rates of predation outside of refugia (Wahle and Steneck, 1992). The median time to the first attack from small, commercially-unimportant predatory finfish is 15 minutes (Wahle and Steneck, 1992; Boudreau et al., 1993). These fish predators (primarily juvenile cunner, sculpins and shannys) are ubiquitous in shallow coastal zones where average densities of nearly one per meter square have been recorded (Malpass, 1992). Coastal settlement of lobsters is primarily within the upper 20 meters (Figure 8.10).

Since lobster settlement is largely confined to shallow (Figure 8.10) cobble nursery grounds (Wahle and Steneck, 1991), this habitat is an “essential habitat” (*sensu* Langton et al., 1996) or a demographic bottleneck for this species (represented as the constriction in Figure 8.11). Furthermore, because early benthic phase lobsters are concentrated in cobble bottoms for the first several years of their life and this habitat comprises no more than 2 to 10% of coastal substrates (Kelley, 1987), this habitat is particularly at risk and should be a high priority for protection (Steneck, 1995b).

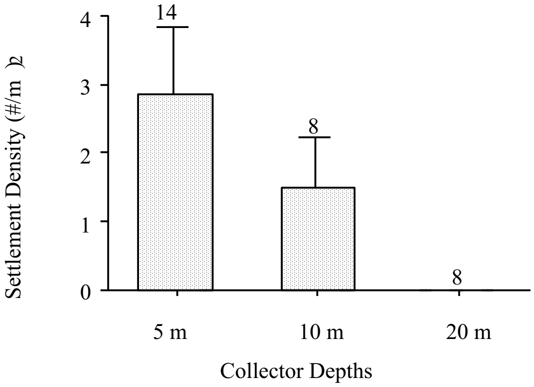

Figure 8.10. Lobster settlement as a function of depth in coastal Maine. Data from artificial lobster post-larval collectors placed 1 July and retrieved 15 September 1995. Number of m^2 collectors is represented above each bar. Error bar indicates one standard deviation.

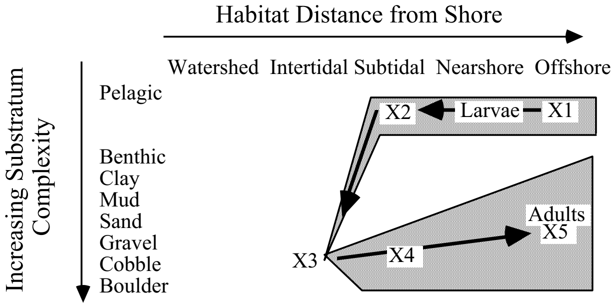

Figure 8.11. A habitat-life history matrix for the American lobster (Langton et al., 1996). Ontogenetic phases (X1, youngest to X5, adult) relative to the distance from shore, or water depth and substrate complexity. The point X3 represents early benthic stages which are critical phases in the life history of this species (most habitat-restricted). Shallow cobble bottoms are an essential habitat because only there are settling lobsters safe from predators.

FISHING IMPACTS ON HABITAT

Lobster nursery grounds and preferred habitats are vulnerable to some fishing and other human activities. The primary risks are from sedimentation (i.e., dredged materials) and dragging, both of which reduce spatial complexity (Auster et al., 1996) and from pollution (Harding, 1992). For example, the increased effort in sea urchin harvesting has recently accelerated dragging activity in some coastal zones and adds to the growing list of other species harvested that way such as scallops and mussels. Recent studies by Canada's Department of Fisheries and Oceans assessing the impacts of dragging for sea urchins in Passamaquoddy Bay, reported the following (Robinson et al., 1995):

> *Visually, the effects of the drags on the habitat were the disruption of the bottom substrate as many boulders have been turned over and dislodged from the sediment.... There was...some loss of macroalgae due to the dragging. Dragging also had an impact on the lobster populations at the Minister's Island site as the density of lobsters in the experimental plot decreased to zero over the course of the dragging while the control plot remained constant.*

Although Robinson et al. (1995) only looked for large, relatively mobile, lobsters (which they conclude may have evacuated the area), the smaller early benthic stage lobsters, if present, would be unable to exit the drag area because of the added risk of encountering predators (Wahle, 1992), such as sculpins, which often increase in abundance as a result of dragging.

POLLUTION CONCERNS: DEMOGRAPHIC IMPACT?

Pollution is often a source of concern for all marine organisms. This is particularly true for organisms such as juvenile lobsters that live in shallow and heavily populated (including industrial) regions. In major reviews of the responses of lobsters to contaminant exposures (Harding, 1992; Mercaldo-Allen and Kuropat, 1994) numerous accounts were given of detectable levels of various pollutants and, when known, lethal limits. It is beyond the scope of this paper to review this subject in detail. However very little data exist on how most pollutants impact natural lobster populations. Most described pollution effects are relatively local. While lobsters readily accumulate detectable levels numerous heavy metals, polycyclic aromatic hydrocarbons, pesticides and other anthropogenic compounds, there is little evidence that these impact the population dynamics of lobsters. Often concentrations in nature are well below those identified as having a lethal impact, however, oil spills are a notable exception.

As with other contaminants, spills of oil and other petroleum products can be highly variable in their impact. Crude oil contains hydrocarbons and metals. Mortality impact is greater for larval and juvenile stages than it is for adults in general. Exposure to no. 2 fuel oil at <0.15 mg/L for 5 d can make lobsters unresponsive to food. Higher exposure (1.5 mg/L) causes gross neuromuscular responses with a loss of coordination and equilibrium. Demographic impacts are variable because weather (wind, sea), temperature and the nature of the petroleum product control exposure and physiological effects. For example, in 1970, Bunker C fuel oil was spilled in Chedabuco Bay, Nova Scotia and in 1979 a similar incident occurred in Cabot Strait, but in neither case was there a measurable impact on mortality or harvest. In contrast, about 825,000 gallons of No. 2 fuel oil leaked into coastal waters of Rhode Island in January 1996. This was coincident with turbulent weather and resulted in significant lobster mortality. “Preliminary estimates suggest that ...a million lobsters were stranded” (Cobb and Clancy, 1996). Adjacent coves exposed to the same storm swell but no fuel oil had no washed up lobsters (Stan Cobb, personal communication). Studies are continuing but it is felt that highly turbulent conditions and cold weather (poor evaporation) conspired to mix sufficient fuel oil downward into the water column to have had a toxic impact on the local population. While this has been locally devastating, it is unknown how widespread the affects will be.

CONCLUSIONS AND SOME MANAGEMENT IMPLICATIONS

Despite numerous predictions that lobster stocks were recruitment overfished and on the verge of reproductive collapse, landings have remained remarkably constant and in the past decade significantly increased. Although repeated and recent reviews by fisheries scientists affirmed past determinations of overfishing, there are reasons voiced by other fisheries scientists to question some of those conclusions. Fundamental components of fisheries models employed for lobsters have been insufficiently tested and perhaps are untestable. There are published and logical arguments against accepting stock-recruitment curves, estimates of total mortality and assumptions of ecosystem stability. Until the abundance of the effective broodstock for harvested stocks of lobsters is known, the stock-recruitment relationship cannot be estimated. Furthermore, fisheries scientists have been unable to sort environmental noise from fisheries-induced signals. As a result, the primary reliance on specific estimates of egg production per recruit relative to estimated unfished populations requires a level of resolution that to date may be unattainable. If this is the case, then we must conclude that we simply do not know if lobster stocks are recruitment overfished. Thus the concern about risk of “commercial extinction” voiced nearly 90 years ago by Herrick (1909) remains, but scientific evidence in support of that concern is still lacking.

I raise these concerns with the hope that a more prudent course of action will be initiated which includes additional new multiple independent estimates of the health of lobster stocks. Specifically, the distribution, abundance and location of the effective broodstock should be determined, monitored and if possible protected. The same should be done for lobster nursery grounds. To that end, regional and temporal patterns in lobster settlement should be determined and monitored. This diverse approach for determining overfishing on lobsters uses appropriate spatial scales, considers differences in lobster ontogeny, associated changes in habitat requirements (e.g., segregated life history phases), and it should filter out environmental noise so that real threats to the reproductive capacity of the stocks can be identified and acted on more effectively. Understanding environmentally-induced changes in stock size is important so that industry and managers alike do not mistake short-term declines for fisheries-induced reproductive collapse (recruitment overfishing). Such knowledge would also improve scientists’ ability to predict natural changes in stock size which, if demonstrably correct, should improve the credibility of the scientific process in the eyes of industry. The ultimate goal for lobster managers is not just to answer the question, “are we overfishing the American Lobster?”, but to convince industry to take action when it is clear that they need to do so. In the meantime, a risk-averse strategy of protecting effective broodstock and nursery grounds would be a logical course of action. It requires protecting essential habitats for critical life-history phases (*sensu* Langton et al., 1996) but if the appropriate spatial scales are selected, this action could be done surgically (Steneck, 1996).

ACKNOWLEDGMENTS

Ideas and perspectives presented here resulted from years of mostly productive discussions and debates with numerous colleagues. Jim Acheson, Bill Adler, Robin Alden, Ted Ames, Peter Auster, Herman (Junior) Backman, Bob Bayer, Stan Cobb, Dick Cooper, Dave Cousens, Dave Dow, Bruce Estrella, Mike Fogarty, Arnie Gamage (and the South Bristol Fishermen's Coop), Joe Idoine, Lew Incze, Jay Krouse, Peter Lawton, Doug McNaught, Jack Merrill, Alvaro Palma, Judith Pederson, Doug Pezzack, Peter Sale, Rick Wahle, Bob Wall, Pat White, Carl Wilson and Jim Wilson. Drafts were read by Peter Auster, Robert Buchsbaum, Mike Fogarty, Bill Robinson, Rick Wahle, Jim Wilson and Steff Zimsen. Some strongly disagree with some of my conclusions, others agree, and to all I am grateful. Whereas some good ideas no doubt came from some of those listed above, all errors are entirely attributable to me. I am indebted to NOAA's National Undersea Research Program's National Research Center at the Univ. of Connecticut at Avery Point (Grant No. NA46RU0146 and UCAZP 94-121) and the University of Maine Sea Grant College Program for funding research cited herein.

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Chapter IX

The Role of Overfishing, Pollution, and Habitat Degradation on Marine Fish and Shellfish Populations of New England: Summary and Conclusions

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> *Suddenly the idea flashed through my head that there was a unity in this complication—that the relation of one resource to another was not the end of the story. Here were no longer a lot of different, independent, and often antagonistic questions, each on its own separate little island, as we had been in the habit of thinking. In place of them, here was one single question with many parts. Seen in this new light, all these separate questions fitted into and made up the one great central problem of the use of the earth for the good of man.*
>
> *-Gifford Pinchot, 1947*

The purpose of this chapter is to synthesize the information provided in the earlier chapters on the impacts of overfishing, pollution, and habitat degradation on certain groups of fish and shellfish populations in the Northeast and to consider what the implication are for the future of the marine ecosystem. These three anthropogenic impacts have affected groundfish, anadromous fish, inshore bivalves and lobster differently. The chapter by Murawski makes a convincing case for overfishing as the major factor responsible for the recent decline in New England groundfish species. The relative importance of each of the three factors is less obvious with the other groups of marine organisms. Habitat degradation, particularly dams that block access to spawning areas, have had a major impact on populations of anadromous fish, but pollution and overfishing have also influenced a number of these species. The chapters by Brousseau and Steneck suggest that populations of lobster and nearshore bivalves are largely determined by natural variability in yearly recruitment of juveniles, at least in the areas they studied. This variability may mask any anthropogenic influences on these two groups, although the absence of reliable population data, particularly with nearshore bivalves, makes predictions difficult.

DEMERSAL SPECIES (GROUNDFISH)

THE ROLE OF OVERFISHING AS A CONTROL ON GROUNDFISH

It would be hard to dispute the notion that overfishing has been the major factor leading to the current decline in groundfish in the Northeast. As Murawski describes in Chapter 2, populations of groundfish most important to the commercial fishing industry are presently at low population numbers compared to historic levels. The cause of these population declines has been extremely high rates of fishing mortality on most groundfish stocks throughout the 1980s and up to the mid-1990s. During this period, not only did gear become more efficient at catching fish but also there was a rapid increase in the number of people entering the fishery, spurred by government programs.

About one-third of groundfish species managed by the New England Fisheries Management Council are currently classified as overfished based on rate of harvest and long term overfishing definitions (NMFS, 2001). Groundfish stocks have been characterized by both growth and recruitment overfishing (i.e. declines in yield attributable to harvesting smaller and smaller fish and declines in recruitment of juveniles due to low spawning stocks). Catch per unit effort, another indicator of the status of fish stocks in relation to fishing effort, steadily has declined for groundfish despite the improvements in fishing technology.

The influence of fishing on New England commercial fish is not just evident from the past two decades. Murawski points out that fish populations, such as cod, haddock, and Atlantic herring, have historically reflected the intensity of fishing effort. This has been particularly pronounced since the increased industrialization of fisheries in the early part of the 20th Century.

There have been some recent improvements in some fish stocks, such as Georges Bank yellowtail and haddock, in response to restrictions on fishing effort and area closures, implemented with increasing severity since the early 1990s. With reduced fishing mortality, older, larger fish are surviving longer, leading to anticipated improved spawning success and less dependence of the fishery on new recruits. Overall biomass numbers, however, are still too low for groundfish to support increases in fishing effort at this time. Exploitation rates of Gulf of Maine cod and whiting are still above rebuilding target.

One piece of evidence that relates the decline in groundfish to overfishing is the observation that the recent decline in groundfish has been limited to those commercial species that have been heavily exploited by commercial fishers. Species that are not being targeted or that are subject to strict management measures have not been depleted or have recovered. As an example, fishing effort on two pelagic species, Atlantic herring and mackerel, has been low since foreign vessels left New England waters after the passage of the Magnuson Fisheries Conservation and Management Act of 1976. The biomass of these two species has increased markedly in the past 30 years, and they are both listed as underexploited and at high abundance by NMFS (1998). Striped bass, whose allowable catch by both commercial and recreational fishers was drastically reduced in the 1980s as a management response to low populations, are now considered recovered and are touted as a fisheries management success story. The recovery of these species has been a direct response to lower exploitation rates.

Have there been Population Effects on Groundfish from Toxic Pollutants?

The population effects of toxic pollution are less clear than overfishing. With the exception of oil spills, pollution rarely causes direct mortality, but rather makes fish more vulnerable to other sources of mortality.

The Gulf of Maine contains a number of the most contaminated sites in United States coastal waters for polycyclic aromatic hydrocarbons (PAHs), chlorinated hydrocarbons (e.g. chlorinated pesticides and PCBs), and several trace metals. The effects of pollutants at the cellular, physiological, and whole organism level in fish from some of these contaminated harbors can be quite striking. Thurberg and Gould (Chapter 4) cite numerous examples of physiological alterations on gadids and flounders caused by heavy metals and organic contaminants. For fish exposed to pollutants, the survivorship of eggs and juveniles is lower than that of adults, although physiological impacts are observed at all life stages. There is very little information on the population effects of these pollutants even where effects on reproductive physiology have been noted.

As difficult as it is to relate toxic pollution to populations of fish in the most contaminated sites in the Northeast, it is even more challenging to understand what the subtle effects, if any, are of long term exposure to low levels of contaminants. This latter situation is more relevant to the overall region since the concentrations of contaminants in all but a few urban harbors within Massachusetts Bays and the Gulf of Maine are below the level where acute effects are possible. Even pelagic species that may travel far offshore, such as tuna and swordfish, are exposed to some level of land-based pollutants, as evident from the recent Environmental Protection Agency/Food and Drug Administration fish advisory on mercury contamination in seafood (http://www.epa.gov/mercury/advisories.htm) (USEPA, 2004).

Studies of impacts of toxicants in New England have been carried out on winter flounder, an inhabitant of some polluted harbors. The goal of much of this research has been to examine site-specific effects on the fish or to explore public health risks. Other than direct toxicity from oil spills, the effects of even high levels of contaminants on the populations of flounder and other marine organisms have proven very difficult to isolate from other variables that influence reproductive behavior and success in the environment. As an example, polluted harbors are also organically enriched, thereby providing a greater amount of food to winter flounder, either directly or indirectly through augmented prey populations. This could lead to faster growth rates of flounder even in the presence of toxicants.

Based on physiological information, pollutants have the potential to impair reproduction and therefore reduce recruitment. The extent to which this actually happens in the field is not known. A number of studies cited by Thurberg and Gould relate high pollutant concentrations to lowered reproductive success. These include winter flounder exposed to PCBs in Long Island Sound and Atlantic cod exposed to oil spills in the North Sea. Thurberg and Gould state that such effects in the field would vary "erratically with time and site." If a pollution "signal" is occurring, one would predict that the decline in fish populations would be more pronounced in those species or populations that occur nearshore, at the higher end of any pollution gradient emanating from land-based sources. Although historically nearshore species were likely overharvested first before fishing effort moved further offshore, the present decline in fisheries has occurred both nearshore and offshore. We conclude that the major cause of low recruitment has been low initial spawning biomass related to overfishing.

EFFECTS OF HABITAT LOSS AND DEGRADATION ON GROUNDFISH

Destruction and degradation of large sections of coastal and nearshore habitats have occurred throughout the New England coast since the arrival of European colonists. Even if habitat effects have not played a major role in the recent New England groundfish crisis, anthropogenic impacts on habitats could slow or inhibit the recovery of groundfish when restrictive fishing measures are implemented. Deegan and Buchsbaum (Chapter 5) reviewed impacts to finfish caused by losses of coastal wetlands, hydrological alterations, dams, eutrophication, damage from fishing gear, ditching for mosquito control, power plants, and exotic species. Like pollution, most habitat impacts on fish are indirect, in that they do not cause direct mortality themselves, but make the fish more susceptible to other sources of mortality, such as increased predation on juveniles due to loss of hiding places. Such indirect impacts are therefore hard to quantify.

There have been a number of difficulties in relating habitat changes to changes in fish populations. First, we do not know under what conditions fish populations are limited by the availability of suitable habitats, even for those species for which habitat preferences are established. As an example, we do not know whether the loss of 30 to 50 percent of the precolonial salt marsh acreage has reduced populations of estuarine-dependent fish, since they may be more affected by other factors. On the other hand, recent studies of cod and hake suggest that there may be some critical habitats at particular life stages that are limiting (Auster and Langton, 1999; Lindholm et al., 1999). This was the rationale for the New England Fisheries Management Council's 1998 designation of a cobble habitat in Georges Bank as a "Habitat Area of Particular Concern" for juvenile cod. A second difficulty is that habitat types that are important to fish are not necessarily obvious to us. It may be relatively easy to characterize the fish community and boundaries of a cobble area or an eelgrass bed, but the location of an offshore salinity discontinuity that may be important to fish larvae changes depending on the relative flow of rivers and tidal currents. A third problem is that it is very difficult to characterize all the potential habitat interactions that may affect a species. Habitat impacts on predators, prey, and competitors may all influence the population. Fourth and closely related to (3) is a general lack of knowledge of fish-habitat relationships. Up until recently, the major focus of fisheries managers and fisheries researchers from federal agencies has been on the population biology of individual species, not on ecological relationships.

Before 1990, most of the well-documented studies of habitat losses and degradation were of coastal wetlands and shallow nearshore habitats because that was where the most obvious physical changes had occurred. In addition, the logistics for carrying out studies, primarily by university researchers who do not have ready access to offshore fisheries research vessels, were easiest. In those parts of the United States and in Australia where fisheries are heavily dependent on estuarine and nearshore species, declines in commercial fisheries have been directly linked to loss of coastal wetlands. Research by Deegan and her coworkers showed that eutrophication of coastal embayments results in a measurable change in the fish communities within eelgrass beds (Deegan and Buchsbaum, Chapter 5).

Recent research on the impacts of fishing gear on benthic communities in fishing grounds further offshore has raised serious concerns about habitat changes that may at a minimum be affecting recruitment of certain species and under a worse case scenario, altering the integrity of the entire marine ecosystem (see Dorsey and Pederson, 1998; Auster and Langton, 1999; Watling and Norse 1999; NRC 2002 for reviews). Changes in the physical structure of benthic communities as a result of the activities of draggers have been documented in both nearshore and offshore waters of New England. Dragging disturbs physical and biogenic habitat features that are attractive to various species of juvenile fish. Lindholm et al. (1999) and Olney and Boehlert (1988) suggest that loss of habitat structure, such as that which occurs during bottom dragging or dredging of seagrass beds, increases predation on juvenile fish.

Three modeling efforts cited by Deegan and Buchsbaum suggest that habitat degradation does have an impact on some fisheries. In the Northwest Shelf region of Australia, dynamic models indicated that the abundance of some commercially important fish species were limited by the amount of suitable habitat provided by epibenthic animals that are prone to removal by bottom dragging. Boreman et al. (1993) concluded that increasing juvenile survival of inshore winter flounder in the northeast United States through habitat restoration in combination with reduced fishing pressure on adults results in a greater overall benefit to the population than reducing fishing effort alone. Based on a simulation model, Schaaf et al. (1993) predict that destroying only 1% of the estuarine habitat of juvenile menhaden could result in a 58% decline in population levels after 10 years.

THE ROLE OF NATURAL ENVIRONMENTAL FLUCTUATIONS ON GROUNDFISH

The suggestion that variations in natural environmental factors has had a severe impact on New England groundfish species was the basis for the application by Commonwealth of Massachusetts for federal disaster relief for its commercial fishing industry in 1995. Recruitment does vary from year to year based on climatic and other ecological conditions, however Murawski (Chapter 2) shows that poor recruitment is not the cause of recent groundfish declines. His simulation model suggests that poor recruitment may have had the effect of exacerbating declines caused by overfishing, but that overfishing was clearly the major driving force. For a sustainably-managed fishery, exploitation rates should account for the potential for poor recruitment in any given year.

Recruitment of yellowtail flounder has been related to seawater temperatures, based on a decline in recruitment and landings during a warming period in the 1940s and 50s and a subsequent increase when temperatures cooled. The last major period of consistent change in seawater temperature, however, was a warming in the early 1960s. Since that time seawater temperature has shown yearly variations but no consistent trend upward or downward that would likely affect recruitment of yellowtail flounder or any other species in a consistent way (Murawski, 1993). Changes in the amount of runoff from the Saint Lawrence River have been discounted as a cause of decline in cod (Frank et al., 1994). Even if there were clear-cut environmental trends that might impact the recruitment or migratory patterns of groundfish, one would expect that the changes would be observed in a variety of species and not just coincidentally in those species that happen to be heavily fished.

Fisheries managers in their projections of New England groundfish populations have generally relied on the assumption of a constant level of instantaneous natural mortality (M), typically set at M=0.2. Habitat quality has never been factored into these models, perhaps because the high level of fishing mortality in recent years has made variations in natural mortality a minor factor in predicting populations. With the decline in fishing mortality rates in the late 1990s and early 2000s under strict regulations, changes in natural mortality and mortality associated with habitat degradation will likely become a more significant factor in population trends.

MULTIPLE STRESSORS AND RECOVERY

The biological community of the Gulf of Maine ecosystem has changed in a number of ways because of overfishing of groundfish (Witman and Sebens, 1992). Some species may experience rapid population growth once the strong influence of a limiting factor (i.e., overfishing in this case) is removed. This is particularly the case if, as suggested by Sinclair (1997), overfishing has not changed the basic structure of the biological community. On the other hand, the reduction of many populations to their present low levels may have changed the community dynamics such that certain species may no longer be able to achieve their former abundance, at least in the short term.

The populations of different marine species in New England are likely never in a state of equilibrium in relation to each other. Natural changes in fish communities occur in response to long term and yearly climatic trends or as different species influence each other through competition and predation. Species that are prey for cod, for example, may become more abundant because of overfishing of groundfish and then exert a controlling influence on future cod numbers by feeding on juveniles.

One cannot assume that there is some predetermined level that a population trajectory will reach once a major source of mortality is removed, particularly for an ecosystem that is subject to natural environmental variations over different time scales that are superimposed on human impacts. Myers et al. (1995) suggest that fish stocks in general can recover if the overfishing problem is addressed, however there is no indication that the population of Atlantic cod off Newfoundland has returned despite many years of a fishing moratorium. The anticipated recovery of New England groundfish due to reduced fishing effort and closures of large areas provide an opportunity to better understand the effects of ecological factors that may regulate groundfish populations.

ANADROMOUS FISH

Moring (Chapter 3) makes it clear that overfishing, pollution, and habitat degradation have all reduced populations of anadromous fish from their former levels of abundance in precolonial times. Anadromous species were declining in southern New England as early as 1870, primarily due to dams and pollution, two products of the Industrial Revolution. Today, habitat degradation and pollution still affect population trends.

Blockage of migration routes by dams and other structures across rivers and streams has eliminated access to large areas of potential spawning habitat. By 1950, damming of rivers had left less than 2% of the original habitat for Atlantic salmon in New England accessible to the fish. A recent survey of 215 coastal streams in southeastern Massachusetts documented 380 obstructions to fish passage, the majority of which are “manmade” dams (Rebeck et al., 2004). Many rivers now have fishways around dams, but these still are not as efficient in allowing fish to successfully migrate both up and downstream as are free flowing rivers. There have been efforts to remove dams that are no longer serving a useful function, such as the Edwards Dam on the Kennebec River in Augusta, Maine.

Although dams have been the most serious factor in declining anadromous fish runs, other habitat factors have also been of concern. These include increased water temperatures and siltation of spawning areas due to the removal of streamside vegetation, siltation caused by sanding of roads in winter, and algal growth on spawning sites due to eutrophication.

Striped bass provide an example of an anadromous fish that in the past suffered from the effects of both overfishing and pollution. Overfishing in the 1970s and early 1980s led to severe population declines. The fish have now recovered well after a period of severe restrictions on both commercial and recreational fishing, so overfishing was clearly a major factor in the decline. Moring also cites pollution reduction activities in the Chesapeake Bay region, the major spawning area along the east coast, as contributing to the recovery of the stock.

Recent problems with other species of anadromous fish have been more difficult to characterize than those of striped bass. American shad and blueback herring runs in Massachusetts increased until 1993 but have been declining since then for reasons that are not understood. Despite intensive efforts at restoration, Atlantic salmon runs to larger New England rivers are still very tenuous, and the Gulf of Maine population segment is now federally listed as endangered. Moring suggests that some as yet undetermined factor occurring when these fish are at sea may be the primary cause for the recent trends in these species. Declines in rainbow smelt runs throughout much of Massachusetts have been linked to site-specific habitat degradation (e.g. siltation, nutrient enrichment) in individual spawning streams. A modeling study cited by Moring predicted that smelt can also be severely impacted by recreational angling.

Since the decline in anadromous fish has been the result of a variety of factors, some of which are still mysterious, their recovery will require a multifaceted approach. Groundfish recovery is complicated because of politics, less so due to their biology. The assumption is that groundfish will recover if overfishing is stopped. In contrast, anadromous fish present both political and biological challenges. Recovery programs must include controlling overfishing and mitigating land-based habitat alterations and pollution, but these still do not guarantee success due to ecological interactions that are not well understood but likely beyond human control.

Bivalve Shellfish

Difficulty of Stock Assessments of Bivalves

An evaluation of the relative importance of overfishing, pollution, and habitat loss and degradation to inshore bivalve populations is clouded by the limited data available. Brousseau (Chapter 6) indicates that scientists cannot accurately assess the status of the three major inshore bivalve species harvested in Massachusetts (hard-shell clams, soft-shell clams and bay scallops) nor can they say whether these species are being overfished or not. There is a lack of reliable population data and only limited quantitative understanding about the natural and biological factors that influence recruitment of juveniles. Landings data for bivalves, although notoriously unreliable, suggest that there has been an overall decline in landings of hard-shell clams over the last twenty years. Total bay scallop landings have shown a great deal of year-to-year variability with no overall trends except for some losses in specific areas. There has been little overall change in landings of soft-shell clams.

Landings data for bivalves are suspect because they are collected by individual towns with no consistent methodology or quality control. In addition, the abundance of the shellfish resource is only one of a number of factors that determine how much is landed. If the local economy is depressed or if shellfish prices are high, more people may turn to shellfishing to earn extra income, leading to an increase in landings. Declining water quality, which reduces the acreage of shellfish beds open to harvesting, may depress landings without influencing the size of the population. Landings per unit effort, therefore, provides a better barometer of how the stocks are doing over time.

There are other difficulties in trying to understand the status of inshore shellfish resources and how to manage them wisely. Traditional fisheries models based on finfish population dynamics do not work well for these bivalves because of the difficulty in defining what a stock is and thereby establishing a stock-recruitment relationship. There may be little relationship between the size of a local shellfish population and subsequent recruitment in the locality since the planktonic larvae may come from a larger functional population that encompasses a much larger region. Thus fishing on a small, local subpopulation may have little influence on the future population size in that particular area. If this is true then shellfish resources are probably better managed at a regional level than town by town.

It has been difficult to incorporate into models the tremendous yearly variation in recruitment that characterizes these bivalves. Variable hydrodynamic and climatic conditions likely have a major effect on the success of settling. Benthic predators may strongly affect the early survival of juveniles. Sensitivity analysis described by Brousseau indicates that population growth rates of a number of commercially important shellfish are more sensitive to changes in larval survival and recruitment than they are to adult survivorship or fecundity.

ARE SOFT-SHELL CLAMS OVERFISHED IN MASSACHUSETTS?

Based on four Massachusetts towns that harvest soft-shell clams, *Mya arenaria*, almost exclusively, Brousseau showed that landings per unit effort fluctuated intensely from 1970-1995 without any consistent trends in either direction. There was also much scatter but no trends when landings per unit effort were plotted as a function of effort. Thus these particular data, admittedly limited, do not support the notion that soft-shell clams are being overfished, at least to the point where recruitment is being affected.

EFFECTS OF POLLUTANTS ON BIVALVES

In Chapter 7 McDowell describes a range of physiological effects exhibited by bivalve mollusks living or transplanted into areas heavily contaminated with organic contaminants and heavy metals. Moore et al. (1994), for example, found that the prevalence of a wide range of pathologies of *Mya arenaria* and *Mytilus edulis* (blue mussel) was strongly correlated with high levels of PCB contamination. Although direct population effects have not been documented in the New England region, a number of the physiological responses of some bivalves to lipophilic compounds, such as PAHs have implications for reproductive success. These include impairment of feeding, slower overall growth rates (which reduce reproductive output), developmental abnormalities, and degeneration of reproductive tissues. McDowell's research indicated that *Mytilus edulis* transplanted into highly PCB and PAH-contaminated New Bedford Harbor showed reduced reproductive effort and degeneration of oocytes compared to mussels transplanted into less contaminated areas.

The interactions between population growth and contaminants are complicated by other environmental influences as well as human harvesting patterns. In a study of the impact of PAH concentrations on populations of *Mya arenaria* along a pollutant gradient in Massachusetts Bay, McDowell and Shea (1997) found that clams from the most contaminated sites differed in the timing of gamete development and had high levels of gonadal inflamation and hematopoietic neoplasia, however population growth rates as estimated from a deterministic model were not directly related to contaminant concentrations. Predator and hydrological variations had a strong influence on recruitment patterns regardless of contaminant levels.

Recruitment of larvae into a contaminated area from a clean outside area may provide a periodic source of new individuals. As described for groundfish, individuals settling in a contaminated area may grow more rapidly than those in a clean area due to organic enrichment, but they may ultimately end up with impaired ability to reproduce.

Clam flats are closed in most urbanized coastal communities not because of toxicants but because of high levels of fecal coliform bacteria. Populations of soft shell clams may be quite abundant in these areas despite the fecal contamination. Fecal coliform contamination is a human health rather than an ecological concern, unless, of course, it co-occurs with heavy metals or toxic organic compounds. Such closed clam flats could serve as a source of new recruits to uncontaminated areas.

EFFECTS OF HABITAT LOSSES AND DEGRADATION

One of the best examples of the impact of habitat loss on a commercially important marine animal is the relationship between eelgrass and bay scallops described by Deegan and Buchsbaum (Chapter 5). The wasting disease epidemic of the 1930s, which wiped out most of the eelgrass along the east coast of the United States, resulted in an almost immediate crash in bay scallop landings (documented for Chesapeake Bay), which lasted until the eelgrass began to recover. Eelgrass fluctuations still occur naturally and due to eutrophication, and these still impact local populations of bay scallops.

There is little information on the impact of habitat losses on other species of bivalves. The historical filling of tidal flats in places like Boston's Back Bay undoubtedly caused losses of suitable habitat for soft-shell clams. Such widespread filling is now limited by wetlands protection regulations, however small scale losses from legal dredging, dock and pier construction, and illegal activities still are likely to occur in the region.

THE ROLE OF MULTIPLE FACTORS

Dramatic fluctuations in adult bivalve populations are probably natural in the northeast, and these may mask any affect of overfishing, pollution, or recent habitat changes. Both Brousseau and McDowell suggest that populations of bivalves are more sensitive to changes in larval survival and recruitment than to variations in adult survival, thus anything that reduces the growth and survivorship of bivalve eggs and larvae could have serious population consequences. The timing of a habitat alteration, whether human induced (e.g., siltation, dragging, remobilization of toxicants) or natural (e.g., drought, storms, annual temperature fluctuations, etc.), is probably critical during the period when larvae are in the water and probably lead to the yearly fluctuations in recruitment.

It is questionable whether human-induced changes in habitat have currently occurred on a wide enough spatial scale to affect recent recruitment in any way, except locally. What is needed to better manage inshore bivalves is to understand factors that affect larval recruitment, to establish the appropriate geographical boundaries of stocks and to collect more reliable population data. This information will enable us to understand better the impact of fishing, pollution, and physical changes in habitats on these inshore bivalves.

LOBSTERS

ARE LOBSTERS OVERFISHED?

Steneck (Chapter 8) describes a debate about the status of American lobsters, the most valuable fishery in New England from an economic perspective. Although the lobster fishery is not currently in as bad a condition as groundfish, lobsters are still classified as overexploited by NMFS due to high fishing mortality (NMFS, 2001). NMFS bases this on an extremely high rate of fishing mortality and the heavy dependency of the fishery on new recruits. They define the recruitment overfishing level for lobsters as the fishing mortality rate that results in a reduction of the production of eggs per recruit to 10% of that of an unfished population.

Steneck presents data from Maine showing that the total tonnage of lobsters landed increased from the mid 1980s through the 1990s with no evidence that the brood stock declined. Despite increased fishing effort on this species and the decline in the average size of individuals landed, the annual landings per effort ratio increased in recent years. Thus he disputes whether recruitment overfishing is occurring now.

In his analysis of data from Maine, Steneck relates periods of lower lobster abundance to lower water temperatures that reduces the success of post larval settlement. Given the extremely high rate of fishing for lobsters, it is surprising that such an environmental signal is detectable. In an analysis of a larger data set, Drinkwater et al. (1996) did not find the same specific relationship between higher seawater temperatures and the increased catch of lobsters from Newfoundland to the Mid-Atlantic Bight during the 1980s and early 1990s. Nonetheless, these authors still propose that a real increase in lobster abundance during this period was related to some as yet undetected environmental control.

Steneck also questions the accuracy of population estimates and the stock recruitment relationship used to conclude that lobsters are overfished. He argues that it is very difficult to get accurate statistics on the stock-recruitment relationship, natural mortality, and the size of the populations throughout the entire range. In addition, the models that NMFS uses in their assessments do not factor in ecosystem changes. The overfishing of lobster predators, such as groundfish, and an increase in kelp habitats attributed to the development of a fishery for sea urchins (a major kelp herbivore) have favored lobsters in recent years.

Steneck suggests threats other than fishing are equally or perhaps more important to this crustacean.

These threats include degradation of the rather limited cobble habitat required by newly settled juveniles and the negative effects of pollution. He also raises the issue of protecting a major part of the broodstock (i.e. large females that may inhabit deepwater refugia that are not fished with traps). These may be the major source of eggs and have been, up to recently, relatively free from fishing pressure.

An Alternate Perspective

In contrast to Steneck, other scientists, particularly those from the National Marine Fisheries Service, have been concerned that a fishery so concentrated on new recruits could be devastated if there were a few poor recruitment years in a row. From their perspective, it is necessary to set the overfishing definition at a precautionary level as a buffer against changes in environmental conditions that, in concert with fishing pressure, could lead to a population crash (M. Fogarty, pers. comm.). The 10% egg production level should be seen in that context rather than a threshold below which lobster populations will definitely collapse.

The trend of the fishery in recent years toward a smaller average size of lobsters and increasing dependency on new recruits is evidence for growth overfishing and is similar to what was observed in groundfish before the collapse of a number of those stocks. There may be long-term consequences to the populations of a fishery-induced truncation of age structure, at least in nearshore populations where most individuals only have the chance to spawn once before they are caught. The large lobsters that currently contribute most to the broodstock would not be limited to deeper offshore habitats if it were not for overfishing nearshore.

There are also economic consequences of the current fishing pressure on lobster. The overall yield of lobsters is not high as it could be if lobsters had a chance to grow to larger average sizes under lower fishing rates. NMFS believes that higher long term yields and a healthier lobster population would result from a reduction in the amount of fishing effort on lobsters.

Lobsters vs. Groundfish

It is interesting to speculate whether the argument Steneck presents about the limited ability of lobster statistics to accurately allow a definition of overfishing can also be applied to groundfish. Do we accurately know the stock-recruitment relationship and do we have an accurate measurement of stock sizes and accurate estimates of natural and fishing mortality? The data presented by Murawski (Chapter 2) indicate that for many groundfish species, we have a good idea of the size of the spawning stock biomass necessary to produce an adequate number of potential new recruits to the fishery. The NMFS trawl surveys undoubtedly portray the populations and the age structures of the various groundfish species with a much greater degree of confidence than is now possible for lobsters. As a consistent, repeated survey, the trawl surveys do provide an index of lobster abundance, however the catch per tow is very low on lobsters leading to much higher statistical variability than one would expect for groundfish. The problem is that trawls cannot sample in those habitats where the greatest densities of lobsters are likely to be found, i.e., nearshore rocky areas where fixed gear is in place and catch efficiencies are low.

One important way in which lobsters differ from groundfish is in the long time lag between egg production and growth to reproductive age, about six to seven years in lobsters, but much less in most groundfish species (i.e., 2-3 years in cod and haddock, 3-4 in yellowtail). Thus, it takes a number of years before anything that influences juvenile survival of lobsters is reflected in the catch.

Pollution and Habitat Impacts on Lobsters

Pollution is not likely a major factor controlling lobster populations over the whole region, however it can be locally important and of concern to human consumers. Oil spills can have devastating, localized effects, as illustrated by the 1996 North Cape oil spill in Rhode Island (NOAA et al., 1999). For chronic pollutants, the cobble habitats preferred by juveniles tend to be areas that are reasonably well flushed and therefore relatively clean. Adults that occur in soft-bottomed urban harbors are more exposed to toxic organic compounds and heavy metals. There may be some potential for a localized effect on reproduction, but whether that causes population impacts, even within urban harbors, is unknown.

The habitat issue of most concern to Steneck (Chapter 5) is the potential for damage to juvenile cobble habitat due to sedimentation and mobile fishing gear. If the availability of this habitat is really limiting lobster populations, then its protection should be a major management goal.

OTHER GROUPS OF FISH AND SHELLFISH

The reports in this volume focused on New England groundfish, anadromous fish, lobsters, and nearshore bivalve shellfish. Our intent was not to provide a survey of all ecologically and commercially important species in New England, but to explore the question of the relative importance of overfishing, pollution, and habitat destruction to representative groups for which there are some data on all three factors. For the sake of completeness, here is a brief look at other groups and the issues they raise.

HIGHLY MIGRATORY PELAGIC FISH

The recent steep declines in populations of a number of pelagic “highly migratory” fish—Western Atlantic bluefin, bigeye and albacore tuna, North Atlantic swordfish, and large coastal sharks—are due to intense overfishing. These are currently classified by the National Marine Fisheries Service as overfished (NMFS, 2001). Their pelagic, migratory life histories make it difficult to connect their population fluctuations with habitat or pollution-related factors. The relatively long life span of these species tends to mask impacts, if any, of natural environmental variations on populations. The prime focus of managers and scientists has been on managing fishing effort and understanding populations dynamics and demography (NMFS, 1997, 1998) without any emphasis on habitat-related factors. Less is known about the environmental factors that influence larval recruitment in these pelagic fish than in groundfish or anadromous species. At the moment, no hypotheses have been proposed that suggest that anthropogenic factors other than fishing mortality is influencing the populations of these species.

SEA SCALLOPS

Fishing pressure on sea scallops, *Placopectin magellanicus*, is intense. Some impacts of natural environmental fluctuations on the success of year classes have also been identified. Variations in the success of recruitment of different year classes have been related to differences in the “tightness” of the autumnal gyre in Georges Bank (Packer et al., 1998). Sea scallops also occur nearshore, particularly in the northern part of the Gulf of Maine, but there has been no research to indicate whether coastal habitat degradation has had any influence on nearshore populations. The rapid increase in the population densities and sizes of scallops in areas of the Gulf of Maine and Georges Bank closed to all gear types in the late 1990s due to the groundfish crisis shows that sea scallop populations have the potential to respond very rapidly when freed from fishing pressure in a protected area.

COMPETITION AND TROPHIC INTERACTIONS

Herring and mackerel, along with other smaller pelagic organisms, such as krill, are considered important components of the marine food chain since they serve as forage for larger fish, marine mammals, and marine birds. Fishing pressure on herring has been cited as the cause of the alteration of the biological community that resulted in an increase in sand lance in the 1970s (Sherman et al., 1981). The mechanism was presumably competition between the two species for food. In recent years, both herring and sand lance have co-occurred in abundance in the Gulf of Maine, leading Sinclair (1997) to conclude that sand lance abundance is independent of that of Atlantic herring. Both vary according to environmental factors rather than from food chain relationships.

The potential for increasing the commercial catch of herring and krill has raised the issue of potential trophic impacts of the large-scale removal of these species if they are targeted for increased fishing (Partington, 1996). Along with clarifying the actual status of herring populations, trophic modeling will be needed.

A related topic is how predation by marine mammals and birds affects the recovery of fish. Moring (Chapter 3) attributed the loss of seven percent of downstream migrating Atlantic salmon smolts in the Penobscot River to predation by cormorants. One modeling effort attributes much of an increase in natural mortality of juvenile cod in the Gulf of Saint Lawrence to predation by the rapidly increasing population of gray seals (Sinclair, 1997). Although the actual percent mortality due to the seals is uncertain, predation by the seals now likely exceeds that by fishing. Sinclair pointed out that the potential for predation by seals is much less in the Gulf of Maine due to the much lower abundance and diversity of seals compared to the Gulf of Saint Lawrence. In sum, although a number of studies have addressed this subject, there is no solid evidence that mammalian and avian predators in the Gulf of Maine and Georges Bank have caused the decline in any fish species or will hinder recovery.

CONCLUSIONS: OVERFISHING VS. POLLUTION VS. HABITAT DEGRADATION

> *Man had always assumed that he was more intelligent than dolphins because he had achieved so much... the wheel, New York, wars, and so on, whilst all the dolphins had ever done was muck about in the water having a good time. But conversely the dolphins believed themselves to be more intelligent than man for precisely the same reasons.*
>
> *-Douglas Adams, 1984*

1. Overfishing is by far the greatest cause of the decline in groundfish species in New England. The “signals” from pollution and other forms of habitat degradation have been impossible to detect, given the “noise” from overfishing. Managing fishing effort is the single most important key to the recovery of these groundfish stocks.
2. There have been no documented impacts of pollution on populations of fish and shellfish in New England, although reproductive impairment related to toxicants is well established, based on physiological studies. One would expect population impacts to be most obvious in heavily polluted urban harbors.
3. Studies from other regions that have simultaneously examined habitat quality and fishing mortality have shown that habitat quality can be very influential on some fish populations.
4. At low population levels, habitat effects could have a strong impact on recovery of groundfish, even if such impacts were not the initial cause of declines. The patterns of recovery will also be affected by any changes in the biological community that have occurred as a result of overfishing.
5. Habitat loss and degradation (including pollution) have been strong influences on populations of anadromous fish in New England. Overfishing has also been a significant factor for some species. Some unknown factor(s) when these fish are out at sea is apparently contributing to recent population declines and lack of recovery of some species.
6. Population fluctuations in bivalve shellfish are more strongly related to interannual variation in recruitment than to fishing pressure. We cannot presently factor out the effects of pollution, habitat degradation, and natural environmental variation on recruitment processes.
7. Despite heavy fishing pressure, lobster populations have remained high. Natural environmental factors that affect settling by larval lobsters may have a stronger impact on lobster populations than fishing mortality. There is disagreement among lobster biologists about whether lobsters should be considered overfished.
8. Ecosystem-level research is needed to understand the impacts of habitats and other ecological factors on commercially important fish and shellfish. Some research topics of special importance to the questions raised in this book:
    - habitat relationships of groundfish, particularly how habitat alterations by fishing gear impact fish populations,
    - population impacts of pollutants
    - causes of presumed mortality of anadromous fish at sea,
    - stock-recruitment relationships in bivalves and lobsters.

ONE FINAL THOUGHT - HAVE WE ACHIEVED THE GOALS SET OUT FOR THIS BOOK?

> *Conservationists are notorious for their dissensions. Superficially these seem to add up to mere confusion, but a more careful scrutiny reveals a single plane of cleavage common to many specialized fields. In each field one group (A) regards the land as soil, and its function as commodity-production; another group (B) regards the land as a biota, and its function as something broader. How much broader is admittedly in a state of doubt and confusion.*
>
> *-Aldo Leopold, 1949*

As stated repeatedly throughout this work, our major purpose has been to evaluate the relative importance of overfishing, pollution, and habitat loss and degradation on finfish and shellfish populations, focusing on the Gulf of Maine region. The degree to which we have succeeded must be judged, ultimately, by our readers.

While acknowledged early in our discussion (Chapter 1), it is clear that finding a common "currency" by which scientists who consider these issues can quantify the extent of the impacts they study relative to the other two constraints is difficult. Given the specialized nature of scientists, it is not surprising that those who study the physiological impacts of toxicants do not generally feel comfortable making statements about population impacts and vice versa. The data on the population impacts of toxicants and other types of habitat degradation are quite limited at this time, hence the reluctance on the part of those working on those subjects to speculate. Nevertheless, we have succeeded in advancing the common "currency" concept.

Although differences in scientific discipline have much to do with it, other factors make it hard to compare fishing impacts, pollution, and habitat degradation. The spatial scale of these major categories of impacts differ. Toxic effects, or at least our ability to detect them, are restricted for the most part to certain urban "hot spots" whereas fishing impacts are more widespread. The spatial scale of fish populations is likely much larger than the scale of toxic impacts. Habitat losses have been widespread, but in scattered localities, such that adequate refugia from those impacts may (or may not) exist. There are also questions about the adequacy of population data for certain groups of organisms, particularly nearshore bivalves and lobsters. If we do not have a firm grasp of population numbers, demography, the spatial scale of a stock, and stock-recruitment relationships, then it will be very difficult to identify the major constraint on that population quantitatively. Complicating the matter is that the natural and human-induced constraints themselves also vary over spatial and temporal scales. Nevertheless, we have succeeded in attempting meaningful comparisons in light of differential spatial and temporal scales.

To the extent that models were discussed that attempted to evaluate the population impacts of factors other than fishing mortality, our effort was also successful. Murawski (Chapter 2) showed the dominance of overfishing as a factor because his population models require no further inputs other than fishing mortality to explain the current low populations of many groundfish species in New England. Deegan and Buchsbaum (Chapter 5) described models indicating that habitat considerations as well as overfishing have the potential to influence at least some populations of fish. The need for more holistic modeling to resolve the relative importance of habitat and fishing pressure is obvious, particularly now that there has been such a large management interest in protecting fish habitats.

We believe that the primary value of this work is that it put the discussion of multiple stressors on fish and shellfish populations in one volume. It forced the authors and editors to try to relate these factors. Hopefully, future efforts based on more comprehensive data collection and increasingly sophisticated holistic models will provide more complete answers and will aid in the achievement of sustainable fisheries and a healthy marine ecosystem.

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Chapter X

Management Implications: Looking Ahead

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Ninety percent of the marine fish comes from the third of the oceans near land.
-Peter Weber, 1994

Now would I give a thousand furlongs of sea for an acre of barren ground.
-William Shakespeare, 1623, The Tempest

A growing body of evidence has documented the accelerating decline of the oceans' most productive fisheries, a trend that is amply chronicled in the northwestern Atlantic (NOAA, 1998; FAO, 1997; NRC, 1998; Figure 10.1). Aside from the obvious concern with landings and the societal impacts to the fishing community, the decline, if prolonged, will continue to elicit sweeping ecological consequences. Yet, over the past couple of decades, ecosystem considerations have been overshadowed by the fishing industry's perspective and

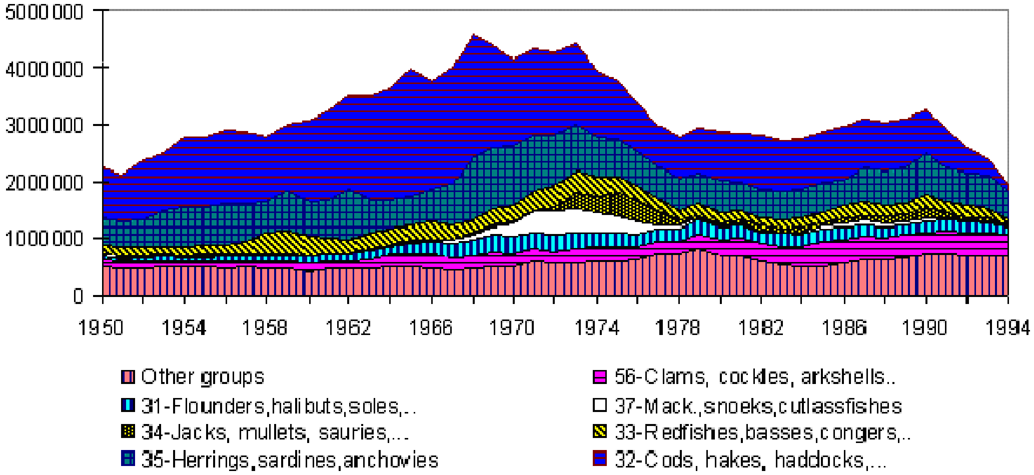

Figure 10.1. Total landings (tons) of groups of marine resources from the northwestern Atlantic Ocean over the period 1950 - 1994 (FAO, 1997). Different hatch patterns distinguish different ISSCAAP fisheries groups. Peak landings in the late 1960s was principally due to ISSCAAP Groups 32 and 35.

needs. Stock assessment models, the mainstay of commercial fisheries management, are based on the population level of biological organization—individual species of commercially important fish and shellfish—and are not based on species interactions and ecosystem dynamics (Applegate et al., 1998).

Our intent with this volume was to compare the effects of overfishing, pollution and habitat alteration on fisheries and identify possible new approaches to managing fisheries and ecosystems that integrate these factors. We have only been partially successful in broadening the scope of this debate with a "common currency" that could allow us to weight the contributions of these three impacts on fisheries' declines. We are limited because the scientific information is either lacking or too fragmented to allow us to rank the relative strengths of each contributing factor in a definitive, unambiguous way for all species. Nor is there a preponderance of models that could guide us in this endeavor. As we look forward, the main challenge to fisheries managers is the integration of ecological concepts, human activities, and social and economic considerations into sustainable fisheries management.

ECOSYSTEM APPROACH

The Magnuson Stevens Fisheries and Conservation Act of 1996 (Sustainable Fisheries Act; SFA), attempts to institute a more ecosystem-based regulatory approach to fisheries management than has been the case in the past. In practice, however, emphasis continues to be on single species, especially specific finfish, lobsters and scallops, and to a lesser extent on individual anadromous fish, squid and shallow water bivalves. This approach neglects other components of the ecosystem, upon which commercially important species often depend for food and refuge (Applegate et al., 1998).

Some ecological shifts that may have already affected marine fisheries have been reported, but most have gone undocumented. One dramatic change has been the loss of several top predators and their replacement with humans who have hunted mammals, birds, and top fish predators or have altered their habitats. Over the past two to three centuries, some fish predators such as marine mammals (e.g., harbor seals, *Phoca vitulina concolor*) and a number of sea birds (e.g., Great Auks, *Pinguinus impennis*) were hunted to very low population levels or extinction (Williams and Nowak, 1986). These large, top piscivores may have shaped species distribution and abundance. Recent population increases of other fish predators, such as striped bass (*Morone saxitilis*) and cormorants (*Phalacrocorax* spp.), may also be impacting the marine ecosystem, at least locally.

Humans, also a top predator, affect even non-targeted fish species. Barndoor skate, *Raja laevis*, caught incidentally as bycatchand discarded, were once plentiful in the northwest Atlantic, but are now thought to be on the brink of extinction due to bycatch (Casey and Myers, 1998). Both the diversity and complexity of benthic ecosystems have been markedly reduced in areas subjected to repeated bottom trawling and dredging (Dorsey and Pederson, 1998). Industrialization and coastal development have significantly increased pollution loading and altered estuarine habitats at unprecedented rates. All of these combined impacts have had and will have lasting effects on marine and coastal ecosystems, in addition to adversely affecting the fisheries of this region.

Overfishing leads to yet another set of ecological consequences. As a fish stock declines, fishermen switch to other species that are typically lower down the trophic level than the original species. In the northwest Atlantic, for example some fishermen are switching from gadoids and other groundfish to clupeids (fish that feed on plankton and are more abundant) (NEFSC, 1998). This trend, termed "fishing down the food web" (Pauly et al., 1998), has been documented in all mature fisheries worldwide over the past 45 years. Other fishermen have switched to less utilized groundfish (e.g., from gadoids to dogfish and monkfish). While the shift initially leads to increased catches, it is soon followed by the decline of the new fish stock. This pattern of exploitation is inherently unsustainable, and leads to far-reaching ecosystem changes due to the disruption in trophic interrelationships.

Our current pattern of stock exploitation has one additional ecological implication. Somehow in the discussion of fisheries' decline, a basic ecological tenet that productivity of the oceans is finite has been lost or neglected (Ryther, 1969; Holt, 1969; Russell-Hunter 1970; Mann 1982; Mann and

Lazier 1991). Primary production has narrowly circumscribed limits, that, in turn, modulate the production of fish and shellfish that are harvested for human consumption. The rise in total fish catch over the past several decades masked the overexploitation of individual fisheries. As early as 1969, however, scientists were predicting an upper limit to fish productivity and expressed concern that fisheries were declining (Holt 1969; Ryther, 1969), but the switch to new and underutilized species and the incremental contribution from mariculture tended to allay these concerns. Current estimates indicate that world marine fish production will probably peak at about 93 million tons (exclusive of a significant expansion of mariculture), only 10 million tons higher than today's landings (FAO, 1997). This maximum production can only be sustained if current overexploited and fully exploited fisheries are regulated for sustainability (World Resources Institute, 2000).

The authors of this book have identified numerous research needs that should be filled to support informed management decisions (Tables 10.1 and 10.2). Many of these recommendations have been proposed previously (MA DMF, 1985; MA MRCC, 1987; Buchsbaum et al., 1991; FAO, 1997; NRC, 1998, 1999), and would be anticipated by our readers. Most of these recommendations need no introduction or detailed explanation. Four recommendations that have not been given sufficient consideration by fisheries managers serve as a framework for achieving sustainable fisheries as presented below: (1) adoption of the precautionary approach; (2) the need for an ecosystem-based fisheries science; (3) developing new models; and (4) the adoption of adaptive management principles.

THE PRECAUTIONARY APPROACH

> *In the face of uncertainty about potentially irreversible environmental impacts, decisions concerning their use should err on the side of caution. The burden of proof should shift to those whose activities potentially damage the environment.*
>
> *-Robert Costanza et al., 1998*

Table 10.1. General categories of research needs that would assist and improve scientifically-based fisheries management. Recommended research needs and management approaches would address all four of the fisheries discussed in this volume.

Improving our understanding of ecosystem processes as they relate to fisheries productivity • Make better use of historical data in describing trends of relationships between onshore and offshore stocks and transport of pelagic larvae and life stages. • Quantify effects of contaminants on populations and relate to other impacts. • Quantify effects of habitat degradation and relate to other impacts.
Developing an ecosystem-based fisheries science, including a new suite of models that can be used to forecast changes • Integrate anthropogenic changes with natural processes in models. • Include environmental fluctuations (e.g. weather, salinity, runoff) and trends (e.g. global warming) into holistic models • Improve calibration and verification of models. • Develop models into forecasting tools for managers.
Implementing new management approaches that support sustainability of valued fisheries • Incorporate the Precautionary Approach into fisheries management • Utilize Adaptive Management techniques • Determine optimal spatial scale for management plans; eliminate management based solely on political boundaries. • Integrate natural history information into policy development and fisheries management. • Develop new electronic data analysis systems, interfacing geographic information systems, historical data sets, the databases that exist in various agencies and organizations. • Test the effectiveness of marine refugia and management closures on fisheries stocks and productivity. • Characterize and define essential fish habitats and evaluate the causes and extent of alteration. • Improve population estimates for species of interest integrating recommendations from academia, fishermen and managers.

Table 10.2. Specific research needs and management options identified by the contributors to this volume.

a. Research Recommendations	**Groundfish**	**Anadromous fish**	**Lobster**	**Bivalve**
Improve current stock assessment	X	X	X	X
Develop new integrative, holistic models	X	X	X	X
Improve understanding of population level effects of contaminants	X	X	X	X
Evaluate contaminant effects on reproduction	X	X		X
Determine the importance of endocrine disrupters on reproduction	X	X		
Evaluate effects of biotoxins on mortality				X
Identify factors at sea affecting mortality		X		
Evaluate predation pressures (e.g. bird, crab, marine mammal)	X	X		X
Develop models relating land use to populations		X		X
Identify stock and substock size to improve understanding of population dynamics	X		X	X
Develop predictive models and verify with data	X	X	X	X
b. Management Recommendations				
Reduce fishing effort to increase spawning stock biomass	X	X		
Enforce regulations, closures and management efforts	X	X	X	
Develop and implement a management plan for herring and mackerel				
Develop shellfish management plans				X
Develop management plans for restoring in-stream habitats		X		
Develop and implement management plans in conjunction with watershed groups		X		X
Create management units based on appropriate scales				X
Focus on research to evaluate broodstock trawling on lobsters			X	
Develop a plan for protecting juvenile lobster habitat			X	
Characterize the lag time between adverse or favorable effects on life history stages			X	
Determine whether predators are increasing or decreasing			X	X
Develop management options for protecting habitat	X	X	X	X

The Precautionary Principle calls for precautionary actions in response to potential threats to the environment or human health, even if causality has not been scientifically established (VanderZwaag, 1994). It is often interpreted in exclusion of the economic and social factors that managers must also consider when making their decisions. Because the language of the Precautionary Principle is rather vague and moralistic (Bewers, 1995), it has proven difficult to incorporate into policy decisions. Strict application of the Precautionary Principle would preclude any action unless it could be proven that the action would be environmentally benign. Such an extreme approach is scientifically unjustifiable, since the scientific method can never be used to prove that harm is impossible. In contrast, scientific analyses are used to assign probabilities to various actions, and to estimate the uncertainty around these probability values (e.g., a 40% probability of a 20% change). In order to include scientifically justified estimations of risks, strict adherence to the Precautionary Principle is being replaced by a more workable "precautionary approach" that includes scientific justification of estimated risk (VanderZwaag, 1994; Bewers, 1995). The precautionary approach, which implicitly recognizes that there is a diversity of ecological as well as socio-economic situations requiring different strategies, has a more acceptable "image" and is more readily applicable to fisheries management systems (FAO, 1994).

The precautionary approach calls for avoidance of serious or irreversible damage, by choosing options that have the lowest probability of long-term risk when uncertainty is high. Although decisions based on a precautionary approach are founded on scientific estimates of probability, they include economic and social factors and incentives for minimizing environmental damage.

The precautionary approach has not been broadly or enthusiastically endorsed by policy makers or managers in the United States. With respect to fisheries, U.S. agencies' policies postponed regulatory action to reduce overfishing until the evidence for stock declines proved overwhelming. This was evident in debates on the northwest Atlantic groundfish fishery. The results were disastrous. In contrast, early adoption of the precautionary approach would have incorporated the best science available. This might have included a call for lowering levels of total allowable catch for almost all species, requiring changes in gear and mesh sizes, and mandating the adoption of other alternatives to prevent overexploitation of fisheries stock (Myers and Mertz, 1998; Applegate, et al., 1998). Applying the precautionary approach might also have resulted in the shutting down of a fishery for prolonged periods, such as has occurred with cod fishing off Newfoundland, or in setting aside refuges, such as the temporary closures now in effect on Georges Bank (Lauck et al., 1998).

Since July 1998, the New England Fisheries Management Council (NEFMC) has adopted some of these more restrictive policies, but for several species the delay in taking action has only added to the length of time needed to reach maximum sustainable yield.

Another area where a precautionary approach has been touted involves the role of habitat in recovery of fisheries. Proponents of the precautionary approach want restrictions on selected gear types and protection of vulnerable habitats (such as hard bottoms in waters deeper than 30 m where recovery of benthic communities takes decades) or in areas where the effects of fishing gear on benthic communities are unknown (Witman, 1998; Collie, 1998; Auster and Langton, 1999; Watling, 1998). They recommend categorizing habitats by their vulnerability to trawling and establishing marine protected areas until data are gathered to demonstrate minimal impacts (Collie, 1998; Auster and Langton, 1999). This is not a new concept:

> *Our present information indicates that it is not fishing with the otter trawl but overfishing which is to be guarded against. ...the restriction of the use of the otter trawl to certain definite banks and grounds appears to be the most reasonable, just and feasible method of regulation which has presented itself to us.*
>
> *-Alexander et al., 1914*

Others would claim that many areas have been fished for years and that habitat alteration by fishing gear is comparable to storms and natural environmental events (Mirarchi, 1998; Pendleton, 1998). There is concern by the fishing community that no amount of data will be sufficient to permit fishing and use of all fishing gear types. The fisherman's concern represents the basic differences between the precautionary principle that restricts all use and the precautionary approach that recommends caution until evidence is collected showing there is minimal impact.

The distinction is worth repeating. Proponents of the precautionary principle would ban all trawling because it negatively impacts habitat. Proponents of the precautionary approach would restrict trawling in vulnerable areas until data were gathered to demonstrate a management approach to minimize damage.

MANAGING FISHERIES USING AN ECOSYSTEM APPROACH

Although fisheries have traditionally been managed individually, an ecosystem approach to fisheries management is receiving increasing support. However, there are many definitions of an ecosystem approach and therefore many different expectations of what its application can achieve in fisheries management. For example, the SFA promoted an ecosystem approach by requiring each management council to include both demarcation and protection of essential fish habitat in their fisheries management plans by October 1998 (Kurland, 1998). Even though the SFA continued to focus on a single species approach, the new provisions of the Act encourage research on life history, biological interactions, and the environmental variables that define habitat (physical, geochemical, and biological components). However, fish habitat is covered in the same large grid size as used by the National Marine Fisheries Service (NMFS) for fish stock assessment (approximately 700 km^2 in the northwestern Atlantic). Information on presence and/or absence and available life history data are assembled for each species managed for each grid area (NEFMC 1998). Fishers have knowledge on a smaller scale, but it is difficult to integrate their data into habitat studies (Pederson and Hall-Arber, 1999; Hall-Arber and Pederson, 1999). For many species, growth, reproduction and productivity data are lacking. Unless new resources or current research funds are reallocated, fisheries data collection and research will continue to support current studies that are not focused on habitat and the relationship of habitat to productivity. The recognition that habitat protection is critical for the development of sustainable fisheries is a major step along the path to ecosystem-based fisheries management (Langton et al., 1995, 1996; Steneck et al., 1997; Deegan and Buchsbaum, Chapter 5).

DEVELOPING NEW MODELS

The current approach to fisheries management uses various stock assessment models that input quantitative population data from fisheries independent surveys, landings data, and the scientific literature to predict future stock abundance (NEFSC, 1998). These models use previous years' data on yield, growth, recruitment and mortality to predict rates of productivity and biomass. The degree to which stocks are exploited, and the existence of growth or recruitment overfishing can be assessed from a model's output. The predictions from stock assessment models are used to recommend levels of fishing in the future. While applicable to the overfishing question, these models are not designed to incorporate impacts caused by changes in critical habitat and contaminant effects on susceptible life stages.

The National Research Council (NRC, 1998) recently reviewed five major stock assessment models and approaches used by NMFS and fishery managers nationwide. They compared the outcomes of each model using five actual or simulated datasets, covering a 30-year period. None of the models were entirely satisfactory in predicting stock abundance and most overestimated the ensuing year's biomass by more than 25%. In addition, the models exhibited a multi-year lag time in detecting trends (overestimating biomass during a simulated decline, and underestimating biomass during a simulated increase in abundance).

The current stock assessment models upon which recent New England groundfish management has been based, were criticized by NRC for not realistically accounting for natural population variability or environmental fluctuations, and for being focused on single species in a multispecies ecosystem (NRC, 1998). In addition, they ignore interspecies interactions (predator-prey, competition for space and food), and make no attempt to overlay stochastic environmental variations (seasonal variations as well as episodic events) or long-term environmental trends on their deterministic algorithms. Natural environmental fluctuations can lead to enormous changes in year class strength in some fisheries (greater in short-lived or r-selected species than long-lived or K-selected species) and are difficult to assess and incorporate into models (Sutcliffe, 1973; Hofmann and Powell, 1998).

Despite its critique of the models, the NRC report (1998) did not recommend abandoning current approaches using single species assessment models and did not really propose an alternative. It did encourage continued research in model development. In the short term, single species assessment models will probably provide the most useful data for fisheries management. However, other models, which incorporate both environmental factors and multi-species interactions should be vigorously pursued and added to the current methods of stock evaluation. These models should include the effects of contamination, fishing mortality and habitat issues, plus stochastic factors to account for temporal environmental variability (Hofmann and Powell, 1998). Since these newer models incorporate the existing single-species assessment models, continued refinement of single-species models can actually be viewed as a step along the way to more ecosystem-based models.

Attempts to integrate physical and chemical parameters with biological data are still few in number. One local example is the three-dimensional Massachusetts Bay model that has been eight years or more in development and is used in assessing the effects of an outfall in Massachusetts Bay (HydroQual, 2000). It combines a hydrodynamic model, based on physical parameters with chemical and biological data (nutrients and phytoplankton response) to forecast plankton productivity. It is considered a successful, chemical-biologically coupled model, yet it fails to identify peaks of spring and fall blooms, uses general values for predation by zooplankton and the benthos, and ignores large species, such as fish and marine mammals (HydroQual, 2000).

Another physical-biological model is one developed to hindcast the likely source of lobster larvae that settled in mid-coast Maine (Incze and Naimie, 2000). These investigators predict that larvae come from a broad section of the upstream coast (both inshore and offshore) of Maine and suggest a link between offshore reproduction and inshore recruitment. Both the MWRA (2000) and the Incze and Namie (2000) models provide information that can supplement current stock assessments and provide managers with additional and relevant information.

Research efforts, mostly from the emerging fields of ecotoxicology and environmental risk assessment, are resulting in better, more comprehensive fisheries models. Waller et al. (1971) and Wallis (1975) first proposed to include contaminant effects into fisheries-derived population models to predict population effects. Similarly, a number of studies incorporated contaminant data into simple ecological models (Daniels & Allen, 1981; Gentile et al., 1983). Summers and Rose (1987), analyzing time series data, were able to differentiate overfishing from hydrographic variability and contaminant effects in striped bass (*M. saxatilis*) and American shad (*A. sapidissima*) populations in the Potomac, Delaware and Hudson Rivers. They pointed out that few previous studies have successfully attempted to examine complex environmental variables and, of those that had, most simply correlated stock size with environmental parameters using an arbitrary time lag. Barnthouse et al. (1987) have proposed a risk-based method to apply toxicity test data to fish population models, although they warn against using their model to predict long-term population impacts. These authors recognized the similarities in such problems as identifying the cause of power plant fish kills, projecting optimal fishing effort, and determining the impact of environmental chemicals on fish populations. They were able to use this reproductive potential fisheries model, combined with chronic toxicity findings, to assess the effects of five chemicals (4 pesticides and methyl mercury) in Chesapeake Bay striped bass populations (Barnthouse et al., 1989). These toxicants not only affected survival, but also had a significant impact on fecundity. In a subsequent study, Barnthouse et al. (1990) demonstrated that contaminants had a relatively greater impact on overfished populations of striped bass and menhaden than on populations not stressed by overexploitation. Life history models have also been developed for bivalves and for assessing the sensitivity of life history stages to environmental changes (Weinberg et al., 1997; Caswell, 1996).

Thus, a number of studies have demonstrated that fisheries models can be integrated with chemical, physical and toxicological data. Incorporating these models into fisheries management decisions, at least as complementary approaches to the current single species models are a good first step. The next step in the development of holistic approaches is the incorporation of habitat impacts, e.g., predator-prey relationships, trophic-level interactions, and environmental factors into models used by fisheries managers. Development of these models is one of most pressing needs for fisheries management and should provide information for addressing those supporting a precautionary approach.

The shift from our current single-species management to an entirely ecosystem-based approach is a millennium jump. Because of the inherent complexity involved (both physical and biological), our current data analysis and modeling algorithms are not sufficiently developed to allow us to take an ecosystem-based approach at the present time. More holistic approaches, using a suite of ecosystem-based models are clearly needed, to augment stock assessment models, if not to replace them.

Models of course are only as good as the data that they use. One of the most productive uses of modeling is to point out critical data gaps and the need for additional monitoring. In addition, models can be used to set priorities for new data development, ruling out the collection of data on parameters that have little effect on the critical ecosystem relationships. The contributors to this volume have identified a variety of data gaps/research needs (Table 10.2), all of which should be incorporated into the holistic approaches and ecosystem-based models that are recommended. For example, long-term monitoring data (multi-decadal) are absolutely vital to all efforts to test and verify newly developed models. While these data are available for offshore groundfish, stock assessment data are scant for inshore bivalve populations. Contaminant effects on aquatic populations are also relatively rare, although they are currently receiving much needed attention by the field of ecotoxicology.

Regional and state fisheries management agencies are likely to resist adopting these more complicated, holistic approaches in their present state of development. Nevertheless, these are powerful analytical tools that can be used to better understand the interactions among a variety of anthropogenic impacts on fish stocks and to explore the impacts that various management options will have on these stocks. These models will continue to be refined as more data are collected and as more fisheries managers and biologists become familiar with not only the power, but the limitations of these models. Fisheries management can facilitate adoption of new approaches by supporting research into model development, testing their robustness and usefulness as applied to fisheries issues and adopting those models that are cost-effective. The opportunity exists for innovative approaches that will invigorate fisheries management.

ADAPTIVE MANAGEMENT

Adaptive management is a scientific approach that decision-makers can use for a variety of environmental problems (Costanza et al., 1998). It is a process whereby managers repeatedly modify their management decisions based on targeted collection of new data and reassessment of the situation. It is akin to scientists stating a hypothesis, and then collecting data that either refutes or supports the hypothesis. Based on the results of initial experiments, scientists may modify the initial hypothesis, and then devise additional experiments to address the new hypothesis. The scientific process is a continuous one. However, managers are usually constrained by regulations and a vague sense of how to incorporate scientific data into management decisions. Adaptive management is a particularly useful technique for managing when uncertainty is high.

The recent round of assessments, limiting days at sea, stock reassessments, and area closures in the Georges Bank groundfish fishery by the New England Fishery Management Council is an adaptive management strategy. When it was obvious that stocks could not be restored first by increasing mesh size and then by limiting the number of days at sea and catch quotas (NEFMC, 1994a), more drastic measures were implemented (NEFMC, 1994b, 1996). Areas throughout the Gulf of Maine were closed to fishing. Depending on the results of new stock assessments, additional closures and restrictions are contemplated. Recent results indicate that total biomass of cod has increased in the closed areas, but that it is too early to see an increase in either the number of juvenile cod or in recruitment (NEFSC, 1998, but see also Murawski, Chapter 2 this volume). It is predicted that rebuilding Georges Bank groundfish stocks will take many years (Murawski, Chapter 2; Myers et al., 1995). Fisheries managers revise their management decisions repeatedly, as new data are available. This adaptive approach needs to be applied to other fisheries. A suite of management options, (e.g., days at sea, total allowable catch, closed areas, mesh size, gear modifications, limited entry, quota systems), offer managers tools for employing adaptive management approaches, but not all are available to New England managers (e.g., individual take quotas or ITQs that have been successful in selected fisheries). To learn from management decisions, each option should be scientifically analyzed to evaluate its effectiveness. Socio-economic data that link management decisions to long-term sustainability and conversely over-exploitation to long-term impacts on fishing communities are scarce (Hall-Arber, Massachusetts Institute of Technology, Cambridge, MA, pers. comm.)

Adaptive management is not effective without a mechanism for gathering new data. Similarly, unpopular restrictions are difficult to enforce without first gaining the confidence of the fishing community that such restrictions are based on good information and can be modified as additional information becomes available. As an example, under the SFA, fishery management plans (FMPs) had to be modified by October 1998 to include identification and protection of essential fish habitat. Habitat data were extremely uneven and spotty. For New England groundfish, there were some relative abundance data for several life stages (eggs to adults), but there was a paucity of information on spawning adults and other detailed life history data. Management decisions still needed to be made, regardless of the detail and extent of the data on each species. When only presence/absence data were available, fishery councils, applied a precautionary approach and generally designated a larger area of habitat as essential to fish than they might have if more precise habitat information were accessible. Adaptive management provides the framework to routinely review the size of essential fish habitat, as more habitat and life history data become available.

The adaptive management approach could easily be applied to a broader spectrum of fishery issues (Table 10.2). For example, temporary closed areas are being used to help build up groundfish stocks, and the possible need for permanent refuges is being discussed. Because this is a relatively new approach, many questions are raised. Do refuges where no fishing is allowed lead to increased spawning biomass both within and outside the refuge? Is there an optimal size, number, and distance for protected areas to provide safe refuge? When are seasonal or permanent closures appropriate? How should refuges be managed? Fisheries scientists in collaboration with marine ecologists can conduct the necessary studies to fill in the data gaps. Using adaptive management, the results of this ongoing research and monitoring could be incorporated into management review and management decisions modified accordingly.

DATA NEEDS

> *One recommendation deserves to be highlighted again - the need for more current data on each commercially important species (e.g., population, recruitment, habitat) and on the various impacts on these species (e.g., predation, competition, habitat alteration, contamination, climate, seasonal weather patterns, episodic events). This need has been emphasized repeatedly over the years: All of the participating agencies agreed that the number one priority in any effort to protect/restore the environmental integrity of our coastal waters is the development and implementation of a research and monitoring program.*
> *-MA MRCC (Marine Resources Coordinating Committee), 1987*

> *The relative impact of fishing, contaminants, and estuarine habitat degradation on marine fisheries needs to be evaluated. This complex issue ultimately needs to draw on several data sources, such as the semiannual MDMF [Massachusetts Division of Marine Fisheries] stock assessment and site specific estuarine surveys…. -Buchsbaum et al., 1991*

> *The absence of adequate data is the primary factor constraining accurate stock assessment. -NRC, 1998*

Because the cry for ‘more data’ has been raised so often, some managers, legislators and the public have grown insensitive to the call and no longer acknowledge it as a vital need. This is unfortunate because there is a genuine need for specific data to answer pragmatic and practical questions raised by fisheries managers. There are several good examples of projects with well defined goals that have implemented research and monitoring programs to address issues (MWRA, 1991; MBP, 1996). The process identified for developing monitoring programs outlined in Managing Troubled Waters: The Role of Marine Environmental Monitoring (NRC, 1990) should be adapted by fisheries managers in developing observing systems that address data needs. Effective programs evolve from planning and involvement of stakeholders. The process, which is easily adapted to identifying research needs, involves identifying the goals, reviewing what data exist, developing a monitoring strategy, analyzing results and reviewing the information produced with respect to the initial goals.

The resource assessment program, established in 1974 in Massachusetts, has a specific goal - sound scientific and statistical input for stock assessment, and is considered a valuable contribution to the National Oceanic and Atmospheric Administration (NOAA) database for groundfish (Howe et al., 1979). To its credit, the program has changed very little since 1979. It provides over 20 years of data using consistent methods with NOAA effort. To its detriment, the program has changed very little since 1979. Lack of resources has limited adding research components or additional monitoring activities to any significant degree.

In addition to the data used for stock assessments, food preference data has been collected by NMFS from 1973 to the present. These data were summarized and synthesized in analyses and models that range from basic descriptions and statistically analyses to predictive and theoretical models (NEFSC 1998).

A Final Word

What's gone and what's past help should
be past grief. -William Shakespeare,
1623, The Winter's Tale

...you better start swimmin'
Or you'll sink like a stone
For the times they are a changin'.
-Bob Dylan, 1963

Today, there is an extraordinary need for a more scientific management of our fisheries. By recognizing that species do not exist in isolation from other species, and that each species has adapted to its own specific habitat, the need for ecosystem-based management of our fisheries becomes apparent. A similar recommendation for "ecosystem approach" in the management of our world's resources has been made in a recent publication of the World Resources Institute (2000). Single-species management can no longer be relied upon to address the fishing pressure on wild stock. Considering only single stressors (fishing mortality, contaminant toxicity, habitat alteration) limits management options and may lead to erroneous decisions that have negative effects on other fisheries. Because pollution has had an impact on some fish (e.g., various anadromous fish; winter flounder (*Pseudopleuronectes americanus*), windowpane (*Scophthalmus aquosus*), striped bass (*M. saxatilis*), mussels (*Mytilus edulis*), and lobsters (*Homarus americanus*)) and habitat modifications have affected some species (e.g., anadromous fish and winter flounder), these factors may be having some effect on all of our fish stocks. A variety of well-established ecological tenets must now be incorporated into our management decisions - the finite limitation of marine productivity; the importance of biodiversity for maintenance of ecosystem health and vigor; and the reliance on habitat at critical life history stages. It is time to move towards a more holistic, ecosystem-based paradigm for scientific fisheries management.

In practical terms, however, the fishing community, including the industry, managers, and scientists) is not ready to make this paradigm shift. Neither the data nor the tools (conceptual and predictive models) are available which would allow us to manage any ecosystem. Developing these models, even though the parameters involved are numerous and the overall system complex, should be a high priority. In addition to collecting data and refining our existing models, the use of these models will eventually evolve into the holistic tools that we need.

Many data needs have been identified (e.g., population and recruitment dynamics, predator-prey and competition relationships, identification of essential habitat for all life history stages, contaminant impacts on populations, socio-economic effects on fishing communities). In general, the majority of these parameters are neglected because of the complexity involved and our inability to manage complexity. Applying a precautionary approach to actions that may have adverse effects is a management option that offers the opportunity to incorporate scientific information and new data. Pilot projects and adaptive management approaches should be used to evaluate an action before it is adopted on a broad scale. Investing in obtaining data that supports holistic approaches to managing fisheries within an ecosystem framework will provide a basis for an integrative strategy toward managing all species.

Our observation is that fisheries managers have not effectively used new sources of information (e.g., research and monitoring data from the National Estuarine Program, Sea Grant College Programs, US Geological Survey, and Massachusetts Water Resources Authority). This occurs, in part, because fisheries models are currently population based and the data have not been distributed effectively. Although not discussed explicitly, the need for improved data management and distribution remains key to successful integration of research data and models into management decisions.

Scientifically-based fisheries management can only be achieved through multidisciplinary collaboration. Fisheries managers will need to actively promote discussion and exchange among practitioners from a variety of disciplines (e.g., fisheries science, marine ecology, aquatic toxicology, sociology). In addition, fishermen's knowledge is now being actively solicited and is recognized as an important source of data that had previously been ignored. Scientific insights into ecosystem dynamics and management constraints are less than ideal, but with a long-term research plan, information can be gathered to improve ecosystem-level understanding. The vision for the future is the integration of information from all sources, the development of holistic models that realistically represent ecosystems, and the wise use of knowledge for improved management decisions. With committed individuals in the planning process, the goal of sustainable fisheries in New England waters for the next generation is at hand. Are we up to the challenge?

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Massachusetts Institute of Technology
Sea Grant College Program
Publication No. 05-5

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