NOAA Technical Memorandum NMFS-NE-106

Selected Living Resources, Habitat Conditions, and Human Perturbations of the Gulf of Maine

Environmental and Ecological Considerations for Fishery Management

U. S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Northeast Region
Northeast Fisheries Science Center
Woods Hole, Massachusetts

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NOAA Technical Memorandum NMFS-NE-106

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Selected Living Resources, Habitat Conditions, and Human Perturbations of the Gulf of Maine

Environmental and Ecological Considerations for Fishery Management

Richard W. Langton[1], John B. Pearce[2], and Jon A. Gibson[2], eds.

[1] *Maine Dept. of Marine Resources, West Boothbay Harbor, ME 04575*

[2] *Woods Hole Lab., National Marine Fisheries Serv., Woods Hole, MA 02543*

U. S. DEPARTMENT OF COMMERCE
Ronald H. Brown, Secretary
National Oceanic and Atmospheric Administration
D. James Baker, Administrator
National Marine Fisheries Service
Rolland A. Schmitten, Assistant Administrator for Fisheries
Northeast Region
Northeast Fisheries Science Center
Woods Hole, Massachusetts

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NOTE ON SPECIES NAMES

The NMFS Northeast Region's policy on the use of species names in technical publications and reports is to follow the American Fisheries Society's (AFS) lists of scientific and common names for fishes (Robins *et al.* 1991)[a], mollusks (Turgeon *et al.* 1988)[b], and decapod crustaceans (Williams *et al.* 1989)[c], and to follow the American Society of Mammalogists' list of scientific and common names for marine mammals (Wilson and Reeder 1993)[d]. This policy applies to all issues of the *NOAA Technical Memorandum NMFS-NE* series.

Generally, the "sportsman's singular" style is used for the common names of all organisms (*i.e.,* the "s" or "es" normally appended to the plural form of a noun is omitted).

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[a] Robins, C.R. (chair); Bailey, R.M.; Bond, C.E.; Brooker, J.R.; Lachner, E.A.; Lea, R.N.; Scott, W.B. 1991. Common and scientific names of fishes from the United States and Canada. 5th ed. *Amer. Fish. Soc. Spec. Publ.* 20; 183 p.

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List of Acronyms

AHH	=	aryl hydrocarbon hydroxylase [an MFO]
ATPase	=	adenosine triphosphatase
AWS	=	Acid Waste Site
BαP	=	benzo-α-pyrene [a PAH]
BOD	=	biological oxygen demand
CETAP	=	Cetacean and Turtle Assessment Program [of the University of Rhode Island]
COE	=	U.S. Army Corps of Engineers
DDE	=	dichlorodiphenyldichloroethylene [a breakdown product of DDT]
DDT	=	dichlorodiphenyltrichloroethane
DEHP	=	di-2-ethylhexyl phthalate
DO	=	dissolved oxygen
DWD-106	=	Deepwater Dumpsite 106
EPA	=	Environmental Protection Agency
EROD	=	ethoxyresorufin 4-O-de ethylase
ESA	=	Endangered Species Act
GOM/BOF	=	Gulf of Maine - Bay of Fundy
Hb	=	hemoglobin
Hct	=	hematocrit
ICES	=	International Council for the Exploration of the Sea
LLRW	=	low-level radioactive wastes
MBIWS	=	Massachusetts Bay Industrial Wastesite
MFCMA	=	Magnuson Fishery Conservation and Management Act
MFO	=	mixed-function oxidase or oxygenase [compounds active in breaking down organic hydrocarbons for elimination]
NEFMC	=	New England Fishery Management Council
NEFSC	=	Northeast Fisheries Science Center [of the NMFS]
NMFMP	=	Northeast Multispecies Fishery Management Plan
NMFS	=	National Marine Fisheries Service [of NOAA]
NOAA	=	National Oceanic and Atmospheric Administration [of U.S. Department of Commerce]
NPDES	=	National Pollution Discharge Elimination System
N/P	=	nitrogen-to-phosphorous ratio
PAH	=	polycyclic aromatic hydrocarbon
PCB	=	polychlorinated biphenyl
PSP	=	paralytic shellfish poison
SLW	=	slope water
SSW	=	Scotian Shelf water
USGS	=	U.S. Geological Survey
WSF	=	water-soluble fraction [of crude oils]
YONAH	=	Years of the North Atlantic Humpback Program

INTRODUCTION

***Richard W. Langton*[1] *and John B. Pearce*[2]**

The Gulf of Maine is a distinct body of water along the eastern seaboard of the United States and maritime Canada. It is a relatively new body of water on a geological time scale since it was a product of the last ice age, approximately 11,000 yr before present (Emery and Uchupi 1972; Ziegler *et al.* 1965). It is bounded on the landward portion by Massachusetts, New Hampshire, Maine, New Brunswick, and Nova Scotia. It is a continental shelf sea with the seaward boundary being formed by Georges and Browns Banks, and has most recently been described as a functional macroestuary (Campbell 1986).

Exploitation of fisheries resources of the region has continued without interruption by European and, ultimately, American fishing fleets for over 500 yr (Collins and Rathbun 1885). Resource harvesting by native American peoples occurred even earlier as evidenced by numerous shell middens dotting the Maine coast (Moorehead 1922; Borque, in press). Efforts to manage this harvesting also have a long history, with restrictions on cod catch, for example, dating to the General Court of Massachusetts Bay Colony in 1639 (Jensen 1984).

Although depletion of fish stocks was almost universally perceived as a local problem, the cumulative effect of fishing has resulted in drastic declines in populations of most demersal fish species (Conservation and Utilization Division 1993). The most dramatic declines, dating back to international exploitation of Georges Bank fisheries resources in the 1960s, resulted in passage of the Magnuson Fishery Conservation and Management Act (MFCMA) in 1976, and ushered in a new era of fisheries management that considers regional-scale effects and solutions for resource harvesting.

The MFCMA created eight regional fishery management councils that are charged with developing fishery management plans (FMPs) for the various fisheries resources under their jurisdiction. The FMPs are designed both to conserve and maintain resources, and to provide optimum yields on a continuing basis. The original demersal fish FMP was introduced in 1977, and focused on species catch quotas to control overfishing. Quota management became unworkable; the New England Fishery Management Council (NEFMC) ultimately replaced it in 1986 with the Northeast Multispecies Fishery Management Plan (NMFMP) which is now operating under Amendment 5 enacted in 1993. Current objective of the NMFMP is to maintain a stock size for each demersal fish stock that insures survival of a specified percentage of that stock's maximum spawning potential.

All these plans to manage demersal fish include environmental impact statements that, like the strategies to reduce fishing mortality *per se*, have undergone evolutionary change. Although overfishing is credited with the magnitude of demersal fish population declines that have been observed (Conservation and Utilization Division 1993; New England Fishery Management Council 1993), other factors have to be considered in the fisheries management process. This has given rise to an expanded environmental impact statement that deals with a variety of issues ranging from pollution, to habitat destruction, to natural life history, to environmental interactions.

Recent publications on the Gulf of Maine include: a National Oceanic and Atmospheric Administration (NOAA) National Undersea Research Program report on benthic productivity (Babb and DeLuca 1988); a NOAA Coastal Ocean Program overview of the gulf (Townsend and Larson 1992); the Gulf of Maine Scientific Workshop proceedings (Wiggin and Mooers 1992); a NOAA Status and Trends Program report on toxic contaminants (Gottholm and Turgeon 1992); three Regional Association for Research on the Gulf of Maine workshop proceedings (Phelps *et al.* 1993; Braasch 1994; Stevenson and Braasch 1994); the most recent issue of the Northeast Fisheries Science Center's (NEFSC's) annual report on the status of the stocks (Conservation and Utilization Division 1993); and a data report prepared by NOAA's Estuarine Living Marine Resources Program (Jury *et al.* 1994). Few of these publications, however, consider both the benthic habitat in the Gulf of Maine and its relation to the demersal fish populations of the region. Because of this information void and as a result of the effort put into the preparation of the NEFMC's environmental impact statement for Amendment 5, this publication has been prepared. It is a revised and expanded version of the NMFMP's environmental impact statement, and reflects contributions by a number of members of New England's scientific community.

The rest of this publication comprises 11 individually authored sections and a unified list of cited references. It begins with a review by Mayo of the life histories and general habitat requirements of 13 demersal fish species. This is immediately followed by a review by Langton on fishing effects on demersal fish habitats. Hain and Waring follow with a discussion of the statuses and human effects upon marine mammals. Mountain, Langton, and Watling next describe the large-scale physical oceanography of the system, and relate this to demersal fish habitat and benthic community structure. O'Reilly considers nutrient-loading and eutrophication effects on the system, and Gould, Clark, and Thurberg thoroughly review pollutant effects on demersal fish species in the region. In a look at specific non-fishing-related activities and their effects on fisheries resources and their habitats: Barr and Wilk consider sewage sludge and industrial waste dumping; Kurland, Ludwig, Gorski, and Mantzaris consider dredging and dredged-material disposal; Pearce considers mining of seabed aggregates; and Wilk and Barr consider the multiple-use issues in estuarine and coastal habitat loss. Finally, a summary by Pearce and Gibson touches on highlights of the publication, and identifies some research priorities for the region.

[1] Maine Department of Marine Resources, West Boothbay Harbor, ME.
[2] National Marine Fisheries Service, Woods Hole, MA.

LIFE HISTORIES AND HABITAT REQUIREMENTS OF DEMERSAL FISHES

Ralph K. Mayo[3]

ATLANTIC COD

Atlantic cod (*Gadus morhua*) are distributed in the Northwest Atlantic from Greenland to Cape Hatteras, North Carolina, and from nearshore waters to depths exceeding 400 m. In U.S. waters, they are assessed as two stocks: (1) Gulf of Maine, and (2) Georges Bank and southward. The Gulf of Maine stock supports important commercial and recreational fisheries. Commercial fisheries are conducted year-round with otter trawls and gill nets as primary gear. Recreational fishing also occurs year-round; peak activity occurs during the late summer in the lower Gulf of Maine.

Off the northeastern United States, the greatest concentrations of cod are commonly found on rough bottoms in waters between 10 and 150 m, and at temperatures between 0° and 10°C. They are omnivorous, eating a wide variety of mollusks, crustaceans, and fishes. Growth is rapid; in the Gulf of Maine, they attain an average total length of 26 cm by the end of their second year of life, and are captured routinely in commercial and recreational fisheries by age 3 (40-50 cm). They commonly attain lengths up to 130 cm and weights up to 25-35 kg. Growth parameters, taken from Penttila and Gifford (1976) are listed in Table 1. Maximum age exceeds 20 yr, although young fish (ages 2-5) generally constitute the bulk of the catches. Instantaneous rate of natural mortality is 0.20 (Conservation and Utilization Division 1993). Median age at sexual maturity is between 1.7 and 2.3 yr, and is attained at average total lengths between 32 and 41 cm (O'Brien *et al.* 1993).

Spawning by cod occurs during winter and early spring, normally at water temperatures between 5° and 7°C. Spawning takes place near the bottom, but eggs are pelagic and drift for 2-3 wk before hatching. Transition from pelagic to demersal life occurs when larvae are about 4-6 cm in length, or about 3-mo old. They are highly fecund; a large mature female may produce 3-9 million eggs.

In New England waters, cod characteristically move into shoal water in spring, followed by a retreat into deeper water during winter. Little interchange occurs between cod in the Gulf of Maine and those on Georges Bank.

HADDOCK

Haddock (*Melanogrammus aeglefinus*) are distributed on both sides of the North Atlantic. In the western Atlantic, they range from West Greenland to Cape Hatteras. Highest concentrations off the U.S. coast occur on the northern and eastern portions of Georges Bank and in the southwestern Gulf of Maine (Clark *et al.* 1982). Two stocks occur in U.S. waters: (1) Gulf of Maine, and (2) Georges Bank.

Haddock are most common at depths of 45-135 m, and temperatures of 2-10°C (Collette and Klein-MacPhee [1996]). They appear to prefer broken ground, and gravelly, pebbly, or sandy bottoms rather than ledges, rocks, or kelp (Collette and Klein-MacPhee [1996]). Adult in the western Gulf of Maine show seasonal coastal movements.

Some spawning of haddock occurs along the Maine coast. Spawning occurs between January and June, with peak activity during late March and early April. Eggs are planktonic, and hatch in approximately 2 wk at typical spawning temperatures (Clark *et al.* 1982). Juveniles remain pelagic for several months before settling to the bottom (Collette and Klein-MacPhee [1996]).

As planktonic juveniles, haddock mainly consume copepods and euphausiids. After assuming a benthic habit, they prey primarily on small invertebrates, but fishes are also consumed by adults (Collette and Klein-MacPhee [1996]).

Haddock are moderately long lived (up to approximately 18 yr), and have relatively rapid growth (Table 1). Instantaneous rate of natural mortality is 0.20 (Conservation and Utilization Division 1993). During the early 1960s, nearly all females age 4 and older were fully mature, and approximately 75% of age-3 females were mature. In recent years, the maturation schedule has shifted by about 1 yr; currently, nearly all age 3 and 75% of age-2 females are mature (O'Brien *et al.* 1993). Individual females may produce up to 3-million eggs, but a 55-cm individual produces approximately 850-thousand eggs (Clark *et al.* 1982).

POLLOCK

Pollock (*Pollachius virens*) occur on both sides of the North Atlantic; in the Northwest Atlantic, they are most abundant on the Scotian Shelf and in the Gulf of Maine. One major spawning area exists in the western Gulf of Maine, and several areas have been identified on the Scotian Shelf. Tagging studies suggest considerable movement between the Scotian Shelf and Georges Bank, and--to a lesser extent--between the Scotian Shelf and the Gulf of Maine. Electrophoretic analyses of pollock tissue samples from the Scotian Shelf and western Gulf of Maine showed no significant differences between fish from the two areas, although differences in some morphometric and meristic characteristics were significant (Mayo *et al.* 1989).

[3] *id.*

Table 1. Von Bertalanffy growth parameters for 12 species of demersal fishes in the Gulf of Maine region

Species (Stock)	Sex	L_∞(cm)	K	t_0
Atlantic cod				
(Gulf of Maine)		146.5	0.116	0.285
(Georges Bank)		148.1	0.120	-0.616
Haddock				
(Gulf of Maine)		72.91	0.352	0.295
(Georges Bank)		73.80	0.376	0.165
Pollock				
(Gulf of Maine - Georges Bank)		107.4	0.1664	-0.3214
Acadian redfish				
(Gulf of Maine - Georges Bank)	M	32.8	0.1693	-0.3546
	F	38.9	0.1452	-0.1200
White hake				
(Gulf of Maine)	M	110.6	0.11	1.17
	F	135.3	0.09	-0.89
Red hake				
(Gulf of Maine to Mid-Atlantic)		60.19	0.191	-0.836
Silver hake				
(Gulf of Maine - Northern Georges Bank		65.5	0.181	-0.275
Yellowtail flounder				
(Georges Bank)	M	41.0	0.748	0.671
	F	46.0	0.629	0.676
Winter flounder				
(Gulf of Maine)	M	39.8	0.41	0.38
	F	49.0	0.27	0.07
(Georges Bank)	M	55.0	0.37	0.05
	F	63.0	0.31	0.05
American plaice				
(Gulf of Maine - Georges Bank)	M	59.83	0.17	-0.04
	F	64.17	0.17	0.12
Witch flounder				
(Gulf of Maine - Georges Bank)	M	58.05	0.1533	-0.0120
	F	61.99	0.1482	0.0542
Windowpane				
(Georges Bank)		37.2	0.39	0.1

Note: The basic von Bertalanffy growth equation is $l_t = L_\infty[1-e^{-K(t-t_0)}]$, where: l_t = length at age t; L_∞ = ultimate length for the stock; e = base of natural logarithms; and t_0 = time when length would theoretically be zero.

Pollock attain total lengths up to 110 cm, and weights of 16 kg. Maximum ages up to 18 yr have been noted in the population, although the major portion of the catch consists of 3-6 yr old fish. Von Bertalanffy growth parameters, taken from Mayo *et al.* (1989), are listed in Table 1. Instantaneous rate of natural mortality is 0.20 (Conservation and Utilization Division 1993). Sexual maturation is essentially complete by age 5, although most fish are mature by age 3 (O'Brien *et al.* 1993).

Adult pollock inhabit depths from 70 to 280 m, with associated bottom temperatures between 5° and 8°C in the Gulf of Maine and along the northern edge of Georges Bank (Leim and Scott 1969). Juvenile "harbor pollock" are common in inshore areas, but move offshore as they grow older. Adults form spawning aggregations during winter months in the western Gulf of Maine where considerable fishing effort is directed.

Juvenile pollock feed primarily on euphausiids and other small crustaceans (Collette and Klein-MacPhee [1996]). As adults, they appear to select larger crustaceans as well as small fishes, primarily Atlantic herring (*Clupea harengus*). There appears to be little preference for bottom type as pollock are often found to be semipelagic.

ACADIAN REDFISH

Acadian redfish (*Sebastes fasciatus*) are distributed throughout the Northwest Atlantic from the Grand Banks to Georges Bank. Off New England, they are most common in deep waters of the Gulf of Maine to depths of 300 m.

Ages in excess of 50 yr, and maximum sizes of 45-50 cm, have been observed. Von Bertalanffy growth parameters, taken from Mayo *et al.* (1990), are listed in Table 1. Instantaneous rate of natural mortality is quite low; a value of 0.05 has been used in evaluating population dynamics of this stock (Conservation and Utilization Division 1993). Median age at maturity for Gulf of Maine redfish (5.5 yr) is attained at an average total length of 20-23 cm (O'Brien *et al.* 1993).

Female redfish are viviparous, retaining eggs in the ovary after fertilization until yolk-sac absorption. Mating takes place in autumn, with subsequent larval extrusion occurring the following spring and summer. Larvae remain planktonic for 4-5 mo before descending to the bottom at a length of about 50 mm (Kelly and Barker 1961).

A strong diel vertical distribution pattern has been observed in adult redfish, with considerable movement off bottom at night and a return during daylight. They are often associated with rocky bottoms in the Gulf of Maine, and are most abundant in relatively cold water, usually below 5°C. Adults feed primarily on small copepods (Bigelow and Schroeder 1953).

In the past, redfish were often distributed in numerous dense local aggregations throughout the Gulf of Maine. Because of this, they were fished quite heavily during the development phase of the fishery. Because of their low fecundity and low instantaneous rate of natural mortality, they are particularly vulnerable to increased mortality.

WHITE HAKE

The white hake (*Urophycis tenuis*), a boreal species that occurs from Newfoundland to Southern New England, is found on muddy bottom throughout the Gulf of Maine. Stock boundaries are uncertain, although research vessel survey data indicate that the Gulf of Maine population is more or less discrete from populations further north and east. White hake in the Gulf of Maine appear to be recruited from an early spring spawning population located on the continental slope south of Georges Bank and Southern New England (Fahay and Able 1989).

White hake attain a maximum total length of 135 cm, and weights of up to 21 kg, with females being larger. Ages of over 20 yr have been documented. Von Bertalanffy growth parameters, taken from Hunt (1982), are listed in Table 1. Median age at maturity is about 1.5 yr at total lengths between 32 and 35 cm (O'Brien *et al.* 1993).

Depth distribution of white hake varies by age and season. Juveniles typically occupy shallower areas than adults, but individuals of all ages tend to move inshore or shoalward in spring and summer, and to disperse to deeper areas in autumn. Most trawl catches are taken at depths of 110 m or more, although white hake are taken as shallow as 27 m during gill netting operations in summer. White hake occur within a relatively broad temperature range. Juveniles occur in shallow water as warm as 15°C in summer, or as cool as 2°C in winter. Adults occur in regions of the inner Gulf of Maine at temperatures as cool as 5°C.

Juvenile white hake feed primarily on shrimp and other crustaceans. Adults feed almost exclusively on fishes, including juveniles of their own species.

RED HAKE

Red hake (*Urophycis chuss*) are distributed from the Gulf of St. Lawrence to North Carolina, but are most abundant between Georges Bank and New Jersey. Stock structure of this species is not clearly defined. There are possibly two stocks, divided north and south in the central Georges Bank region.

Red hake have a broad geographic and depth distribution throughout the year, undergoing extensive seasonal migrations. They overwinter in deep waters of the Gulf of Maine and along the outer continental shelf and slope south and southwest of Georges Bank. They are most common in relatively deep water, and apparently prefer sandy or muddy bottoms (Collette and Klein-MacPhee [1996]). Adults prefer water temperatures of 5-12°C.

Spawning of red hake occurs from May through November, with major spawning areas located on southwestern Georges Bank and south of Montauk Point, Long Island. Spawning generally occurs at temperatures between 5° and 10°C. Eggs are small, buoyant, and pelagic, with hatching occurring within approximately 3-7 days at typical spawning temperatures (Collette and Klein-MacPhee [1996]). Young fish remain pelagic for the first few months of life, becoming demersal after reaching approximately 30 mm in length. After settling, juveniles are commonly occur within mantle cavities of sea scallops (*Placopecten magellanicus*), maintaining this association until roughly 100 mm in length (Collette and Klein-MacPhee [1996]).

As planktonic larvae and juveniles, red hake feed largely on copepods and other small crustaceans. Demersal red hake also feed primarily on crustaceans (*e.g.*, decapod shrimps, euphausiids, amphipods, and crabs), but adults also feed extensively on fishes (Collette and Klein-MacPhee [1996]).

Red hake are relatively short-lived, reaching a maximum age of about 12 yr. Few fish survive beyond age 8, however. Growth is initially rapid, but the species does not reach a large size. Von Bertalanffy growth parameters, calculated from Penttila *et al.* (1989), are listed in Table 1. Instantaneous rate of natural mortality is 0.40 (Conservation and Utilization Division 1993). Maturity is reached at age 1.7-1.8 (*i.e.*, a length of 25-27 cm) for females, and age 1.4-1.8 (22-24 cm) for males (O'Brien *et al.* 1993).

SILVER HAKE

Silver hake (*Merluccius bilinearis*) are widely distributed, ranging from Newfoundland to South Carolina. Center of abundance is from Maine to New Jersey. Two stocks have been identified, based on morphological differences: one extends from the Gulf of Maine to northern Georges Bank, and the other extends from southern Georges Bank to the Mid-Atlantic area. Migration is extensive, with overwintering in deeper waters of the Gulf of Maine for the northern stock.

Silver hake are found at a variety of depths, from the shoreline to depths as great as 900 m. Their preferred temperature range is 6-18°C (Collette and Klein-MacPhee [1996]). Although they show seasonal migrations, movements within these broad depth and temperature ranges appear to be related to distribution of food organisms.

A major spawning area of silver hake is the coastal Gulf of Maine from Cape Cod to Grand Manan Island. They are a summer spawner, with peak egg production occurring during July and August (Collette and Klein-MacPhee [1996]). Eggs are buoyant and hatch within 2-3 days of fertilization (Collette and Klein-MacPhee [1996]). Larvae are passive planktors until reaching a length of approximately 20 mm when they become able to migrate vertically within the water column in search of preferred water temperatures and prey (Collette and Klein-MacPhee [1996]).

As juveniles, silver hake feed primarily on small crustaceans such as copepods, amphipods, and euphausiids (Bowman 1981). After reaching approximately 20 cm in length, their diet shifts to primarily fishes, squids, and decapod shrimps (Bowman 1984). Feeding occurs mainly at night (Bowman and Bowman 1980; Bowman 1984).

Growth of silver hake is initially rapid, but after reaching 25 cm in total length, there is a divergence in growth between males and females, with females growing more rapidly and achieving a larger maximum size (Hunt 1980). Von Bertalanffy growth parameters, calculated from Penttila *et al.* (1989), are listed in Table 1. Ages up to 15 yr have been reported, but few fish beyond age 6 have been observed in recent years. Instantaneous rate of natural mortality is 0.40 (Conservation and Utilization Division 1993). More than 50% of age 2 fish (20-30 cm), and nearly all age 3 fish (25-35 cm), are sexually mature (O'Brien *et al.* 1993).

OCEAN POUT

The ocean pout (*Macrozoarces americanus*) is a demersal eel-like species ranging from Labrador to Delaware that attains lengths of up to 98 cm and weights of 5.3 kg. Stock identification studies suggest existence of two stocks: one occupying the Bay of Fundy - northern Gulf of Maine region east of Cape Elizabeth, and the other ranging from Cape Cod Bay south to Delaware (Olsen and Merriman 1946). The southern stock is characterized by faster growth rates, and to date has supported the commercial fishery.

Ocean pout prefer depths of 15-80 m (Grosslein and Azarovitz 1982), and temperatures of 6-9°C (Collette and Klein-MacPhee [1996]). Tagging studies and bottom trawl survey data indicate that ocean pout do not undertake extensive migrations, but rather move seasonally to different substrates. During winter and spring, they feed over sand or sand-gravel bottom, and are vulnerable to otter trawl fisheries (Olsen and Merriman 1946). In summer, they cease feeding and move to rocky areas, where spawning occurs in September and October.

Median length at which maturation occurs in the northern stock of ocean pout is 30.3 cm for males and 26.2 cm for females (O'Brien *et al.* 1993). Egg development is longer than 1 yr, and fecundity is relatively low. Eggs are demersal and guarded by both parents until hatching.

During nesting, ocean pout are not available to commercial fishing operations. Catches typically increase again when adults return to their feeding grounds in late autumn and winter.

Diet consists primarily of invertebrates, with fishes being only a minor component.

YELLOWTAIL FLOUNDER

The yellowtail flounder (*Pleuronectes ferrugineus*) ranges from Labrador to Chesapeake Bay. Off the U.S. coast, the largest commercial concentrations are found on Georges Bank, off Cape Cod, and in Southern New England waters. Some fishing for this species by the U.S. fleet does occur, however, in the northern Gulf of Maine. The are generally found at depths between 10 and 100 m, and on sand or sand-mud substrates (Collette and Klein-MacPhee [1996]). Although they are relatively sedentary, some seasonal movements have been documented. Tagging studies and other information indicate that Southern New England, Georges Bank, and Cape Cod yellowtail flounder form relatively discrete groups, although some intermingling of fish among these groups occurs (Lux 1963).

Maturity is attained at roughly ages 2-3 for all stocks, with median length at maturity occurring at 24.9 and 25.6 cm for females and males, respectively, in the Georges Bank stock, and at 27.6 and 29.0 cm for females and males, respectively, in the Southern New England stock (O'Brien *et al.* 1993). Von Bertalanffy growth parameters, taken from Moseley (1986), are listed in Table 1. The ultimate length listed in the table is lower than might be expected because of the modeling technique and the lack of older age samples in the available data. Historical data provide evidence of significant numbers of fish reaching 50 cm. Instantaneous rate of natural mortality is 0.20 (Conservation and Utilization Division 1993).

Fecundity is 350-570 thousand eggs per spawner. Spawning occurs in spring and summer, peaking during April to June at water temperatures of 4.5-8.1°C (Collette and Klein-MacPhee [1996]). Larvae drift for 1 mo or more, then assume adult characteristics and become demersal.

Food habits data suggest that diets consist almost entirely of small invertebrates such as mysids, amphipods, and polychaetes (Collette and Klein-MacPhee [1996]).

WINTER FLOUNDER

The winter flounder (*Pleuronectes americanus*) is distributed in the Northwest Atlantic from Labrador to Georgia. Abundance is highest from the Gulf of St. Lawrence to Chesapeake Bay. Tagging and meristic studies indicate discrete groups north (Gulf of Maine) and south (Southern New England - Middle Atlantic) of Cape Cod, and on Georges Bank. Movements are generally localized. Restricted movements and differences in growth, meristic, and morphometric characteristics suggest that relatively discrete local groups also exist.

Winter flounder may attain total lengths up to 58 cm, and ages in excess of 15 yr. Von Bertalanffy growth parameters for Georges Bank fish, taken from Lux (1973) and Witherell and Burnett (1993), are listed in Table 1. Median age at maturity for male and females north of Cape Cod is 3.3 and 3.5 yr, respectively (28 and 30 cm) (O'Brien *et al.* 1993).

Spawning of winter flounder commences in early winter at the southern extent of the range, and may extend into April and May on Georges Bank. Height of spawning in most areas is between January and March. Spawning among coastal winter flounder occurs in estuaries, embayments, and saltwater ponds. There is evidence that winter flounder migrate to the same spawning locations in consecutive years.

Habitat preferences of winter flounder range from gravel to sand to muddy sand. Populations on offshore banks are generally on hard bottom. Normal distribution of winter flounder covers a wide range of temperatures, from the freezing point of salt water at the northern edge of the range, to a maximum of 19-20°C at the southern limit. Preferred temperatures appear to be between 3° and 15°C. Peak spawning activity occurs when water temperatures range from 3.3° to 5.6°C.

Diet of winter flounder consists primarily of small benthic invertebrates, such as shrimps, amphipods, small crabs, annelid worms, mollusks, hydrozoans, and anthozoans (Klein-MacPhee 1978).

AMERICAN PLAICE

The American plaice (*Hippoglossoides platessoides*) is distributed along the Northwest Atlantic continental shelf from southern Labrador to Rhode Island in relatively deep waters. Primary concentrations of the Gulf of Maine - Georges Bank stock occur in inshore waters along coastal Maine and Massachusetts from Casco Bay to Cape Cod Bay including Jeffreys Basin and Stellwagen Bank, and in offshore waters from the central Gulf of Maine to the Great South Channel.

Maximum age of American plaice is between 24 and 30 yr, and maximum total length is between 70 and 80 cm (Bigelow and Schroeder 1953). Growth rates are similar between sexes until age 4 when females grow faster than males. Von Bertalanffy growth parameters, taken from Sullivan (1982), are listed in Table 1. Instantaneous rate of natural mortality is 0.20 (Conservation and Utilization Division 1993).

Median age at maturity for American plaice females (3.6 yr) and males (3.0 yr) is attained at an average total length of 26.8 and 22.1 cm, respectively (O'Brien *et al.* 1993). Spawning occurs from February to June in the coastal areas of the western Gulf of Maine from Cape Elizabeth to Cape Cod (including Stellwagen Bank and Cape Cod Bay), and on central and western Georges Bank in depths no greater than 90 m (Smith 1985; Collette and Klein-MacPhee [1996]).

American plaice inhabit areas either with a fine but gritty mixture of sand and mud, or with soft oozy mud, primarily in depths of 27-108 m. Optimum temperatures are 1.7-7.7°C; however, plaice can survive in water from -1.5 to 13°C. Spawning occurs in a narrower range of 2.7-4.4°C (Collette and Klein-MacPhee [1996]).

Preferred prey of adult American plaice are echinoderms, amphipods, and polychaetes; however, they are opportunistic and will feed on most bottom-dwelling animals small enough to devour. Juveniles prefer small shrimps, other crustaceans, and polychaetes.

WITCH FLOUNDER

The witch flounder (*Glyptocephalus cynoglossus*) is common throughout the Gulf of Maine, and also occurs in deeper areas on and adjacent to Georges Bank and along the shelf edge as far south as Cape Hatteras. Research vessel survey data suggest that the Gulf of Maine population may be relatively discrete from populations in other areas. Witch flounder appear to be sedentary, preferring moderately deep areas; few fish are taken shallower than 27 m, and most are caught between 110 and 275 m.

Witch flounder attain total lengths up to 60 cm and weights of approximately 2 kg. Von Bertalanffy growth parameters, taken from Burnett *et al.* (1992), are listed in Table 1. Instantaneous rate of natural mortality is 0.15 (Conservation and Utilization Division 1993).

Median age at maturity of witch flounder (3.6 yr for males and 4.4 yr for females) is attained at an average total length of 25.3 cm for males and 30.4 cm for females (O'Brien *et al.* 1993). Spawning occurs in late spring and summer, with peak spawning occurring in July and August.

Witch flounder are frequently caught on smooth bottom where muddy sand, clay, or mud occurs. They occupy waters with temperatures of about 1.7-14.5°C.

Diet of witch flounder consists primarily of polychaetes. It also includes echinoderms, amphipods and other small crustaceans, and squids and other small mollusks (Collette and Klein-MacPhee [1996]).

WINDOWPANE

Windowpane (*Scophthalmus aquosus*) is a thin-bodied, left-handed flounder distributed on the Northwest Atlantic continental shelf from the Gulf of St. Lawrence to Florida. There is no information on stock structure. Greatest commercial concentrations exist on Georges Bank and in Southern New England waters at depths less than 46 m.

Sexual maturity of windowpane occurs between ages 3 and 4. About 50% of fish that are 22 cm in total length are mature (O'Brien *et al.* 1993). Spawning occurs from late spring to autumn, peaking in July-August on Georges Bank and September in Southern New England. Windowpane spawn at temperatures between 6 and 17°C, and at depths less than 40 m (Collette and Klein-MacPhee [1996]).

Windowpane commonly attain total lengths up to 41 cm. Von Bertalanffy growth parameters, taken from Thorpe (1991), are listed in Table 1.

Windowpane primarily inhabit waters between the tide mark and 45 m where bottom sediments are sand or mud. Young fish settle in shallow water inshore, and move offshore and deeper as they grow. Adults tolerate a wide temperature range, between 0 and 26.8°C, but in summer they occur primarily in waters where surface temperatures are 13°C or higher.

Windowpane feed intensively on mysid shrimp as juveniles and adults. Adults over 20 cm in total length also take small or young fishes as prey (Collette and Klein-MacPhee [1996]).

FISHING EFFECTS ON DEMERSAL FISH HABITATS

Richard W. Langton[4]

TRAWLING AND DREDGING

Trawling and dredging effects on the seabed have been a concern for centuries. An English report from 1376 pointed to the potentially destructive nature of dredging (see Messieh *et al.* 1991; Shepard and Auster 1991; Jones 1992). Nevertheless, such harvesting practices continued, and there is now renewed interest in their effects on target species and associated fauna. The International Council for Exploration of the Sea (ICES), a scientific body primarily concerned with the North Atlantic, has recently produced three documents discussing effects of trawling and dredging (International Council for Exploration of the Sea 1988, 1991a, 1992b), while three other reviews, focusing on effects of trawling on the seabed, have been published by Australian, Canadian, and New Zealand scientists (Hutchings 1990; Messieh *et al.* 1991; Jones 1992). There are also comprehensive bibliographies produced by ICES to document fishing effects at the ecosystem level (Redant 1987, 1990). Salient points for Northwest Atlantic fisheries from these reviews are summarized below, but ancillary information relating ecosystem effects of fishing activities can be found, in particular, in the 1991 and 1992 ICES documents, while a more exhaustive list of literature is available in the mentioned bibliographies and the other reviews.

Although concern about effects of fishing on the environment has been extant for centuries, gear and techniques used prior to the 20th century had very restricted, localized, environmental effects. The first steam engines and steam-powered capstans were, for example, introduced in the 1860s and 1870s. Mechanical power soon replaced sail power, and diminished the importance of wind as a factor controlling fishing. The first American steam trawler, the *Spray*, built in Boston and launched in 1905, was to be used together with the otter trawl. The steam trawler and implementation of the otter trawl ushered in an era of potential environmental effects in the Northwest Atlantic that is just beginning to be researched (*e.g.*, Brylinsky *et al.*, in press) and understood.

One of the first quantitative investigations of the effects of gear on the bottom was carried out by Graham (1955) on plaice (*Pleuronectes platessa*) in the North Sea when he compared catches from fished grounds and control areas. He concluded that trawling had no long-lasting effect on macrobenthos. Other European researchers have subsequently reported on effects of gear utilized by their fleets (Rendant 1987, 1990; Bergman and Hup 1992). Studies on effects of otter trawling are rare (Ketchen 1947; Krost *et al.* 1990), and specific studies of otter trawling effects on soft-bottom areas off the U.S. coast are virtually nonexistent, although Apollonio (1989) suggested that banning the otter trawl might be a key to better fisheries management.

Effects of fishing on hard bottom have been investigated along the southern coast of the United States, with Wenner (1983) suggesting that trawl gear may reduce the amount of productive fish habitat. This stimulated a study of sponge assemblages and corals. After a single trawling event through the area, Van Dolah *et al.* (1987) followed a hard-bottom assemblage of sponges and corals. They concluded that, after 1 yr, trawling effects could not be detected. In contrast, a study by Brylinsky *et al.* (in press) documented persistence of trawl tracks for 2-7 mo in intertidal areas of Minas Basin, although actual effects on biota were slight in the more-or-less pristine, naturally stressed environment of the Bay of Fundy. Furthermore, in a region that experienced a substantial increase in pair trawling over 16 yr, an Australian study showed a significant decline in sponge bycatch (Sainsbury 1987). Loss of sponges and associated cnidarian benthos led to a reduction in fish catch (Sainsbury 1988; Hutchings 1990).

European literature on trawling effects is much more extensive than comparable work in the U.S. or Canadian waters (International Council for the Exploration of the Sea 1991a; Messieh *et al.* 1991; Brylinsky *et al.*, in press). Studies in the North Sea focused on beam trawls which are generally not used in New England fisheries. Nevertheless, some conclusions regarding physical disturbance are potentially applicable to investigations of the Northwest Atlantic fisheries. It is interesting, for example, that beam trawling in the southern North Sea has actually been associated with an increase in growth rate of flatfish (especially sole). In areas that are trawled, habitat has been altered to favor production of polychaetes which are a primary food for sole. The ICES young fish surveys do not show any reduction in recruitment of flatfish populations, and, therefore, it was concluded that there is no major deterioration in the environment of the nursery areas (International Council for the Exploration of the Sea 1988). This conclusion reflects an alteration of the environment towards reduced species diversity, and a maintenance essentially of the area for monoculture of fish. Other studies have also shown that one immediate consequence of trawling is more food being available to predatory fishes (Arntz and Weber 1970; Medcof and Caddy 1971; Caddy 1973). In these cases, neither increased growth rates for fish nor changes in species diversity were investigated.

[4] Maine Department of Marine Resources, West Boothbay Harbor, ME.

Trawling physically removes or destroys many epibenthic animals, and the effects that this has on the ecosystem must be considered. As stated above, a decrease in species diversity is to be expected in areas that are heavily fished (Jones 1992; Hutchings 1990; Sainsbury 1987). Although this may be considered in positive terms from the point of view of fish production (*e.g.*, North Sea sole), a number of papers point to the negative implications that this has on overall benthic communities, as well as some fish populations. Jones (1992) reviewed papers dealing with destruction of nontarget benthos and pointed out several examples of habitat loss, some of which have resulted in a necessity to close a fishery. Bryozoan beds in Tasman Bay, New Zealand, for example, serve as habitat for juvenile snapper and tarakihi; many of these beds were destroyed by trawling. Remaining beds were ultimately closed to fishing (Bradstock and Gordon 1983). Trawling has also been suggested as a cause of spreading benthic animals, such as mussels, through the Norderaue area of the Wadden Sea (Reise 1982; Reise and Schubert 1987).

BYCATCH AND PROCESSING DISCARDS

In addition to physical effects of trawls themselves, the consequences of discarding both bycatches and processing wastes must be discussed. Bycatch information specifically for the Northwest Atlantic is limited to a paper by Jean (1963) dealing with the New Brunswick fisheries, as well as a paper by Howell and Langan (1987) that focuses on the Gulf of Maine. Howell and Langan (1987) found that mean discard rates in the flatfish fisheries in 1983 ranged from 5 to 25% of the catch, by weight, and 12 to 57% of the catch numerically, depending on the fish species and gear. There are also reports by the NEFSC (Mayo *et al.* 1981) as well as current efforts by the NEFSC to collect such information for use in the Northeast Regional Stock Assessment Workshops (*e.g.*, Clark and Power 1991).

Data on the amount of fish processing wastes are not available, but a rough estimate of organic matter returned to the sea could be calculated based on landings data and knowledge of a total-weight-to-gutted-weight relationship for different demersal fish species (see International Council for the Exploration of the Sea 1991a). Fates of this organic matter are also unknown, although there are two reports of sea scallop processing wastes being identified in stomachs of Atlantic cod (Langton and Bowman 1980; Bowman and Michaels 1984).

LOST GEAR

Lost gear and the potential for continued fishing by that gear are also an environmental effect that should be considered. Unfortunately, quantitative data documenting the extent of lost gear are extremely limited. Cooper *et al.* (1988) estimated that there were 2,497 ghost gill nets on the seabed of Jeffreys Ledge, or within the vicinity of Stellwagen Bank, based on manned submersible surveys in 1985 and 1986. They do, however, point out that this is an extrapolation from direct observations over a series of submersible transects; a much more extensive survey would be required to confirm this number.

Cooper *et al.* (1988) observed behavior of lost gill nets, and Carr (1988) noted that ghost nets continued to fish with a vertical profile of 0.1-2.0 m off the bottom, although the profile of most of the nets observed rarely reached 2.0 m. They estimated that most derelict gill nets observed were 4-7 yr old, and were lost when the Gulf of Maine gillnet fishery was very active and composed of fishermen with "relatively little experience."

Subsequent to these studies, Carr *et al.* (1992) did some experimental work to try to reduce the ghost gillnet fishing problem. They observed an experimental net over 2 yr, and concluded that it continued to fish over time with the species composition of the catch changing with a reduction in the net's vertical profile. Greatest reduction in fishing seemed due to fouling of the net and to an increased visibility of it to fish.

FISHING EFFECTS OVERVIEW

As pointed out in an ICES ecosystem report (International Council for the Exploration of the Sea 1991a), and as essentially stated by Cooper *et al.* (1988) in regard to their gillnet study, a fundamental problem with many, if not all, fishing effects studies is that of scale. The time and space scale necessary to judge accurately the consequences of fishing do not correspond with the actual extent and intensity of these research activities. The tendency is to extrapolate from relatively small localized studies to regions, or even other geographic localities.

It must be recognized that investigations of fishing disturbances are likely to result in "patchy" data within and between different habitats. The degree of patchiness and the frequency of disturbance have a profound influence on the ability of the seabed and on fish populations *per se* to recover from such perturbations.

STATUSES OF AND HUMAN EFFECTS UPON MARINE MAMMALS

James H.W. Hain[5] *and Gordon T. Waring*[6]

SPECIES COMPOSITION AND DISTRIBUTION

There are 15 species of cetaceans and two species of pinnipeds that regularly inhabit the Gulf of Maine, Georges Bank, and adjacent continental shelf edge (Table 2). Several additional cetacean species--blue whale (*Balaenoptera musculus*), killer whale (*Orcinus orca)*, and beluga (*Delphinapterus leucas*)--are occasionally sighted, but must be regarded as unusual or extralimital to the area. Likewise *Mesoplodon* spp. beaked whales and Curvier's beaked whale (*Ziphius cavirostris*) may be sighted along the southern edge of Georges Bank, but more normally occupy waters offshore of the shelf break. The pygmy sperm whale (*Kogia breviceps*) is known in these waters primarily from a few strandings, and is thought to be more common to the south and offshore.

Of the pinnipeds, harbor seal (*Phoca vitulina*) and gray seal (*Halichoerus grypus*) are year-round residents that migrate seasonally between Northern and Southern New England coastal waters. Harbor seal, particularly, use the region for feeding, breeding, and pupping (Gilbert and Stein 1981; Katona *et al.* 1993). Three additional pinnipeds--harp seal (*P. groenlandica*), hooded seal (*Cystophora cristata*), and ringed seal (*P. hispida*) --while common in Canadian waters, appear rarely in New England waters and normally only in winter.

As of December 1994, six Northeast cetaceans are listed as endangered under the federal Endangered Species Act (ESA): northern right whale (*Eubalaena glacialis*), humpback (*Megaptera novaeangliae*), blue whale, fin whale (*B. physalus*), sei whale (*B. borealis*), and sperm whale (*Physeter catodon*). As of March 1994, harbor porpoise (*Phocoena phocoena*) have been proposed to be listed as threatened by the National Marine Fisheries Service (NMFS). A final decision is pending. Critical habitats for northern right whale were designated in July 1994 for the Great South Channel and Cape Cod Bay areas. Information on these habitats is summarized in Kraus and Kenney (1991).

In general, distribution and abundance of the more common cetaceans can be described as an ebb and flow with the seasons--an ebb and flow that can almost solely be linked to food. Katona and Whitehead (1988) describe cetacean distributions as far from regular or random; such distributions are almost always where the food is. As a general pattern, in spring, cetaceans "flow" into the region, appearing on feeding grounds through summer and into fall. However, some individuals of most species appear to occupy the area through winter, with, in broadest terms, perhaps 20% of the warm-season abundance present in the cold season. Depending on the species, this cold-season estimate varies from 0 to 38% of the warm-season estimate.

Table 2. Common species of cetaceans and pinnipeds found in New England shelf waters

Common Name	Scientific Name
Large Whales (length generally greater than 20 ft)	
Fin whale	*Balaenoptera physalus*
Minke whale	*B. acutorostrata*
Sei whale	*B. borealis*
Humpback	*Megaptera novaeangliae*
Northern right whale	*Eubalaena glacialis*
Sperm whale	*Physeter catodon*
Small Whales (length generally less than 20 ft)	
Harbor porpoise	*Phocoena phocoena*
Atlantic white-sided dolphin	*Lagenorhynchus acutus*
White-beaked dolphin	*L. albirostris*
Long-finned pilot whale	*Globicephala melaena*
Risso's dolphin	*Grampus griseus*
Bottlenose dolphin	*Tursiops truncatus*
Common dolphin	*Delphinus delphis*
Atlantic spotted dolphin	*Stenella frontalis*
Striped dolphin	*S. coeruleoalba*
Pinnipeds	
Harbor seal	*Phoca vitulina*
Gray seal	*Halichoerus grypus*

The ebb and flow of marine mammals in New England waters are closely linked to occurrence in other areas: the shelf waters of the Mid-Atlantic and southeastern United States, portions of the West Indies, offshore waters over the continental slope and basin, and Canadian waters of the Scotian Shelf and northward. While some migration patterns have been described, the whereabouts and habitats of whales while absent from New England waters generally remain unknown. This is particularly true for wintering and calving grounds for most species.

[5] National Marine Fisheries Service, Woods Hole, MA.
[6] *id.*

Any treatment of marine mammals in New England waters must address fluidity and variability of the animals' occurrence. Certain feeding ground locations are occupied both by individuals that are "resident" for periods of weeks or months, as well as by individuals that appear transitory and pass through in a day or days. On the broader scale, it often seems that shortly after a "pattern" is established, a "shift" occurs. A given species, or collection of species, may be in an area for a number of days, and disappear completely on a following day. The same appears true on a decadal scale. Whales may seem to establish a pattern in given months or years, for several periods running, then markedly decrease or disappear in a following month or year. For example, a major "biomass flip" has been described for the Gulf of Maine/Georges Bank region twice in the last several decades (Fogarty *et al.* 1991; Sherman 1989). These changes in abundance of prey species have resulted in corresponding changes in distribution of whales. This has been perhaps most evident for humpbacks, and is described below.

DATA SOURCES

Surveys

With few exceptions (detailed below), the data on which present knowledge is based are now a decade old. The last and only large-scale survey for cetaceans was conducted by the Cetacean and Turtle Assessment Program (CETAP) at the University of Rhode Island from 1979 through 1982. During 39 mo of field studies, 11,156 sightings of 170,012 individuals were recorded (Cetacean and Turtle Assessment Program 1982). Aerial surveys were a major data source for this program. The data are archived at the University of Rhode Island as well as at the NEFSC. A number of publications have resulted from CETAP data, including Hain *et al.* (1992), Winn *et al.* (1987), Kenney and Winn (1986), and Kenney (1990). The CETAP data remain the most comprehensive data set to date. Even so, despite the scale of the project, sighting effort was modest, with an average of 5% coverage of the area for any particular seasonal survey. There was likewise geographic, annual, and seasonal variability. Therefore, while a considerable advance in knowledge resulted, the characterization of cetaceans in the area was not complete.

A second, somewhat smaller, areawide data set exists. From 1980 to 1986, the Manomet Bird Observatory of Manomet, Massachusetts, conducted cetacean, sea bird, and sea turtle sighting surveys during 124 NEFSC fisheries research surveys in shelf waters of the northeastern United States. The resulting data set included 1,357 sightings of large cetaceans and 1,604 sightings of small cetaceans (Smith *et al.* 1988). These data are archived at the NEFSC.

Several NEFSC surveys have added additional data. A large-scale aerial survey was conducted from August through October 1991. Shipboard and aerial surveys were conducted for harbor porpoise during 1990-93, and shipboard surveys were conducted for pelagic species over the shelf edge and slope region during the same period. Some analyses are in progress. Other analyses have been completed and are reported below.

Three species--humpback, northern right whale, and harbor porpoise--are exceptions to the general lack of recent data, and have been the focus of major research efforts since the CETAP concluded in 1982. As a result, there is considerable new information. First, the northern right whale is the most endangered whale in the western North Atlantic, with a population of about 300 individuals. The considerable information acquired in the last decade is summarized in proceedings of a 1992 workshop (Hain 1992) and in a research review in 1994 (Marine Mammal Investigation 1995). Data sets are reposited at the New England Aquarium in Boston, Massachusetts, the University of Rhode Island's Narragansett Bay Campus, and the NEFSC. Second, the humpback has been the subject of a large-scale, whole-ocean (North Atlantic) study beginning in January 1992, named the Years of the North Atlantic Humpback Program (YONAH). Results will comprehensively describe biology of this population (Allen *et al.* 1994). Data sets are reposited at the Center for Coastal Studies in Provincetown, Massachusetts, and the College of the Atlantic in Bar Harbor, Maine. Third, because of sink gillnet fishery interaction with harbor porpoise in both U.S. and Canadian waters, research efforts on this species have increased. Results are summarized in three recent workshop reports (Anonymous 1992; Smith *et al.* 1993; Palka 1994). Data are at the NEFSC and the Woods Hole Oceanographic Institution in Woods Hole, Massachusetts.

Three more restricted data sources exist. First, data are collected in localized research efforts from some dedicated vessels, but primarily by naturalists aboard whalewatching cruises. These data are located at the Center for Coastal Studies, the Cetacean Research Unit in Gloucester, Massachusetts, the Atlantic Cetacean Research Center in Gloucester, Massachusetts, the Whale Research Fund in Cape Neddick, Maine, the College of the Atlantic, and elsewhere. Secondly, under provisions of the Marine Mammal Protection Act of 1972 and the MFCMA, observers have been placed aboard fishing vessels, both foreign and domestic. In recent years, emphasis has been on domestic vessels. The Sea Sampling Program at the NEFSC uses observers to collect data on incidental takes of protected species in a variety of fisheries. This has provided biological data on marine mammals taken as bycatch in fisheries operations. Standard measurements and samples are collected, and in some cases, body parts or whole animals are retained by seagoing observers and are transported to the NEFSC for further study. Third, data on beached or stranded marine mammals are collected and archived. These latter two data sources (bycatch and strandings) are described below.

Seal populations are counted, typically by aerial survey over coastal haulout sites. Harbor seal surveys in the Gulf of Maine were conducted by J.R. Gilbert and colleagues at the University of Maine in 1981, 1982, 1986, and 1993, during pupping and molting periods in May and June. Also, because some portion of the population winters in Southern New England waters, surveys during January and February were conducted off Massachusetts and New Hampshire annually during 1983-87, and south to Rhode Island and Connecticut during 1986 and

Table 3. Summary of reported cetacean and pinniped strandings for the New England area, 1968-93

Common Name	Scientific Name	Strandings Total (1968-1993)	Strandings Avg. (1989-1993)
	Cetaceans		
Harbor porpoise	*Phocoena phocoena*	326	17
Atlantic white-sided dolphin	*Lagenorhynchus acutus*	348	15
Long-finned pilot whale	*Globicephala melaena*	515	36
Risso's dolphin	*Grampus griseus*	18	1
Common dolphin	*Delphinus delphis*	85	6
Dwarf sperm whale	*Kogia breviceps*	15	0
Striped dolphin	*Stenella coeruleoalba*	33	2
Humpback	*Megaptera novaeangliae*	28	2
Northern right whale	*Eubalaena glacialis*	4	0
Minke whale	*Balaenoptera acutorostrata*	49	4
Bottlenose dolphin	*Tursiops truncatus*	10	1
Miscellaneous		16	0
	Pinnipeds		
Harbor seal	*Phoca vitulina*	2,131	119
Gray seal	*Halichoerus grypus*	79	11
Hooded seal	*Cystophora cristata*	45	7
Harp seal	*P. groenlandica*	43	8
Ringed seal	*P. hispida*	5	1

Source: New England Aquarium, Boston, MA.

1987, by P.M. Payne and colleagues at the Manomet Bird Observatory. Studies of gray seal in a winter breeding colony on Muskeget Island, west of Nantucket Island, Massachusetts, have been conducted since 1966 by V. Rough of Spruce Head, Maine.

Strandings

Marine mammals are known to drive themselves ashore from all oceans of the world, and the coast of New England is not excluded. Marine mammals strand as singles, pairs, or in large numbers. To maximize the scientific value of beached and stranded marine mammals, efforts were initiated in the late 1970s to upgrade and formalize marine mammal stranding networks in the United States (Hofman 1991). Goals were to: (1) provide coordination; (2) establish scientific protocols for transport, release, euthanasia, specimen requests, and disposal of carcasses as appropriate; (3) clearly describe and evaluate data collection; and (4) develop and maintain tissue sample inventories and lists of all network-authorized institutions and individuals. Six regional network offices in the United States and one in Canada have been established. The Northeast Regional Marine Mammal Stranding Network is collaborative, responding to strandings from the Canadian border through Virginia. In the New England area, the College of the Atlantic and the New England Aquarium serve as centerpoints for strandings.

A review of activities during 1977-89 is provided by Early and McKenzie (1991), and a summary of strandings during 1968-93 is provided in Table 3. For New England waters, harbor seal is the most commonly beached pinniped. Long-finned pilot whale (*Globicephala melaena*), white-sided dolphin (*Lagenorhynchus acutus*), and harbor porpoise are the most commonly stranded cetaceans (Table 3). The National Museum of Natural History at the Smithsonian Institution acts as a repository for specimen material and for all cetacean stranding records.

Stranding events provide an opportunity for scientists to collect specimens that are used to aid in determination of various biological parameters. Much of what is known about life history and biology in general of marine mammals today is directly linked to strandings (Geraci and St. Aubin 1979). However, although stranded animals are a source of physiological and anatomical data, S.A. Rommel (pers. comm.[7]) cautions that stranded animals are typically in poor health and are usually not representative of the population as a whole.

HUMAN EFFECTS

Direct Effects

Human effects on marine mammals are either direct or indirect. Direct effects are primarily net entanglement, ship strikes, and fisheries bycatch. Indirect effects include habitat degradation and competition for fisheries resources.

Direct effects can be widespread. As an example, Kraus (1990) estimated that 32% of northern right whale mortalities were caused by either ship strike or net entanglement. Further, 57% of northern right whale have scars indicative of entanglements with fishing gear, and 7% have propeller scars. Likewise, there are an average of 4-6 reported entanglements of humpback per year, with additional reports of ship-collision scars (D.L. DeKing, pers. comm.[8]). Minke (*B. acutorostrata*) and fin whales are also subject to entanglements and occasional ship strikes (unpubl. data[9]). There is also a fisheries take or bycatch for several species. The incidental take of harbor porpoise, seals, and Atlantic white-sided dolphin has been documented in New England gillnet fisheries for demersal fishes (Gilbert and Wynne 1985; International Whaling Commission, in press; Anonymous 1992). In offshore areas, common dolphin (*Delphinus delphis*), long-finned pilot whale, and other species have been, or are, taken in various fisheries.

7 Eckerd College, St. Petersburg, FL.

8 Center for Coastal Studies, Provincetown, MA.

9 NMFS entanglement data base, Northeast Regional Operations Office, Gloucester, MA.

During the past decade, fisheries observers and researchers documented the incidental take or entanglement of 16 marine mammal species in commercial fishing operations in New England shelf and shelf-edge waters (Waring *et al.* 1990; NMFS 1991; International Whaling Commission, in press; Smith *et al.* 1993; Gilbert and Wynne 1987). These fisheries fall generally into three categories: (1) mobile gear (*i.e.*, trawls, dredges, purse seines); (2) fixed gear (*i.e.*, sink gill nets, stop seines, long lines); and (3) drift gill nets. Whereas most interactions and mortalities involved small dolphins, harbor porpoise, and seals, incidental mortalities in three endangered species (northern right whale, humpback, and sperm whale) have also been recorded (unpubl. data[10]; Right Whale Recovery Team 1991; Humpback Whale Recovery Team 1991). (Note that with the exception of the harbor porpoise take in the Gulf of Maine sink gillnet fishery, described below, the fisheries bycatch as presented here includes all shelf waters from North Carolina to eastern Georges Bank, and has not been subsetted specifically for New England waters.)

In the Gulf of Maine, most marine mammal - fisheries interactions occur in sink gillnet fisheries, although entanglements have been also documented in lobster pot, weir, purse seine, tub trawl and bottom trawl gear (Gilbert and Wynne 1985; Smith *et al.* 1993; unpubl. data[11]). As described previously, the principal take is of harbor porpoise. Since this take in U.S. Gulf of Maine and Canadian Bay of Fundy sink gillnet fisheries, relative to population size, may be biologically significant (Anonymous 1992; Smith *et al.* 1993), the NMFS has been petitioned to designate this species as threatened under ESA. Aside from harbor porpoise, domestic fisheries observers have also recorded the incidental mortality of harbor seal, gray seal, and Atlantic white-sided dolphin in Gulf of Maine sink gillnet fishing operations (International Whaling Commission, in press; unpubl. data[12]). Further, entanglements of larger cetaceans, particularly of humpback and northern right whale, have been documented (Right Whale Recovery Team 1991; Humpback Whale Recovery Team 1991).

In offshore waters, bycatch has occurred in foreign and U.S. joint-venture squid and Atlantic mackerel trawl fisheries, and U.S. pelagic drift gillnet, longline, and pair trawl fisheries (Waring *et al.* 1990; National Marine Fisheries Service 1991; International Whaling Commission, in press; P.A. Gerrior, pers. comm.[13]). Although long-finned pilot whale and common dolphin were the species most frequently taken in foreign squid and mackerel trawl fisheries, Atlantic white-sided dolphin, bottlenose dolphin (*Tursiops truncata*), and Risso's dolphin (*Grampus griseus*) were also occasionally taken (Waring *et al.* 1990; National Marine Fisheries Service 1991; Fairfield *et al.*, in press). Foreign squid and mackerel fishing activity ended in December 1986 and 1991, respectively (Waring *et al.* 1990; P.A. Gerrior, pers. comm.[14]).

In the recently developed U.S. pelagic drift gillnet and pair trawl fisheries, several species of pelagic dolphins and beaked whales account for most of the bycatch mortality (International Whaling Commission, in press; unpubl. data[15]). These include common dolphin, bottlenose dolphin, long-finned pilot whale, grampus, striped dolphin (*Stenella coeruleoalba*), and beaked whales. Further, documented mortalities of a sperm whale and a humpback in pelagic gillnet gear, and a single entanglement of a northern right whale (released alive but injured) in July 1993 along the southern edge of Georges Bank, have been recorded (unpubl. data[16]). Fisheries observers have also reported incidental takes of small cetaceans and large whales in pelagic longline gear. However, most of the animals were released alive (Waring *et al.* 1990; unpubl. data[17]).

Currently, marine mammal - fisheries interactions are governed by the 1994 amendments to the Marine Mammal Protection Act. The regime that resulted retains the act's goal of reducing mortality and serious injury to/of marine mammals incidental to commercial fisheries to insignificant levels approaching a zero rate. However, it additionally establishes a 7-yr time frame by which such reductions should occur. The Secretary of Commerce is required to prepare and implement take-reduction plans, and to promulgate regulations as needed to reduce the level of take. Because a significant reduction in current bycatch levels will likely be required, additional restrictions on New England commercial fisheries will likely be imposed.

Indirect Effects

Indirect human effects on marine mammals in New England waters due to habitat degradation presently focus on the proposed Massachusetts Water Resources Authority sewer outfall. Discharge of primary-treated sewage at the end of a 9.5-mi effluent outfall tunnel terminating in about 100 ft of water in Massachusetts Bay was scheduled to begin in June 1995, but is now experiencing delays. The secondary treatment of the effluent is scheduled to be phased in over a following period. While the NMFS "Biological Opinion" held that the proposed action would not jeopardize whales in the area, continued studies and monitoring activities were recommended. Discussion and review of the project are ongoing.

Additional indirect effects may result from dredging, harbor construction, and competition for fisheries resources. In some cases, competition for fisheries resources may be linked to commercial overexploitation of the resource, which in turn may affect health and vitality of some marine mammal populations, shifts in habitat use, prey selection, and recovery of endangered species.

INDIVIDUAL SPECIES STATUS

The species accounts that follow emphasize the more common species. These 17 species were selected because they are likely to play a more important role in the New England shelf ecosystem, interact with commercial fisheries, and experience human effects. These species can be grouped as being primarily

[10-17] *id.*

on-shelf or shelf-edge, although this designation is not absolute (Hain *et al.* 1985; Winn *et al.* 1987). The on-shelf species are northern right, humpback, fin, and minke whales, harbor porpoise, Atlantic white-sided and white-beaked dolphins, and harbor and gray seals. The shelf-edge species are sperm, sei, and long-finned pilot whales, and Risso's, bottlenose, Atlantic striped, spotted, and common dolphins.

Northern Right Whale

The northern right whale is the most endangered large whale in the world. The western North Atlantic population numbers about 300, with perhaps an average of 11 calves born per year. Individuals of this population range from wintering and calving grounds in coastal waters of the southeastern United States to summer feeding, nursery, and mating grounds in New England waters and northward to the Bay of Fundy and the Scotian Shelf. Recently, Knowlton *et al.* (1992) reported several long-distance movements as far north as Newfoundland and the Labrador Basin, suggesting an extended range for at least some individuals, and perhaps other habitat areas not presently well described.

Cornerstones of northern right whale research are the Right Whale Catalog, containing photographs of identified individuals, maintained at the New England Aquarium, and the centralized data base of sighting records, environmental data, and survey tracks maintained at the University of Rhode Island. Research results suggest five major habitats or congregation areas (*i.e.*, southeastern United States coastal waters, Great South Channel, Cape Cod Bay, Bay of Fundy, and Scotian Shelf) for western North Atlantic right whale. However, movements within habitats and within regions may be more extensive than previously thought. Daily tracking of several individuals using satellite tags suggests that local sightings separated by perhaps 2 wk should not be assumed to indicate a stationary or resident animal (Mate *et al.* 1992). Instead, data have shown lengthy and distant excursions--casting new light on movements and habitat use, and raising questions about the purpose or strategies underlying these excursions.

Genetic analyses of tissue samples are proving productive. Schaeff *et al.* (1993) have suggested that the present northern right whale population may be based on three matrilines. However, one of these lines may have its summer nursery and feeding grounds at a presently unknown site. A related question is where approximately 85% of the population (those other than calving females and a few juveniles) overwinter. One or more major wintering grounds have yet to be described.

The Final Recovery Plan for the Northern Right Whale was released in March 1992 (Right Whale Recovery Team 1991). A science and management meeting, coordinated by the NEFSC, was held in Silver Spring, Maryland, on 14 and 15 April 1992. The document of meeting proceedings contains summaries of research and recommendations for implementation (Hain 1992). A research review meeting in October 1994 (Marine Mammal Investigation 1995) updated the research status and recommendations. The best available information suggests that the western North Atlantic right whale population may be recovering at an annual rate of 2.5%, a growth rate that is less than that of the four stocks of southern hemisphere right whale (Knowlton *et al.* 1994). Human effects (*e.g.*, ship strikes, net entanglements) may be a major impediment to recovery. Also, inbreeding due to small population size may affect viability and decrease reproductive success. Because of the small population size, there is a concern that a precipitous event (*e.g.*, sudden increase in human effects, die-off as has occurred in other species) could negate what population growth may have occurred, and threaten the species with extinction.

The principal factors retarding growth and perhaps recovery of the population are ship strikes and net entanglement. Some 60% of the total population is likely affected. Marks or scars from entanglement with fishing gear were reported from 57% of northern right whale, and 32% of known mortalities are due to human activities. Young animals, ages 0-4, are apparently the most affected portion of the population. In this age group, 20-30% of mortality is due to ship strikes (Kraus 1990). An awareness and mitigation program, involving 10 agencies and organizations, was begun in 1992 for the coastal waters of northeast Florida. This program has been upgraded and expanded for the 1993-94 and 1994-95 seasons. This effort will likely serve as a model for other areas that may include New England waters. An additional present concern is the possibility of habitat degradation in Massachusetts and Cape Cod Bays due to a Boston sewage outfall pipe now under construction.

As with other cetaceans, New England waters are a primary feeding habitat for northern right whale. Northern right whale in this area appear to feed primarily on calanoid copepods. Analyses suggest that northern right whale must locate and exploit extremely dense patches of zooplankton to feed efficiently. These dense zooplankton patches are likely a primary characteristic of the spring, summer, and fall northern right whale habitat (Kenney *et al.* 1986; Mayo and Marx 1990). During the peak feeding season in Cape Cod and Massachusetts Bays, the acceptable surface copepod resource is limited to perhaps 3% of the region (Mayo and Goldman, in press). While feeding in coastal waters off Massachusetts has been better studied, feeding by northern right whale has been observed elsewhere over Georges Bank, in the Gulf of Maine, in the Bay of Fundy, and over the Scotian Shelf. New England waters also serve as a nursery for calves, and in some cases, for mating.

Humpback

The humpback is endangered. Population estimates for the western North Atlantic population are on the order of 5,100 individuals. In winter, humpback from all western North Atlantic feeding areas migrate to the West Indies where courtship, breeding, and calving occur. The majority (85%) are found on Silver and Navidad Banks off the north coast of the

Dominican Republic. The remainder are scattered in Samana Bay (Dominican Republic), along the northwest coast of Puerto Rico, through the Virgin Islands, and along the eastern Antilles chain south to Venezuela (Katona and Beard 1990).

During summer, there are at least five geographically distinct feeding aggregations from latitudes 42°N to 78°N. These areas are (with approximate aggregation sizes in parentheses): Gulf of Maine (400), Gulf of St. Lawrence (200), Newfoundland and Labrador (2,500), western Greenland (350), and the Denmark Strait (up to 2,000) (Katona and Beard 1990). During 1978-82, the abundance estimate for the Gulf of Maine during spring was 555 (±735, 95% confidence interval), and for Georges Bank it was 146 (±229, 95% confidence interval). For the Gulf of Maine, the summer abundance estimate is 61%, fall estimate is 11%, and winter estimate is 0% of the spring peak-abundance estimate; while for Georges Bank, the summer abundance estimate is 51%, fall estimate is 164%, and winter estimate is 0% of the spring value. These values are dive-time corrected for submerged animals not sighted from the survey aircraft, using a scale-up factor of 3.6 (Cetacean and Turtle Assessment Program 1982). The best available data suggest an annual rate of increase of 9% for the North Atlantic humpback (Katona and Beard 1990). There is uncertainty, however, about the abundance estimates as well as the rate of increase.

In a long-term study of identified humpbacks in waters off Cape Cod, Clapham *et al.* (1993) reported a high degree of site fidelity, both within years and between years. For this same area, some reproductive parameters are known. The mean birth rate for 1979-87 was 8% (2% standard deviation) with no significant year-to-year differences. The calving interval was 2.35 yr (0.70 standard deviation). The average age at attainment of sexual maturity for males and females was 5 yr (Clapham and Mayo 1990; Clapham 1992).

The principal activity for humpbacks in New England waters is feeding. In these waters, they are believed to be largely piscivorous, feeding on Atlantic herring (*Clupea harengus*), northern sand lance (*Ammodytes dubius*), and others. The distribution of humpbacks has been largely correlated to prey species and abundance, although behavior and bottom topography are factors in foraging strategy (Payne *et al.* 1986; Payne *et al.* 1990). In the mid 1970s, a "biomass flip" occurred, where commercial depletion of Atlantic herring and Atlantic mackerel (*Scomber scombrus*) led to a concurrent increase in northern sand lance in the southwestern Gulf of Maine. For much of the late 1970s and early 1980s, humpbacks were densest over the sandy shoals in the southwestern Gulf of Maine favored by the northern sand lance, and humpback distribution appeared shifted to this area (Payne *et al.* 1986). At the same time, humpback occurrence decreased in the northern Gulf of Maine. In the mid-1980s, a second biomass flip began, and a gradual return of Atlantic herring and Atlantic mackerel corresponded with a decrease in northern sand lance (Fogarty *et al.* 1991). In 1992-93, humpbacks in the northern Gulf of Maine have increased dramatically, along with a major influx of Atlantic herring (T. Fernald, pers. comm.[18]). Likewise, in the 1992-93 summer distribution, humpbacks were few in nearshore Massachusetts waters, and were more abundant in the offshore waters of Cultivator Shoals and the Northeast Peak of Georges Bank, and on Jeffreys Ledge--where more traditional Atlantic herring areas are located (D.K. Mattila, pers. comm.[19]).

A vigorous program of humpback research has taken place in the western North Atlantic since the mid-1970s. This program has been conducted through various organizations such as the NEFSC, Center for Coastal Studies, College of the Atlantic, New England Aquarium, Woods Hole Oceanographic Institution, and University of Rhode Island. A cornerstone of humpback research is the identification of individuals using photographs of the flukes. The catalog and computerized data base are maintained at the College of the Atlantic. Presently, the catalog contains photographs of about 4,000 individually identified whales.

In early 1992, a major new research initiative--YONAH (mentioned earlier)--was begun. This project, planned to run for several years, is a large-scale, intensive, oceanwide study of humpback throughout their entire North Atlantic range. Photographs for individual identification, and biopsy samples for genetic analyses, have been collected both from summer feeding areas in the more northerly latitudes and from breeding grounds in the West Indies. Data will be analyzed to determine the population status and genetic relationships of humpback throughout their range. The project is being supported by the United States (through the NEFSC), Canada, Greenland, Iceland, Norway, and Great Britain.

An early 1992 development was the reporting of an increased number of sightings of young humpbacks in the Mid-Atlantic region--in the vicinity of the Chesapeake and Delaware Bays (Swingle *et al.* 1993). Whether this is in fact a distributional change or is simply due to an increase in sighting effort is unknown at present.

The "Final Recovery Plan for the Humpback Whale, (*Megaptera novaeangliae*)" was released in March 1992 (Humpback Whale Recovery Team 1991). The plan recommends actions that will double population size during the next 20 yr. The objectives are: (1) maintain and possibly enhance habitat; (2) identify and reduce human-related mortality, injury, and disturbance; (3) continue to measure and monitor key population parameters to determine if actions are successful; and (4) improve administration and coordination of the recovery effort. Implementation plans are under development.

As described previously, humpback occassionally become entangled in fishing gear, and some bear wounds from ship collisions. In addition, several recent mortalities are noteworthy. Between November 1987 and January 1988, 14 humpback died after consuming Atlantic mackerel containing a dinoflagellate saxitoxin. The whales subsequently stranded in the vicinity of Cape Cod Bay and Nantucket Sound. During the first 6 mo of 1990, seven juvenile (25-30 ft long) humpback stranded, with no causitive agent apparent, between North Carolina and New Jersey (National Marine Fisheries Service 1991). These die-offs are a cause for some concern.

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[18] College of the Atlantic, Bar Harbor, ME.

[19] Center for Coastal Studies, Provincetown, MA.

Fin Whale

The fin whale is endangered. The species compromised 46% of the large whales and 24% of all cetaceans sighted over the continental shelf between Cape Hatteras and Nova Scotia during 1978-82. The abundance of fin whale in this area is estimated to range from 1,500 (winter low) to 5,000 (spring/summer high). It is possible that, given a 4-10% annual rate of increase (as has been found for several other unhunted populations), the numbers of fin whale may have increased substantially since the data on which these estimates are based were collected (Hain *et al.* 1992). This is consistent with Agler *et al.*, (1993), who, based on photographically identified fin whale, estimated gross annual reproduction at 8%, with a mean birthing interval of 2.7 yr.

For New England waters, based on 1978-82 data, the population estimate for the spring season on Georges Bank was 283, and in the Gulf of Maine was 370 (total=653). With a dive-time correction of 4.86 applied, these estimates become 1,375 and 1,798, respectively (total=3,173) (Hain *et al.* 1992). There is a good deal of uncertainty with the estimates (the variances are large, often equal to the estimate itself), as well as with the dive-time correction for submerged animals that may have been unsighted by the survey aircraft.

There is little doubt that New England waters constitute major feeding grounds for fin whale. There is evidence of site fidelity by females, and perhaps of some substock separation on the feeding range (Agler *et al.* 1993). Seipt *et al.* (1990) reported that 49% of identified fin whale on Massachusetts Bay feeding grounds were resighted within years, and 45% were sighted between years. While recognizing localized as well as more extensive movements, these authors suggested that the fin whale on these grounds exhibited patterns of seasonal occurrence and annual return that are in some respects similar to those shown by humpback.

Based on an analysis of neonate stranding data, Hain *et al.* (1992) suggested that calving takes place during approximately 4 mo from October through January in latitudes of the U.S. Mid-Atlantic region. It is unknown with any greater accuracy, however, where calving, mating, and wintering for the population occur.

While much is unknown, the magnitude of the role of the fin whale is impressive. In New England shelf waters, fin whale is the dominant cetacean species in all seasons. It has the largest standing stock, the largest food requirements, and the largest effect on the ecosystem of any cetacean species (Hain *et al.* 1992).

Minke Whale

The minke whale is the thirdmost common large whale in the area (Cetacean and Turtle Assessment Program 1982). Like fin whale, they are commonly and widely distributed. However, because of its smaller size, more rapid movements, and less observable behavior, there is more uncertainty about abundance, distribution, and behavior.

There is a strong seasonal component to minke whale distribution in the area. Spring and summer are times of relatively widespread and common occurrence. In fall, the number of individuals, as well as the areas they occupy, are reduced. In winter, the species appears to be largely absent from the area. Like most other baleen whales, minke whale generally occupy the shelf proper, rather than the shelf-edge region. As described, minke whale do occasionally become entangled in fishing gear in the Gulf of Maine. Although some mortalities have been documented, in most instances the animals are released alive.

Abundance of minke whale in the Gulf of Maine and Georges Bank was estimated at 225 (± 160, 95% confidence interval) for the spring season. Summer abundance was 55% of this level, fall abundance was 43%, and winter abundance was 0%. These estimates, based on CETAP's 1978-82 surveys, are not corrected for submerged animals not sighted from the survey aircraft.

Sei Whale

The sei whale is endangered. Indications are that the sei whale population is centered in more northerly waters, perhaps on the Scotian Shelf (Mitchell and Chapman 1977). As such, the northern portions of the Gulf of Maine and Georges Bank are at the southern part of the species' range. In these more northerly sections, sei whale have a predominantly spring and summer distribution. The period of greatest abundance is in spring, with sightings concentrated along the eastern margin of Georges Bank, into the Northeast Channel area, and along the southwestern edge of Georges Bank in the area of Hydrographer Canyon (Cetacean and Turtle Assessment Program 1982). Of the baleen whales, the sei whale is generally found in the deeper waters characteristic of the shelf-edge region (Hain *et al.* 1985).

This general pattern of sei whale distribution, however, is disrupted during episodic incursions into more shallow and inshore waters. Unlike the more piscivorous baleen whales, the sei whale, like the northern right whale, is largely planktivorous, feeding primarily on euphausiids and copepods. In years of copepod abundances, sei whale are reported in more inshore locations such as the Great South Channel and Stellwagen Bank areas (Kenney, in preparation; Payne *et al.* 1990) One recent influx to the southern Gulf of Maine occurred in summer 1986 (Schilling *et al.* 1992).

Abundance is in the hundreds at peak times. For the band of water ringing the eastern and southern edge of Georges Bank (between the 100- and 2,000-m depth contour) in spring, the abundance estimate is 253 (±321, 95% confidence interval). There may be tens of sei whale in the somewhat shallower waters of the Gulf of Maine and Georges Bank. These estimates are based on 1978-82 CETAP data). Abundance in the summer season is 45% of the spring level, in fall it is 31%, and in winter it is 26%. These estimates are not corrected for submerged animals that may have gone unsighted by the aerial surveys.

Because of its more offshore distribution and generally lower abundance, there are few if any data on fisheries interactions or human effects. In November 1994, however, a sei whale was reported impaled in the bow of a ship arriving in Boston Harbor, indicating that human effects do occur to this species.

Sperm Whale

The sperm whale is endangered. The species generally does not occur in the Gulf of Maine or the Georges Bank area. However, in spring and summer, the species occurs in the shelf-edge region of eastern and southern Georges Bank (Cetacean and Turtle Assessment Program 1982; Hain *et al.* 1985; unpubl. data[20]). The abundance estimate for the area between the 100- and 2,000-m depth contours along the southern margin of Georges Bank is 57 ± 111 during peak abundance in summer. Spring abundance was 70% of this level, winter was 37%, and fall was 0%. Estimates based on CETAP's 1978-82 surveys are not corrected for submerged animals not sighted from the survey aircraft.

Atlantic White-sided Dolphin

The Atlantic white-sided dolphin is the dolphin most often sighted in shelf waters of the New England region. This is in contrast to most other dolphins that are more commonly associated with the more offshore shelf-edge region. The sightings of this species are broadly distributed, and in 25% of cases, occur in association with fin whale, humpback, and/or other species. The mean group size for 584 sightings was 54 (Cetacean and Turtle Assessment Program 1982).

Life history characteristics for this species in New England waters were described based on stranded animals in 1973-74. Mature animals are 2.0-2.5 m (6.6-8.2 ft) in length. Age at maturation ranges from 6 to 12 yr. The birth season is May to August, with a peak in June and July, following a gestation period of 11 mo. Prey species include squids, rainbow smelt (*Osmerus mordax*), hakes, shrimps, and most likely, Atlantic herring (Sergeant *et al.* 1980).

Based on 1978-82 data, abundance estimates for spring were 23,979 (±10,242, 95% confidence interval) in the Gulf of Maine, and 14,838 (±15,433, 95% confidence interval) in the Georges Bank area. For the Gulf of Maine, summer values were 56%, fall values were 89%, and winter values were 19% of spring values. For Georges Bank, summer values were 78%, fall values were 68%, and winter values were 6% of spring values.

Incidental mortality (44 animals) was reported in foreign trawl fishing operations for Atlantic mackerel and squids. In recent years, observers have recorded an annual take of 6-8 animals in the Gulf of Maine sink gillnet fishery. The effect of this level of incidental mortality is unknown, but is believed to be negligible, relative to estimated population levels.

Since 1968, 348 Atlantic white-sided dolphins have stranded on the New England coast (Table 3), the second-highest stranding rate for any cetacean. Some specimens provided the basis for the Sergeant (1980) report. When the additional samples are worked up, valuable population and life history information will be added.

White-beaked Dolphin

The white-beaked dolphin (*L. albirostris*) likely has its population centered in Canadian waters of the Gulf of St. Lawrence and Newfoundland. Sightings are, however, reported in coastal waters from Cape Cod to northern Maine. The species is said to have been more common around Cape Cod during the 1950s than at present (Katona *et al.* 1993). Distribution, habits, and abundance in New England waters are poorly known.

Long-finned Pilot Whale

Long-finned pilot whale are distributed along the shelf edge at the southern margin of Georges Bank. The distribution, however, is broadly defined, and extends up over Georges Bank and into the Gulf of Maine (Cetacean and Turtle Assessment Program 1982).

Based on 1978-82 data, the spring abundance estimate was 581 (±1,254, 95% confidence interval) for the Gulf of Maine, and 4,754 (±5,190, 95% confidence interval) for the Georges Bank area. For the Gulf of Maine, summer values were 0%, fall values were 56%, and winter values were 0% of spring values. For Georges Bank, summer values were 106%, fall values were 11%, and winter values were 18% of spring values.

Observers have reported takes of long-finned pilot whale in foreign mackerel and squid trawl fisheries, as well as U.S. pelagic gear fisheries in shelf and slope waters off the northeastern United States (Waring *et al.* 1990; Fairfield *et al.*, in press; International Whaling Commission, in press). From 1977 to 1991, 409 long-finned pilot whale were killed in foreign fishing operations (Waring *et al.* 1990; unpubl. data[21]). The propensity of long-finned pilot whale to chase trawls during net retrieval operations may be a significant factor contributing to the high take (G.T. Waring, pers. comm.[22]). Since 1989, an additional 52 animals were recorded in pelagic fishing gear. There are no reports of bycatch in Gulf of Maine fisheries. As shown in Table 3, long-finned pilot whale are the most commonly stranded cetacean in New England waters, with an average of 36 per year coming ashore. Effect of this natural and incidental mortality on long-finned pilot whale is unknown.

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[20] National Marine Fisheries Service, Woods Hole, MA.

[21] *id.*

[22] *id.*

Common Dolphin

The common dolphin occurs in a broad band along and inward of the shelf edge in all four seasons. For a sample that included waters from Cape Hatteras to northeastern Georges Bank, the mean group size for 453 sightings was 55 (Cetacean and Turtle Assessment Program 1982).

Seasonal abundance of common dolphin appears different from most, with the greatest estimated abundance in fall and winter. Based on 1978-82 data, the species was absent in the Gulf of Maine except for winter when the abundance estimate was 591 (±2,453, 95% confidence interval). In the Georges Bank area, the fall abundance estimate was 20,858 (±47,793, 95% confidence interval). The spring estimate was 14%, summer estimate was 2%, and winter estimate was 17% of the fall estimate. As described, for most species in this report, estimates have typically been for spring/summer, as this is when abundances generally appear highest. In the case of the common dolphin, however, the peak appears shifted to later in the year.

Takes of this species are reported as bycatch in foreign mackerel and squid trawl fisheries, as well as in U.S. pelagic gear fisheries in the shelf and slope waters off the northeastern United States (Waring *et al.* 1990; International Whaling Commission, in press). During 1977-91, 226 takes were recorded in foreign mackerel and squid trawling operations. Most of these takes, however, were in the shelf-edge waters off the Midddle Atlantic Bight. Since 1989, 322 animals have been incidentally taken in pelagic gear fisheries (unpubl. data[23]). In addition, a few animals have been killed in Gulf of Maine sink gillnet fisheries (unpubl. data[24]). The size/sex composition of animals taken in these fisheries indicates that most were sexually immature (Waring *et al.* 1990; unpubl. data[25]). Effects of incidental mortality are unknown, but are believed to be negligible relative to likely population levels.

Striped Dolphin

This species is common south of New England, off the Mid-Atlantic region, and offshore of the shelf break. In warm seasons, however, large groups that may include several hundred animals, are sighted along the southwestern edge of Georges Bank and seaward of the shelf break (Cetacean and Turtle Assessment Program 1982; unpubl. data[26]). Abundance and distribution in New England waters are unknown. During 1989-93, observers recorded 19 entanglements in pelagic fishing gear (unpubl. data[27]). Effects of incidental mortality are unknown, but are believed to be negligible, as levels of take are low relative to likely abundance levels.

Atlantic Spotted Dolphin

The Atlantic spotted dolphin (*S. frontalis*) is commonly sighted off the Mid-Atlantic region and southward, as well as in offshore waters. In warm seasons, this species is also sighted along the southwestern edge of Georges Bank (Cetacean and Turtle Assessment Program 1982). Abundance and distribution in New England waters are unknown. During 1989-93, observers recorded 19 entanglements in pelagic fishing gear (unpubl. data[28]). Effects of incidental mortality are unknown, but are believed to be negligible, as levels of take are thought to be low relative to likely abundance levels.

Risso's Dolphin

Risso's dolphin occur along the southern edge of Georges Bank, primarily in summer and fall. These sightings appear to be at the northern margin of the species' range, however, as the species appears to be considerably more abundant along the shelf edge south of Martha's Vineyard and east of the Middle Atlantic Bight. For the Cape Hatteras to eastern Georges Bank area, the mean group size for 478 sightings was 17 (Cetacean and Turtle Assessment Program 1982). Based on 1978-82 data, the abundance estimate for all four seasons in the Gulf of Maine was 0. Like the common dolphin, peak abundance along the Georges Bank shelf edge was in the fall, when the estimate was 4,488 (±13,042, 95% confidence interval). The winter abundance estimate was 0. The spring value was 7%, and the summer value was 30%, of the fall value.

Observers reported four takes of Risso's dolphin in fishing activities in the shelf and slope waters off the northeastern United States (Waring *et al.* 1990; International Whaling Commission, in press). Effects of the incidental mortality are unknown, but are believed to be negligible.

Bottlenose Dolphin

Bottlenose dolphin traditionally have been considered inhabitants of shallow coastal waters. Based on recent data, most authors now concur that there are both coastal and offshore populations, which may be more or less isolated. It appears that larger, more robust animals are generally in the offshore stock. It is this stock that is the principal one in New England waters.

The bottlenose dolphin was the small cetacean sighted most commonly during the CETAP study. It is a significant component of the marine ecosystem of the northeastern U.S. continental shelf. Abundance estimates based on the CETAP data indicate that the northeastern U.S. population of this species contains approximately 10,000-13,000 individuals. The inshore stock may comprise only 3-4% of this value (Kenney 1990). These abundance estimates must be considered as minimum values, since they are not adjusted for animals missed due to submergence during overflight of the survey aircraft.

In the defined New England shelf waters, bottlenose dolphin are found primarily along the shelf break at the southern boundary of Georges Bank. Abundance is highest in summer when the estimate is 1,900 (±2,325, 95% confidence interval).

23-28 *id.*

The spring estimate is 87%, fall estimate is 28%, and winter estimate is 38% of the summer estimate.

The center of the bottlenose dolphin population is to the southwest, with higher abundances off New Jersey to Virginia, and extending south to Florida. In this area, the nearshore or coastal form is recognized as a separate stock that ranges seasonally as far north as Long Island, New York. During the 11-mo period from June 1987 to April 1988, a total of 742 stranded bottlenose dolphin were reported from New Jersey to the east coast of Florida. The mortality rate for this coastal Mid-Atlantic stock of may have produced a potential decline for this stock on the order of 50%, causing some concern (National Marine Fisheries Service 1991). Under the Marine Mammal Protection Act, NMFS listed this stock as depleted, and convened a workshop to discuss the status and management (Wang *et al.* 1994)

Bottlenose dolphin, principally the offshore form, is incidentally taken in several mid-shelf and shelf-edge fisheries (Waring *et al.* 1990; International Whaling Commission, in press; unpubl. data[29]). During the 1980s, nine animals were taken in foreign mackerel and squid trawl fisheries (Waring *et al.* 1990). From 1989 to 1993, observers recorded 61 mortalities in pelagic gillnet and pair trawl fisheries (unpubl. data[30]). Effects of this mortality are unknown.

Harbor Porpoise

The harbor porpoise is the smallest cetacean in the region, reaching about 1.5 m (5 ft) in length. The Northwest Atlantic harbor porpoise is found from Greenland to Florida. It is hypothesized that there are three subpopulations: Newfoundland, Gulf of St. Lawrence, and Gulf of Maine/Bay of Fundy (GOM/BOF). However, new information on the movements of individuals, a mitochondrial genetic study, and two abundance surveys have decreased confidence that the hypothetical three sub-populations are isolated from one another (Palka 1994). Little is known about the seasonal movements of this species, except for the presence of summer aggregations in the three aforementioned population areas. During spring and fall, at least part of the GOM/BOF subpopulation is in New England waters. During summer, a third to a half remains in U.S. waters while the majority is in Canadian waters of the Bay of Fundy and western Scotian Shelf (D.L. Palka, pers. comm.[31]). In mid-August, the GOM/BOF subpopulation aggregates in the lower Bay of Fundy around Grand Manan Island and along the coast of Maine east of Penobscot Bay. The best current abundance estimate of harbor porpoise in summer in the GOM/BOF (based on 1991-92 data) is 47,200 (range=39,500-70,600, 95% confidence interval) (Smith *et al.* 1993). Stranding, fisheries bycatch, and sighting survey data suggest that at least some animals migrate south of Cape Cod in winter, and that North Carolina may be the southern extent of the range (Anonymous 1992).

The maximum lifespan estimates range from 10 to 15 yr, although only a few animals older than 7 yr are seen. The estimated female age at sexual maturity is 3.0-3.4 yr, and the pregnancy rate is 86% per year (Read 1990a). The reproductive cycle of the GOM/BOF animals is seasonally synchronized. Ovulation and conception occur in late June, gestation is approximately 10.6-mo long, parturition occurs during mid-May, and lactation occurs for 9 mo. Many lactating females are pregnant, and many adult females calve every year (Read 1990b). Atlantic herring is the primary prey species for harbor porpoise in the Bay of Fundy during summer. Other prey species include silver hake and Atlantic cod (Recchia and Read 1989). To continue to increase knowledge on life history and food habits, during 1992-94, 44 harbor porpoise killed in gill nets in the Gulf of Maine were necropsied at the NEFSC. Results are in preparation (J.R. Nicolas, pers. comm.[32]).

The species is taken as bycatch in fisheries operations, particularly gill nets, throughout the region. Because of uncertainties in population structure as well as in estimates of abundance and bycatch, the ratio of bycatch to population size is presently unknown (Palka 1994). Actions to determine the status of harbor porpoise, as well as to reduce bycatch, are ongoing.

The GOM/BOF subpopulation was the subject of a petition for listing by NMFS as threatened under ESA in December 1991.

Harbor Seal

The Atlantic harbor seal population is found in the western North Atlantic from eastern Canadian Arctic islands south to New York (Katona *et al.* 1993). There is no information on stock structure. The harbor seal is a year-round inhabitant of coastal waters of eastern Canada and Maine (Katona *et al.* 1993), and occurs seasonally in Southern New England and New York waters from late September to May (Schneider and Payne 1983). A northward migration from Southern New England to Maine occurs prior to pupping season, which takes place from mid-May through June principally in the Blue Hill Bay region (Wilson 1978; Gilbert and Wynne 1983). In recent years, researchers have reported an increase in animals and haulout sites during winter in Southern New England, and an expansion of pupping sites in mid-Maine coastal waters (Payne and Selzer 1989; P.M. Payne, pers. comm.[33]; J.R. Gilbert, pers. comm.[34]). The largest aggregation of harbor seal in the northeastern United States occurs in mid-winter at Monomoy Island, Massachusetts, and adjacent shoals. A single high count of 1,672 seals occurred here in the mid-1980s (Payne and Selzer 1989).

Population estimates are obtained from counts made during aerial surveys of pupping ledges and overwintering sites (Gilbert and Stein 1981; Payne and Selzer 1989). The number of seals along the New England coast was 10,500 animals in 1981, more than double the population present when the Marine Mammal

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[29-32] *id.*

[33] National Marine Fisheries Service, Silver Spring, MD.

[34] University of Maine, Orono, ME.

Protection Act was implemented in 1972 (Payne and Schneider 1984; Gilbert and Stein 1981). This number is considered a minimum abundance estimate, as it does not include animals in the water. Aerial surveys conducted in 1986 and 1993 by the University of Maine indicated that the population is still increasing. The 1993 counts estimated 28,810 harbor seal, including 4,250 pups. This number is more than double the 1981 counts. (J.R. Gilbert, pers. comm.[35]).

During the mid-1980s, researchers at the University of Maine reported an average of 22 seals entangled annually in sink gill nets off the Maine coast. In recent years, fisheries observers recorded a similar level of entanglements (unpubl. data[36]). Since levels of take are low relative to abundance levels, the effect of incidental mortality is believed to be negligible. Furthermore, the capability of the population to withstand significant losses can be demonstrated by the fact that the harbor seal population increased over a period when more than 350 seals were found dead in the Cape Cod area from an in influenza outbreak (National Marine Fisheries Service 1991).

Gray Seal

In the western Atlantic, there is one stock that is centered in the Gulf of St. Lawrence (Katona *et al.* 1993; Davies 1957). This stock is separated by both geography and differences in the breeding season from the eastern Atlantic stock (Bonner 1981). The western Atlantic stock is distributed and breeds principally in eastern Canadian waters. However, small numbers of animals and pupping have been observed on several isolated islands along the Maine coast and in Nantucket-Vineyard Sound, Massachusetts (Katona *et al.* 1993; V. Rough, pers. comm.[37]; J.R. Gilbert, pers. comm.[38]). A winter breeding colony on Muskeget Island, west of Nantucket Island, may provide some measure of gray seal population trend and expansion. During the 1980s, as gray seal population and range expanded in eastern Canada, sightings in New England increased. In 1988, five pups were born at Muskeget. The number of pups increased to 12 in 1992, 30 in 1993, and 59 in 1994. Maximum counts of adults on Muskegat Island were obtained during the spring molt. This count did not exceed 13 in any year during the 1970s, but rose to 61 in 1984, 192 in 1988, 503 in 1992, and 1,549 in 1993. (V. Rough, pers. comm.[39]).

Estimates of the total western Atlantic population are not available. Since 1962, pup production on Sable Island, Nova Scotia, has been about 12% per year. The 1986 population estimate for individuals that are 1-yr old and older was between 100,000 and 130,000 animals (Stobo and Zwanenburg 1990). The population in U.S. waters has increased from about 30 in the early 1980s to 500-1,000 animals in 1993 (J.R. Gilbert, pers. comm.[40]). Furthermore, in March-April 1993, V. Rough (pers. comm.[41]) sighted 1,500 gray seal on Muskeget Island, the highest on record.

In recent years, observers have annually recorded a small number (2-3) of gray seal incidental takes in Gulf of Maine sink gillnet fisheries. In addition, V. Rough (pers. comm.[42]) has documented several animals with netting around their necks. Effects of incidental mortality are unknown, but are believed to be negligible, as levels of take are low relative to abundance levels.

STATUS AND HUMAN EFFECTS SUMMARY

One word largely explains occurrence of marine mammals in New England waters--food. New England waters represent a feeding habitat for more than a dozen species of marine mammals. The majority, including fin, minke, and humpback whales, are piscivorous--feeding mostly on Atlantic herring, Atlantic mackerel, northern sand lance, and other schooling fishes in this size range. Two species, right and sei whales, are planktivorous--feeding mostly on copepods. Two of the more offshore species, sperm and long-finned pilot whales, are teuthivorous--feeding primarily or secondarily on squid.

Occurrence of whales in these waters can be thought of in two levels--regular and predictable, and less regular and less predictable. At the first level, there is a pattern of migration into the area from wintering grounds, and some degree of site fidelity both within seasons and between years. At the second level, superimposed on the first, is the presence of some whales throughout winter, shifts in response to changes in distribution and abundance of prey species, and a general fluidity in behavior. While site fidelity sometimes fits a pattern, recent tagging studies by Mate *et al.* (1992) show that sometimes lengthy excursions can occur between two resightings in the same area. As this section was written, a major shift in distribution and prey species for many marine mammals in the area was apparently underway. In addition, increasing populations are indicated for at least several species--including fin and humpback whales and harbor and gray seals.

Human effects on marine mammals include direct and indirect. Direct effects are net entanglement, ship strikes, and fisheries bycatch. Indirect effects include commercial fisheries effects on prey species, alteration of habitat through pollution (chemical and acoustic), and perhaps introduction of disturbance and stress from vessel traffic. In 1994, many commercial fish stocks had become severely reduced. Efforts were underway to reduce fishing effort and assist stock recovery. How will these activities affect marine mammals?

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[35] *id.*
[36] National Marine Fisheries Service, Woods Hole, MA.
[37] 484 Clark Island Rd., Spruce Head, ME.
[38] University of Maine, Orono, ME.
[39] 484 Clark Island Rd., Spruce Head, ME.
[40] University of Maine, Orono, ME.
[41] 484 Clark Island Rd., Spruce Head, ME.
[42] *id.*

Marine mammals likewise affect humans. The whalewatching industry has economic as well as quality-of-life effects. But, what will be the effect of a distributional shift in whales (to more offshore and less accessible areas)? With regard to the fisheries, using a quite conservative application of abundance estimates in this report, during the peak spring/ summer feeding season, there may easily be 2,000 large whales, 50,000 small whales, and 28,000 seals in New England waters inshore of the shelf edge. Effects on fisheries resource may be substantial. While whales and man do not always compete directly (whales often feed at trophic levels below the level fished by man), the biomass of fishes taken by whales is likely comparable to that taken by man. As previously described, in the early 1970s, the commercial fisheries dramatically affected the prey species taken by whales. On the other hand, Sissenwine *et al.* (1984) speculated that in the mid- and late 1970s, fin whale contributed to demise of the already overfished Georges Bank stock of Atlantic herring. In 1994, what are the effects of marine mammals on fisheries resources? As managers endeavor to reduce commercial fishing effort and restore depleted fish stocks, the effects of the likely increasing marine mammal populations will require consideration. Yet, the size and effect of marine mammal stocks are poorly known.

Looking to the future, environmental and management decisions are enhanced by good information. In the case of marine mammals, it can be fairly said that, with few exceptions, glimpses, hints, and inferences are the norm. While some very good information is on hand or being acquired, there are substantial gaps. A major data source is naturalists aboard whalewatching vessels in fairly localized areas. While there are some limitations, good information is available. The very great majority of New England waters, however, go unsampled. In particular, offshore areas are poorly studied, and the community of sperm, long-finned pilot, and beaked whales, and Risso's, common, and striped dolphins, as well as others, is mostly undescribed. The first, last, and only major survey effort for the whole area--itself characterized by relatively low sighting effort and large variances--is now more than a decade past. Cataloging and data centers exist on somewhat precarious and unreliable funding. Some large and valuable data sets exist (for example, strandings data), but require workup. Questions exist about abundance, trends, vital rates, stock segregation, interchange with other areas (Scotian Shelf, Mid-Atlantic, offshore waters), and location of wintering grounds for most species. This is all in view of growing and changing marine mammal populations with a high level of direct and indirect human interaction, as well as legislation that requires conservation, management, and recovery. A re-energized, focused, cohesive, and decision-oriented marine mammal program, one that includes both short- and long-term goals, would therefore have merit.

OCEANOGRAPHIC PROCESSES AND BENTHIC SUBSTRATES: INFLUENCES ON DEMERSAL FISH HABITATS AND BENTHIC COMMUNITIES

David G. Mountain[43], Richard W. Langton[44], and Les Watling[45]

OCEANOGRAPHIC PROCESSES

The Gulf of Maine is bordered on the east, north, and west by the coasts of Nova Scotia, New Brunswick, Maine, New Hampshire, and Massachusetts. To the south, the gulf is open to the North Atlantic at the surface. Below the 50-m depth, however, Georges Bank forms a southern boundary for the gulf, making it semi-enclosed. The gulf is connected to the deep North Atlantic by three channels, the major passage being the Northeast Channel between Georges Bank and the Scotian Shelf. The interior of the gulf is characterized by deep basins (greater than 200 m) which are separated by irregular topography that includes shallow ridges, ledges, and banks. The largest and deepest basins are Georges Basin near the mouth of the Northeast Channel, Jordan Basin to the northeast, and Wilkinson Basin in the southwest. Jordan and Wilkinson Basins are separated by irregular, shallower topography that extends toward the central gulf from the Casco Bay - Penobscot Bay coastal region, and includes Jeffreys Bank, Platts Bank, and Cashes Ledge. Generalized bathymetry is shown in Figure 1.

Waters in the region derive from two primary sources: Scotian Shelf water (SSW) and continental slope water (SLW). The SSW enters the Gulf of Maine around Cape Sable in the near-surface layers (Smith 1983), while the SLW enters at depth through the Northeast Channel (Ramp *et al.* 1985). The two water types mix as they travel in a general counterclockwise motion around the gulf. Near the coast, currents also move the waters in a general counterclockwise direction along the coast, except south of the Penobscot Bay region where a portion of the

[43] National Marine Fisheries Service, Woods Hole, MA.
[44] Maine Department of Marine Resources, West Boothbay Harbor, ME.
[45] University of Maine, Walpole, ME.

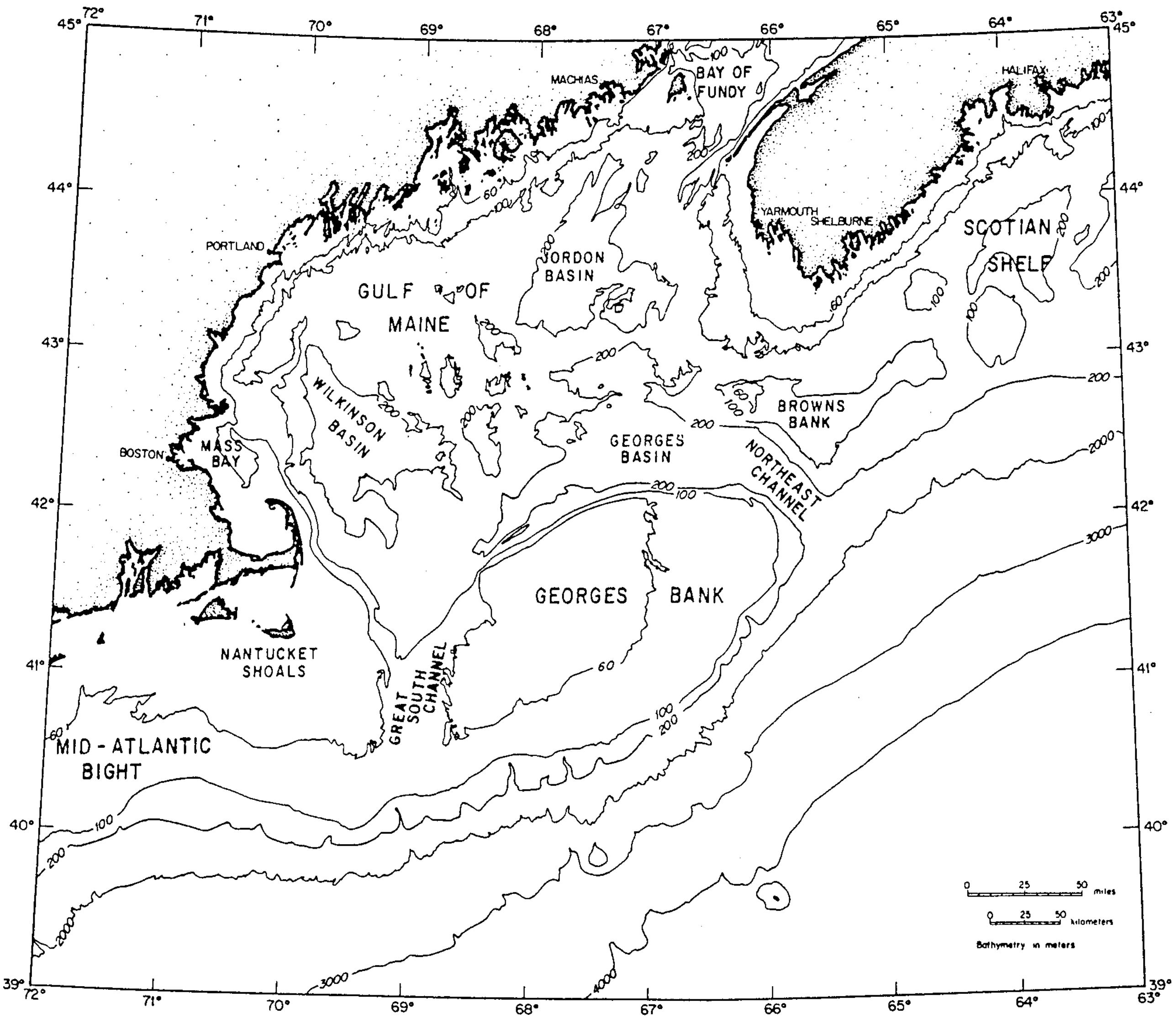

Figure 1. General bathymetry of the Gulf of Maine.

coastal flow turns offshore toward Jeffreys Ledge and the shallow topography between Jordan and Wilkinson Basins (Brooks 1985). From the southwestern gulf, surface waters over Wilkinson Basin enter a clockwise gyre on Georges Bank which takes water eastward to the northeastern part of the bank and then southwestward along the bank's broad southern flank (Hopkins and Garfield 1981). Mean residence time for water in (or travel time through) this region is approximately 1 yr.

Mean circulation of water through the region is characterized by velocities of 5-20 cm/sec, but actual water motion at any time is dominated by tidal currents and local wind-induced flow, and has surface velocities of 20-100 cm/sec. The Gulf of Maine is in near resonance with the M2 tidal component (Garrett 1972) which results in large tidal currents (80 cm/sec) in the eastern gulf - Bay of Fundy region, and in tidal currents of 10-20 cm/sec at the surface in the western gulf (Moody *et al.* 1984).

Water properties in the gulf vary in time and space. Primary temporal variability is associated with the seasonal cycle (*e.g.*, winter cooling, summer heating). Spatially, properties vary: (1) vertically, (2) in the east-west direction across the gulf, and (3) between the near-coastal regions and the more central portions of the gulf.

The surface layer of the Gulf of Maine experiences a large seasonal cycle in temperature due to surface heating and cooling. Surface temperature ranges from about 4°C across the gulf in March to about 18°C in the western gulf and 14°C in the eastern gulf during August. Salinity of the surface layer also varies seasonally, with minimum values in the west occurring during summer (from accumulated spring river inflow), and in the east during winter (from the low salinity of inflowing SSW). This low in salinity in the east originates from the peak outflow of the St. Lawrence River at the northern end of Nova Scotia during the previous spring. The seasonal range in surface-layer salinity is about 0.8 practical salinity units (PSU), but can vary among years with changes in the amount of precipitation and river inflow. With seasonal temperature and salinity changes, the density stratification over the upper layer also exhibits a seasonal cycle. From well-mixed, vertically uniform conditions in winter, stratification develops through the spring and reaches a maximum in late summer. The degree of stratification is greater over the western gulf than in the eastern half.

Vertically, the water column in the Gulf of Maine is characterized by three layers. The surface layer (0-50 m) is relatively fresh (31-33 PSU). Its temperature changes greatly through the year as a result of seasonal heating and cooling. An intermediate layer is found at mid-depths (50-100 m), and is identified by a temperature minimum which results from vertical convection driven by surface cooling and wind mixing during winter (Hopkins and Garfield 1979). As surface layers warm through spring and summer, the intermediate layer remains cool, forming a temperature minimum in the water column that is a remnant of the previous winter's cooling. The deeper portions of the gulf contain the Maine bottom water, which originates from SSW entering through the Northeast Channel, and which is warmer and saltier than the intermediate layer above it (Mountain and Jessen 1987).

Bottom waters of the Gulf of Maine exhibit a significant east-west difference in water properties that is larger than the seasonal variability (Mountain and Jessen 1987). In Georges Basin, near the inflow of SSW through the Northeast Channel, the bottom water has properties of 6-9°C and 34-35 PSU, while in Wilkinson Basin the values are 4-7°C and 33-34 PSU. Jordan Basin is intermediate between the other two basins.

Conditions near the coast of the Gulf of Maine are greatly influenced by local river input. Inflowing freshwater mixes with coastal waters, and forms a low-salinity, coastally trapped band of water which can extend 20 km or more from the coast. Dissolved and particulate content of river inflow (*e.g.*, nutrients, sediment, contaminants) is transported by the coastal currents and dispersed along the near-coastal region. The coastally trapped band also transports phytoplankton and can influence the temporal and spatial distribution of toxic phytoplankton blooms in the southwestern gulf (Franks and Anderson 1992a,b).

OCEANOGRAPHIC INFLUENCES ON DEMERSAL FISH HABITATS

Larger physical features and oceanographic phenomena, as described above, have constituted much of the research effort in the Gulf of Maine and Northwest Atlantic, but there is a growing awareness of the need to consider different, biological scales in these research programs (Cohen *et al.* 1991; Werner *et al.* 1993). Knowledge of variations in the fish populations, for example, is well documented for the region (Conservation and Utilization Division 1993). On the other hand, knowledge of factors controlling events at the level of physical habitats, such as specific fishing grounds or demersal fish nursery areas, is limited but essential for the sustained production of the region's fisheries.

Linking spawning strategies to the benthic habitat that supports the juvenile stages of major demersal fish species is an area of research that needs to be addressed. This stage of life history represents a shift from control of fish stocks by large-scale physical and oceanographic features to control by a finer scale represented by fishing grounds and nursery areas. It is at this finer scale that fishing as well as other human perturbations of the environment may have their most significant effect. This was recognized over 100 yr ago in the introductory remarks by Richard Rathbun to a book that described the fishing grounds of North America. Rathbun (1885) noted: "Many of the data furnished by the ordinary class of hydrographic work are, therefore, entirely unsuited to fisheries purposes, and it is of the greatest importance that special surveys be undertaken in the immediate interest of the fishery, and with the object of ascertaining the full extent and character of all the larger grounds that may be profitably resorted to by our fishermen." Rathbun goes on to point out that fishing grounds seldom conform exactly to contour lines in hydrographic charts, since fishes are not always influenced by difference in depths as much as by abundance of food and other physical factors such as temperature. A detailed series of charts and descriptions are then given for all known fishing grounds from Greenland to Mexico (Collins and Rathbun 1885). These and additional data on fishing grounds specifically in the Gulf of Maine were again summarized in a 1929 publication by Rich. Nevertheless, a U.S. research program designed to describe biological and physical interactions at this scale was not initiated until the late 1980s (Lough *et al.* 1989; Valentine and Lough 1992).

Demersal fish spawning strategies, spawning areas, recruitment, and their relation to large-scale oceanographic features have been investigated to some degree in the Northwest Atlantic for commercially important demersal fish species. Sherman *et al.* (1984) identified three spawning strategies for 11 continental shelf fish species. Hakes were classified as ubiquitous spawners which describes animals that have a protracted spawning period over a large geographic area. Atlantic cod, haddock, and Acadian redfish, on the other hand, have developed spawning strategies that optimize their chances of survival by being in synchrony with temporal and spatial increases of their zooplankton prey. Atlantic cod and haddock, for example, were concentrated on Georges Bank, and their maximum abundance was synchronized with the zooplankton within the bank's gyre. Werner *et al.* (1993) recently described a circulation model that helps to explain the interrelation between current flows on Georges Bank and the retention of larval Atlantic cod and haddock. Yellowtail flounder larvae have also been found to be retained on the Grand Banks of Newfoundland by the current regime (Walsh 1992).

Cohen *et al.* (1991) used time-series correlations to evaluate the effect of large-scale versus local-scale processes on recruitment for Atlantic cod and haddock in the Northwest Atlantic. Their results suggest that local-scale processes, rather than larger-scale climatic change for example, dominate recruitment patterns for these two species of fish. On a local scale, spawning aggregations of Atlantic cod have been known in the Gulf of Maine for many years, since fish hatcheries at the turn of the century conducted exploratory plankton tows in search of eggs and larvae. Knowledge of existing spawning grounds is perhaps less complete than years ago, but one spawning aggregation of Atlantic cod has persisted over time in Sheepscot Bay. Fish tagged in this region between 1978 and 1983 were recaptured in subsequent years on their return to the bay (see *Commercial Fisheries News*, January 1988). Furthermore, groups of fish tagged from the same tow were, on occasion, recaptured together in other areas by commercial fishermen or back in Sheepscot Bay during the following year (unpubl. data[46]).

Concentrations of age 0 and older juvenile fish generally indicate a nursery for the given species. Coastal areas often serve in this role. In Europe, in particular, coastal nurseries for cod, pollock, and plaice are well documented (Macer 1967; Edwards and Steele 1986; Zijlstra 1972, 1986; Lockwood 1974, 1980a,b, 1984; Thijssen *et al.* 1974; Kuipers 1975; Daan 1978; Burd 1978; Rauck and Zijlstra 1978; de Veen 1978; Zijlstra *et al.* 1982; Basimi and Grove 1985a,b; Hawkins *et al.* 1985; van der Veer 1986). Recently, fish nurseries have been a focus of research, and the idea that nurseries are restricted to estuaries has to be reconsidered. Coastal areas (Langton *et al.* 1989; Lenanton 1982), as well as exposed beaches (Bennett 1989; Gibson *et al.* 1993), the surf zone (Ross *et al.* 1987), and fjordic environments (Gordon and DeSilva 1980; Gordon 1981) have all been documented as important for production of small and juvenile fishes. Finally, for some commercially important Northwest Atlantic demersal fish, such as Atlantic cod, haddock, yellowtail flounder, and American plaice, offshore banks have been identified as oceanic nurseries (Lough *et al.* 1989; Walsh 1991, 1992).

BENTHIC SUBSTRATES AND THEIR INFLUENCES ON DEMERSAL FISH HABITATS

Bottom type within the Gulf of Maine is quite patchy and generally related to topography (Schlee 1973). Deep basins exhibit silty clay or clay sediments, while the irregular topography between basins generally has a higher fraction of sand. Topographic highs within the gulf are exposed to the winnowing action of currents, and are characterized by sand and gravels. In the near-coastal region (within about 16 km of the coast), the bottom type south of Casco Bay is largely sand, while north and east it is generally silt and clay (Schlee 1973). However, the bottom type, particularly in coastal and estuarine areas, may exhibit a large degree of small-scale variability (*e.g.*, see Butman *et al.* 1992, Figure 5).

Distributions of benthic species and assemblages of species in the Gulf of Maine are strongly related to bottom type (Watling *et al.* 1988; Langton and Uzmann 1989; Langton *et al.* 1990). Multidisciplinary surveys to map the seabed, describe surficial sediments, and identify faunal associations particularly on eastern Georges Bank, in the Gulf of Maine, and on the Grand Banks of Newfoundland have demonstrated the relationship between the physical and biological environment for juveniles of a number of demersal fish species (Valentine and Lough 1992; Langton *et al.* 1989; Langton and Watling 1990; see *Commercial Fisheries News*, December 1992; Walsh 1992). On eastern Georges Bank, for example, when juvenile Atlantic cod assumed a demersal life, they were, in contrast to the pelagic stage, limited to a gravel pavement area. Lough *et al.* (1989) and Valentine and Lough (1992) suggested that the juvenile coloration of Atlantic cod mimics the appearance of gravel, thus protecting these fish from predation. Recently, this hypothesis was confirmed experimently. Gotceitas and Brown (1993) found that juvenile Atlantic cod selected a cobble substrate in the presence of a predator, and preferred sand or gravel-pebble substrate in the absence of a larger conspecific.

In Sheepscot Bay, Gulf of Maine, juvenile winter flounder, yellowtail flounder, longhorn sculpin, and little skate associated with gravel and sand substrates, and preyed extensively on an amphipod species that was restricted to gravelly areas, while American plaice and ocean pout were dominant over muddy substrates (Langton *et al.* 1989; Langton *et al.*, in press). On the Grand Banks, yellowtail flounder associated with sand to gravelly sand (Walsh 1992). Bottom type selection by demersal fishes has been described for many of these species (Scott 1982; MacDonald *et al.* 1984), but this does not mean that bottom type is the only factor controlling juvenile fish distribution (*e.g.*, Horne and Campana 1989).

BENTHIC COMMUNITIES

As a result of its glacial history, the bottom of the Gulf of Maine consists of a wide diversity of sediment and rock mixtures (Oldale *et al.* 1973). Large moraines left by glaciers form some of today's offshore sand and gravel banks. Rock ledges, such as Cashes Ledge, which resisted glacial rasping, form some of the gulf's most noticeable landmarks. Since the last glacial retreat, several deeper basins have been filling with very fine silts and clays.

This wide variety of bottom types, when combined with the vertical variation in water properties, produces a diversity of habitats utilizable by fishes and their invertebrate prey. Invertebrate densities in these different habitats reflect the local-scale physical features and oceanographic processes of the region since they are a function of the supply of food from overlying waters. Based on numerical classification techniques, bottom invertebrates of the Gulf of Maine have been arranged into six community types: (1) nearshore shallow, (2) boreal mud, (3) sand bank, (4) rock ledge, (5) boreal-slope transition, and (6) a mimic of the upper slope (Watling *et al.* 1988). Figure 2 depicts the general location of these communities in the gulf.

The nearshore shallow (0-50 m) community is perhaps the most diverse and has the greatest abundances of individuals. All

[46] H.C. Perkins, Maine Department of Marine Resources, West Boothbay Harbor, ME.

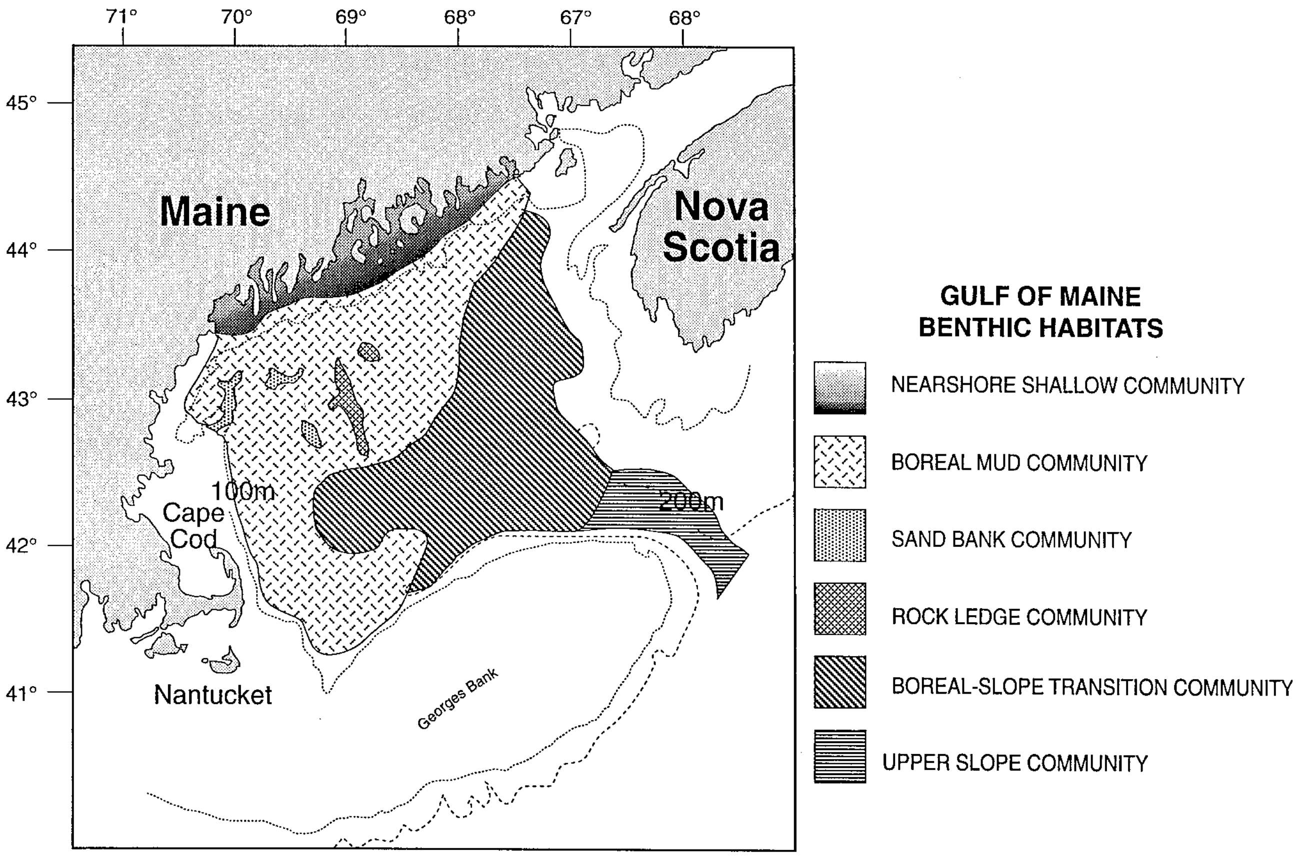

Figure 2. General location of benthic community types in the Gulf of Maine.

combinations of sediment type are found here, but perhaps most importantly, this community receives organic input from terrestrial sources, from attached macroalgae from the nearby rocky shore, and from phytoplankton. More than 500 macrofaunal species representing nearly all invertebrate phyla can be found here. Many species are typical in having their major period of reproduction in summer, but several are true boreal forms (*e.g.*, *Unciola inermis*, in Morrison 1993), and reproduce when water is colder (*i.e.*, from November to February).

Just offshore of the nearshore shallow community, is a widespread, fine-sediment community--the boreal mud community--under the influence of Maine intermediate water (Hopkins and Garfield 1979; Langton and Uzmann 1989). Because water temperatures here are colder and vary only a few degrees over the year, the sole seasonal cue for reproduction is input of the spring phytoplankton bloom. In this area, species diversity is still quite high, but because the organic input is lower, population abundances of infaunal species are much reduced, being as low as 25% of that seen in the nearshore shallow community (Packer 1988). Megafaunal species which are characteristic include the sea pen *Pennatula aculeata* (see Langton *et al.* 1990) and a large cerianthid anemone, probably *Cerianthus borealis* (see Shepard *et al.* 1986). In addition, for many species there is no indication of a seasonal reproductive pulse, and for several others, reproduction seems to be driven by the spring bloom (Ward 1988).

The sand bank community, which is found in the area where Maine intermediate water predominates, is typified by Fippenies and Jeffreys Ledges (Langton and Robinson 1990). Sediment is always sand and gravel, with little or no fine material. Filter feeders, such as the polychaete *Myxicola infundibulum*, the sea scallop, and various sponges predominate. Because these banks protrude into the photic zone from quite deep water, they are subjected to localized upwelling, and thus plankton blooms in their vicinity can be longer in duration than those near shore.

The rock ledge community is also found in the area where Maine intermediate water predominates. Organisms in this habitat are also primarily filter feeders, and are generally large. Because density and quality of food particles are often low, this community's organisms do not completely cover the available surface (Hulbert *et al.* 1982; Witman and Cooper 1983; Witman and Sebens 1988). Much of the biodiversity in this habitat is due to the wide variety of smaller epibiotic macrofauna that live on colonial filter feeders.

Between the boreal mud community and the upper continental slope is the boreal-slope transition community. This community is found primarily in Jordan Basin, on swells west of Georges Basin, and to a limited extent in Wilkinson Basin. Sediment is predominantly mud or a mud veneer overlying sand and gravel. This is a zone of mixing of Maine bottom water with

Maine intermediate water, so bottom-water temperatures might vary by several degrees over a period of years, depending on the extent of inflow of warmer waters through the Northeast Channel. Characterisitic species include the brittle star *Ophiura sarsi* and the tube-making amphipod *Erichthonius* sp.

The deepest part of the Gulf of Maine is inhabited by species representative of an external community--the upper slope community (Rowe *et al.* 1975, 1982). These species enter the gulf with upwelling slope water, and can be found wherever slope water is not strongly diluted with Maine intermediate water. Sediments in this area are generally sand mixed with both fine particles and gravel. Characteristic species include the large (about 5-cm long) foraminiferan *Bathysiphon* and several deepwater species of isopods. Abundances are generally low, as are sediment bacteria numbers, suggesting low levels of organic input. Reproduction is most likely continuous for most species.

NUTRIENT LOADING AND EUTROPHICATION

John E. O'Reilly[47]

During the past few decades, estuarine and coastal waters of the northeastern United States adjacent to highly urbanized and high-intensity agricultural areas have experienced significant increases in inputs of nitrogen, phosphorus, and other plant nutrients. In the eastern United States and western Europe, contemporary nutrient loading of rivers is probably 10-50 times greater than prehistoric loading (Hinga *et al.* 1991) as a consequence of deforestation, sewage disposal, urban road runoff, and agricultural fertilizers (Walsh 1981; Stoddard *et al.* 1986; Rosenberg 1985; Larsson *et al.* 1985; Smith *et al.* 1987). Upgrading treatment of sewage wastes to a "secondary" level has, in the past several decades, effectively reduced the inputs of some sewage pollutants (*e.g.*, biochemical oxygen demand (BOD), particulates, and pathogens), and has generally resulted in greater minimum summer concentrations of dissolved oxygen (DO) (Environmental Protection Agency 1993). Some sewage-derived nutrients such as ammonium and phosphorus appear to have decreased following improved sewage treatment, while others such as nitrite-nitrate appear to have been only slightly affected (Ayers *et al.* 1988; New York City Department of Environmental Protection 1991).

Other inputs of nutrients to the coastal zone, including agricultural and suburban runoff as well as nitrates from acid rain, may also contribute significantly to nutrient overenrichment (Fisher *et al.* 1988; Paerl *et al.* 1990; Hinga *et al.* 1991). In coastal rivers entering the Northwest Atlantic, a trend of increasing nitrate loading concurrent with decreasing phosphorus loading between 1974 and 1981 was reported by Smith *et al.* (1987). Unless controlled, nutrient loading will likely increase from a variety of human activities concomitant with population increases expected in coastal counties. Population in coastal counties bordering the Gulf of Maine region increased approximately 6% per year (Colgan 1992) between 1980-81 and 1986. The projected demographic trend for the Northeast coastal zone, combined with anticipated modest efforts to control loadings of nutrients, suggests that nutrient overenrichment will continue to affect coastal water quality and fisheries adversely well into the next century.

[47] National Marine Fisheries Service, Narragansett, RI.

The problem of nutrient overenrichment in U.S. coastal waters is a national issue which NOAA is addressing through the Coastal Ocean Program, and the U.S. Environmental Protection Agency (EPA) through its estuarine programs.

NUTRIENT FERTILITY, PRIMARY PRODUCTIVITY, AND FISHERIES PRODUCTIVITY

Nutrient fertility, that is, the availability of nutrients through new inputs and recycling, plays a significant role in trophodynamics, productivity, and consequent health of aquatic ecosystems. Eutrophication, or the state of "enhanced nourishment" (Joint Group of Experts on the Scientific Aspects of Marine Pollution 1990), may occur through natural processes (*e.g.*, upwelling) or from inputs of nutrients derived from human activities. Effects of eutrophication are not necessarily always adverse ones. For instance, a doubling of benthic production and increases in growth rates of bottom-feeding fishes followed increases in planktonic productivity which resulted, in turn, from experimental additions of inorganic fertilizers to Scotland's sea lochs (Raymont 1949; Mearns *et al.* 1982). Iverson (1990) developed convincing arguments that carnivorous fish production in coastal and open-ocean environments (including the Gulf of Maine) is controlled by the amount of new nitrogen entering the euphotic layer and consequent new primary production, and not by systemic differences in trophic transfer efficiency or number of steps in the food chain.

Nixon (1988, 1992) reported a strong association between the landings of fishes and invertebrates and the annual phytoplankton primary production and nitrogen input to numerous estuarine and marine ecosystems; fisheries yield approached 1% of phytoplankton carbon production in the most productive systems. In a series of nutrient enrichment experiments in mesocosms, a 15-fold increase in phytoplankton chlorophyll and a 4.5-fold increase in daytime primary production accompanied a 60-fold nitrogen enrichment; total system production of the mesocosms increased with nutrient loading (Nixon 1992). High production in marine ecosystems is "virtually always associated with a high rate of inorganic nutrient input to the surface water" (Nixon 1992, p. 64). In freshwater systems, similar relationships have been evident for many years from both empirical (Vollenweider 1976; Lee and Jones 1984) and experimental studies (Schindler 1975, 1987), except that in limnotic systems, phosphorus, not nitrogen, is usually the key nutrient.

Thus, a quantitative relationship between nitrogen inputs, primary production, and fisheries yields is evident when comparing a spectrum of ecosystems. Within some individual systems, evidence suggests that elevated nutrient inputs increase not only the "quantity" of primary production, but also affect the "quality" (phytoplankton species, size composition, and nutritional content) of the organic matter produced. Such qualitative changes may not always be most advantageous to the food web or for human utilization of resource species. Thus, enhanced nutrient availability to phytoplankton represents a "double-edged sword" in that it enhances overall system production, but may simultaneously generate conditions favoring more frequent or persistent blooms of noxious or toxic phytoplankton. Enhanced nutrient availability can also generate depressed concentrations of DO and consequent fish and invertebrate mortalities when algal blooms greatly exceed either the assimilation capacity of herbivores or the capacity of the system to disperse the blooms physically.

NUTRIENT OVERENRICHMENT EFFECTS

There is considerable concern and **some** evidence that nutrient overenrichment has resulted in several undesirable eutrophication effects in estuarine and coastal areas of the northeastern United States, including: (1) increased incidence, extent, and persistence of blooms of noxious or toxic species of phytoplankton associated with mortality or reduced productivity of economically or ecologically important marine species, as well as decreased fisheries harvests and reduced aesthetic value of coastal areas; (2) increased frequency, severity, areal extent, and persistence of hypoxia, the condition of depressed concentrations of DO in bottom waters such as reported to occur during summer in Chesapeake Bay, Raritan-Hudson Estuary, western Long Island Sound, inshore New York Bight, and portions of Boston and Portland Harbors, resulting in mortality of benthic organisms, reduced growth and production of fisheries resources, and changes in resource distributions; (3) alterations in dominant phytoplankton species and size compositions, as well as in the nutritional-biochemical "quality" of the phytoplankton community, causing changes in structure, function, and productivity of the food chain culminating in the fisheries; and (4) greatly increased turbidities of surface waters from planktonic algae leading to "shading" and consequent losses of bottom macrophytes such as eelgrass, to reductions in critical estuarine habitats for early life stages of fishes, and to reduced aesthetic appeal of recreational waters.

EUTROPHICATION STATUS IN THE GULF OF MAINE

Long-term, coherent, time-series measurements of *in situ* concentrations of nutrients, processes affecting nutrients, and nutrient loading in the Gulf of Maine are lacking (Loder *et al.* 1991). Therefore, other scientific approaches, each with many assumptions and much imprecision, are required to assess fully the status, trends, and ecological consequences of nutrient loading in this region.

Retrospective approaches have been used successfully to derive assessments of nutrient loading and eutrophication. For example, stratigraphic analyses of sediment cores have shown that anoxia and eutrophication have increased in Chesapeake Bay since the time of European settlement (Cooper and Brush 1991).

An extensive nationwide review of oxygen depletion and eutrophication in estuarine and coastal waters was conducted during the mid-1980s by Whitledge (1985) and coworkers. They reviewed the scientific literature as well as federal, state, and county data bases and reports, and contacted individuals familiar with these waters. Their general finding was that extensive hypoxia (*i.e.*, episodes of DO concentrations less than 4 mg/l) was more chronic in river-estuarine systems in the southern portion of the Northeast--Narragansett Bay to Chesapeake Bay--than in the northern portion, except for episodes of low DO in Boston Bay/Charles River and the freshwater portion of the Merrimack River between river miles 20 and 30. Generally, hypoxia has been associated with loading from municipal and combined municipal-industrial sewers. Only two areas on the open U.S. continental shelf have been reported to be affected by recurring hypoxia: coastal waters of the New York Bight and the inshore Gulf of Mexico off Louisiana (Whitledge 1985).

Hypoxia has not been noted to have occurred in offshore bottom waters of the Gulf of Maine. From a biological oceanographic perspective, Sinclair *et al.* (1992) indicated that there is no evidence of pollution effects in the deep offshore basins in the Gulf of Maine. Loder and Becker (1989, p. 83) concluded that: "At the present there does not seem to be a problem of eutrophication in the Gulf of Maine overall, but as we have seen there are problems starting to occur in localized coastal areas. These are the first warning signs that parts of the system are being stretched beyond their normal assimilation capacity with regards to nutrients. The problems will only get worse if steps are not taken to set control practices in place."

Assessment of the magnitude of nutrient overenrichment, using field measurements of nutrient concentrations, can have limitations because dissolved inorganic forms of major plant nutrients (*e.g.*, nitrogen, phosphorous, and silicon) can be rapidly photoassimilated by phytoplankton into particulate

Table 4. Summary of physical and hydrological characteristics, nitrogen and phosphorus discharges, and predicted nutrient concentration statuses for estuarine systems bordering the Gulf of Maine [a]

Estuarine System	Total Drain. Area (mi^2)	Water Surf. Area (mi^2)	Water Depth (ft)	Average Daily Freshw. Inflow (10^2 cfs)	Volume (10^9ft^3)	1980 Pop. Density (no./mi^2)	Total Nitrogen Disch. (tons/yr)	Nitrogen Conc. & Status[b] (mg/l)	Total Phos. Disch. (tons/yr)	Phos. Conc. & Status [b] (mg/l)
Pasamaquoddy Bay	3200	157	72	62	315	11	293	0.008L	28	0.001L
Englishman Bay	900	76	38	16	80	12	150	0.014L	22	0.002L
Narraguagus Bay	400	70	32	9	63	17	104	0.016L	11	0.002L
Blue Hill Bay	800	115	75	13	241	28	154	0.016L	35	0.004L
Penobscot Bay	9400	361	72	161	725	58	7808	0.102M	771	0.010M
Muscongus Bay	300	72	43	6	85	67	56	0.013L	16	0.004L
Sheepscot Bay	10100	103	41	176	118	66	8745	0.077L	641	0.006L
Casco Bay	1200	164	42	21	191	172	1412	0.086L	465	0.028M
Saco Bay	1800	17	32	36	15	71	1257	0.057L	193	0.009L
Great Bay	1000	15	11	20	5	243	636	0.098L	204	0.031M
Merrimack River	5000	6	12	84	2	423	10111	1.021H	1625	0.164H
Massachusetts Bay	1200	364	77	29	786	2228	7995	0.216M	4091	0.110H
Boston Bay	700	69	26	18	50	2789	N/A	N/A	N/A	N/A
Cape Cod Bay	800	548	77	18	1178	392	377	0.026L	187	0.013M

[a] Data sources:Strategic Assessment Branch (1990); NOAA/EPA Team on Near Coastal Waters (1989).
[b] Concentration status: L = low; M = medium; H = high.

organic matter, and quickly recycled by the plankton community and seabed into inorganic forms. An additional complication is the emerging evidence that denitrification is a potentially significant component of nutrient budgets of coastal ecosystems (Seitzinger 1988; Christensen *et al.* 1992). Unless all forms of a nutrient are measured frequently, then sporadic field measurements of just the inorganic dissolved form will probably not yield a picture of the total nutrients in the system which would be adequate for constructing nutrient budgets and tracking temporal trends in nutrient loading. Nutrient forms that must be assessed are inorganic salts, gasses, nutrients bound organically in particulates or in dissolved organic matter, key rate processes, and nutrient sources and sinks. Thus, from a practical point of view, algal growth (and blooms) may be more closely related to the **rate** of nutrient supply for a region than to nutrient standing stocks and field measurements of nutrient standing stocks (Howarth 1988).

Another approach to eutrophication assessment, used by the NOAA/EPA Team on Near Coastal Waters (National Oceanic and Atmospheric Administration/Environmental Protection Agency 1988, 1989) combines information on the estuarine drainage area, the nitrogen and phosphorus loading rates, and the physical and hydrological characteristics (*i.e.*, freshwater inflow, flushing rates) to **infer** the nutrient concentration status of each system and its relative **susceptibility** to increases (or decreases) in nutrient loading and eutrophication. This approach is not a definitive assessment, but a screening device. It does not account for nutrient recycling or offshore natural inputs or atmospheric nutrient inputs. It deals with annual scales (*e.g.*, average annual freshwater inflow and salinity) and assumes that nutrient inputs are uniformly distributed vertically and horizontally within each system, and are regularly related to water mass inputs. Though the "nutrient concentration status" categorization scheme is a relative one, it may have management application. A low concentration status supports maximum diversity of benthic resources, submerged aquatic vegetation, and fisheries. A medium concentration status supports moderate diversity and may result in reduction of submerged aquatic vegetation and in occasionally high chlorophyll levels. A high concentration status results in a significant reduction in resource diversity, in loss of submerged aquatic vegetation, in frequently high levels of chlorophyll, and occasionally in red tide or algal blooms (NOAA/EPA Team on Near Coastal Waters 1988, p. 5).

Following are excerpts from the NOAA/EPA Team on Near Coastal Waters (National Oceanic and Atmospheric Administration/Environmental Protection Agency 1989) which highlight those estuarine systems bordering the Gulf of Maine having a medium or high nutrient potential concentration status (Table 4 presents actual nutrient levels for all bordering estuarine systems):

1. Penobscot Bay: "the medium phosphorus concentration classification may be influenced by a minor reduction

(<20%) in phosphorus loading. The N/P [nitrogen/phosphorous] molecular ratio of the loading is 22, suggesting that phosphorus may be a limiting nutrient in the estuary." (p. 19)

2. Casco Bay: "the low nitrogen concentration classification may be influenced by a minor increase (<20%) in nitrogen loading. The N/P molecular ratio of the loading is 7, suggesting that nitrogen may be a limiting nutrient in the estuary." (p. 22)

3. Great Bay: "the low nitrogen concentration classification may be influenced by a minor increase (<20%) in nitrogen loading. The N/P molecular ratio of the loading is 7, suggesting that nitrogen may be a limiting nutrient in the estuary." (p. 24)

4. Merrimack River: "the high nitrogen concentration classification may be influenced by a minor reduction (<20%) in nitrogen loading. The N/P molecular ratio of the loading is 14, and does not strongly indicate the presence of a limiting nutrient in the estuary." (p. 25)

5. Massachusetts Bay: "the high phosphorus concentration classification may be influenced by a minor reduction (<20%) in phosphorus loading. The N/P molecular ratio of the loading is 4 suggesting that nitrogen may be a limiting nutrient in the estuary." (p. 26)

6. Cape Cod Bay: "these concentration classifications are not likely to be influenced by minor changes (<20%) in nutrient loadings. The N/P molecular ratio of the loading is 4, suggesting that nitrogen may be a limiting nutrient in the estuary." (p. 27)

Table 4 suggests that certain coastal areas in the Gulf of Maine are experiencing or may experience problems related to human-induced nutrient loading. For comparison, using the NOAA/EPA Team on Near Coastal Water approach (National Oceanic and Atmospheric Administration/Environmental Protection Agency 1989), the predicted concentration status for Long Island Sound is 0.271 mg/l for nitrogen (*i.e.*, medium) and 0.041 mg/l for phosphorus (*i.e.*, medium). Episodes of hypoxia (*i.e.*, DO less than 3 mg/l) occurred in the sewage-polluted western portion of Long Island Sound during the 1970s, and perhaps became more recurrent and possibly more severe in the late 1980s. In summer 1987, an unprecedented episode of anoxia and fish kill occurred in both the Western and Eastern Narrows (Parker and O'Reilly 1991). During intensive bottom trawl surveys of the western sound, the Connecticut Department of Environmental Protection observed "the late summer absence or reduced presence of fishes and lobsters in hypoxic zones. Both the number of species and the total number of fishes caught were greatly reduced in hypoxic waters" (Long Island Sound Study 1990, p. 12).

The Long Island Sound Study (1990) identified hypoxia as the environmental issue of highest priority, and has recommended several interim management strategies which would reduce nutrient loadings and improve water quality. Similar strategies were proposed to control the amounts of nitrogen and phosphorus entering Chesapeake Bay (Scientific and Technical Advisory Committee 1986).

Coastal water near densely populated areas, such as that bounded by Cape Ann, Cape Cod, and Stellwagen Bank, may experience changes in nutrient concentrations, nutrient ratios, and phytoplankton species (Loder and Becker 1989) resulting from redirection of sewage nutrients from Boston Harbor to an offshore effluent outfall originally scheduled to begin operation in July 1995 (Massachusetts Water Resources Authority 1991), but recently rescheduled for 1996 or later. Loder and Becker (1989) estimate that nitrogen loading from the outfall will be 3-5 fold that estimated for the Merrimack River, or, expressed another way, approximately 10% of nitrogen loading from Gulf of Maine rivers. There is some concern that redistribution of sewage effluents from nearshore Boston Harbor surface waters to offshore subpycnocline waters could potentially accelerate delivery of nutrients and organic matter (*i.e.*, BOD) to Stellwagen Basin, and potentially lower near-bottom DO concentrations during summer (Kelly 1991, 1993).

A comprehensive monitoring plan has been devised by the Massachusetts Water Resources Authority (1991). One facet of the plan consists of field surveys of nutrients and DO, and an analysis of their potential effects on phytoplankton, primary productivity, and zooplankton in "nearfield" and "farfield" areas surrounding the effluent outfall, before and during its operation. Detectible changes in phytoplankton chlorophyll in the nearfield area are expected; the exact magnitude of change depends upon many physical and biological factors, and is not certain (Massachusetts Water Resources Authority 1991).

ATMOSPHERIC DEPOSITION OF NITROGEN

The importance of human inputs of nitrogen from atmospheric dry and wet deposition to the nutrient budgets of coastal areas in the Northeast has emerged only relatively recently. Atmospheric deposition of biologically available nitrogen has been estimated to represent a potentially significant (20-30%) contribution to estuarine and coastal nutrient budgets and must be considered in eutrophication management strategies (Fisher *et al.* 1988; Paerl *et al.* 1990; Hinga *et al.* 1991). Atmospheric nitrogen emissions are projected to increase during the next several decades (Irving 1991). To date, the exact contribution of acid rain and atmospheric dry deposition to the nitrogen budget of the Gulf of Maine - Georges Bank region has not been determined. The estimated atmospheric wet deposition of nitrate in Gulf of Maine coastal waters is approximately 15 kg/ha, and dry deposition might add a comparable, but probably lesser amount, of nitrate (R.S. Artz, pers. comm.[48]).

[48] NOAA Air Resources Laboratory, Silver Spring, MD; October 1990.

The importance of atmospheric deposition relative to other major human sources of nitrogen (*i.e.*, sewage treatment plants and agriculture) is likely to vary with the type and size of the system and the watershed. Atmospheric nitrogen may be eclipsed by other human sources in heavily farmed and urbanized areas, such as Long Island Sound, Chesapeake Bay, and New York Harbor, while in coastal ecosystems having relatively lightly developed watersheds (such as Gulf of Maine), atmospheric nitrogen may dominate over inputs from sewage and agriculture (Hinga *et al.* 1991).

The nitrogen budget developed by Schlitz and Cohen (1984) supports the generalization that nitrogen flux from rainfall is twice that from river transport. Based on the revised higher estimate for river-borne nitrogen, the contribution from rivers and atmosphere is comparable (Loder and Becker 1989). Nevertheless, for the Gulf of Maine - Georges Bank region as a whole, human sources of nitrogen, via rivers and atmospheric deposition, may represent only 3-5% (Schlitz and Cohen 1984; Loder and Becker 1989) of the flux of new nitrogen, and could not generate ecosystemwide eutrophic conditions.

RELATIVE IMPORTANCE OF NATURAL AND HUMAN-INDUCED NUTRIENTS

Several nutrient budgets have been constructed for the Gulf of Maine - Georges Bank region in an effort to explain the high primary and fisheries productivity of the area (Schlitz and Cohen 1984; Walsh *et al.* 1987; Loder and Becker 1989; Townsend 1992). These budgets indicate that for the entire Gulf of Maine - Georges Bank region, nitrate-rich slope water, entering through the Northeast Channel, strongly dominates over other sources of new nitrogen. The Northeast Channel represents a major nutrient artery which potentially controls the fertility and production of the Gulf of Maine region (Schlitz and Cohen 1984; Ramp *et al.* 1985; Mountain and Jessen 1987).

Townsend (1992) categorized the major nutrient sources to the phytoplankton primary producers in the Gulf of Maine euphotic layer: (1) winter convective overturn, (2) flux from the eastern Maine coastal current/plume system, (3) vertical eddy diffusion, (4) coastal upwelling, and (5) recycled production. Their study indicated that the greatest uncertainties are in estimates of the vertical eddy diffusion across the seasonal thermocline and in the contribution of recycled nitrogen.

In near-surface, stratified waters in the Gulf of Maine, nutrient depletion is evident from May through October when rates of phytoplankton production are high, while below the thermocline, high concentrations of major nutrients are present (Pastuzak *et al.* 1982; Draxler *et al.* 1985; Walsh *et al.* 1987). This concentration of nitrogen, phosphorus, and silicate below the seasonal thermocline makes this natural source of nutrients particularly important to phytoplankton in areas near tidal mixing fronts and upwelling areas (Yentsch and Garfield 1981; O'Reilly *et al.* 1987).

In the nitrogen budget proposed by Schlitz and Cohen (1984), the estimated contribution from the Northeast Channel, Scotian Shelf, river discharge, and rainfall is 81.7%, 15.6%, 0.9%, and 1.8%, respectively, of the total. Loder and Becker (1989) suggest that the contribution from rivers was underestimated by Schlitz and Cohen by a factor of 2-3. Nevertheless, in this ecosystemwide analysis, the magnitude of human sources via rivers and atmospheric deposition is quite small when compared with natural sources of new nitrogen from offshore. The relative importance of human sources of nutrients will be greater in estuarine and nearshore coastal areas influenced by estuarine plumes (Loder and Becker 1989).

Offshore, oceanic nutrient sources apparently influence not only the deeper central regions of the Gulf of Maine, but also the nutrient status and dynamics of coastal waters in the western Gulf of Maine. Townsend *et al.* (1987) estimate that "as much as 44% of the new nitrate which enters the Gulf of Maine at depth through the Northeast Channel upwells in the eastern Gulf becoming part of the [coastal] plume, and hence this feature appears to be very important to the nutrient budget and general biological oceanography of the inner Gulf of Maine" (p. 699). Consequently, with this type of mechanism to deliver offshore nutrients into nearshore waters, the resolution of river-borne human sources from natural sources of nutrient enrichment becomes very difficult. This is particularly true regarding attempts to discern the relative importance of human and natural sources of nutrients and episodes of noxious plankton blooms in nearshore areas (such as paralytic shellfish poison (PSP) blooms).

NUTRIENT LOADING AND NOXIOUS ALGAL BLOOMS

Information accumulating on the incidence of noxious phytoplankton blooms worldwide over the last 30 yr has persuaded some specialists that the problem is increasing. Various factors, including increases in inputs from human activity and global change in conditions that regulate growth of phytoplankton, have been postulated to explain the apparent increased bloom incidence. Long-term increases in loading from human sources, and alterations in the ratios of nitrogen, phosphorus, and silicon in coastal rivers, have been linked to new appearances of noxious species, and to increased frequency and duration of noxious phytoplankton blooms (Smayda 1991; Anderson 1985; Riegman 1991). Increases in ratios of nitrogen or phosphorous to silicate may favor nuisance/toxic flagellate species over diatom species (Officer and Ryther 1980; Smayda 1990). Blooms of noxious and toxic phytoplankters can have effects throughout the marine food web (White 1982, 1988, 1992), including "reduced fecundity, survival and recruitment, and increased mortality of first feeding, juvenile, and adult stages" (Smayda 1991, p. 275).

Of particular concern in the Northeast are: (1) recurring blooms of *Alexandrium tamarense* in the western Gulf of Maine

(Anderson *et al.* 1982; Franks and Anderson 1992a,b); (2) emergence of PSP in bivalve mollusks on Georges Bank, presumably from the toxic phytoplankter *A. tamarense* (Anderson and Keafer 1992; White 1992); (3) "brown tides" of *Aureococcus anophagefferens* and their catastrophic effects on bay scallop, blue mussel, eel grass, and other resources in Long Island embayments and Narragansett Bay (Cosper *et al.* 1989); and (4) blooms of the ichthyotoxic dinoflagellate *Gyrodinium aureolum* and several "red tide" species in coastal waters in the New York Bight and the Gulf of Maine (Mahoney *et al.* 1990; Heinig and Campbell 1992), as well as extensive blooms of *Emiliania huxleyi* in the Gulf of Maine (Ackleson *et al.* 1988).

Historically, PSP and its causative agent, *Alexandrium tamarense*, in eastern North America were a regional problem, primarily localized in the northern half of the Gulf of Maine in relatively unpolluted areas (Hurst 1975). Apparently, during about the past 25 yr, the causative dinoflagellate and PSP have spread southward. The greatest expansion occurred in 1972 when high PSP levels were detected at sites along the entire Maine coast and, for the first time, southward to New Hampshire and Massachusetts. In Maine during 1980, estimated economic losses from PSP were in excess of $7 million (Shumway *et al.* 1988). In the Gulf of Maine - Georges Bank region, the relative roles played by human and nonhuman sources of nutrients in triggering PSP blooms are not known. To date, a strong recurring association has been found between blooms of *Alexandrium tamarense*, PSP toxicity, and the coastal plume of relatively brackish, stratified waters forming in the western Gulf of Maine during spring (Franks and Anderson 1992a,b). The extent to which this inshore water mass is enriched with human-induced nutrients, and whether humans have had an increasing direct role (more nutrients or changes in nutrient ratios) or indirect role (increased freshwater runoff and enhanced stratification of the coastal plume) in fostering environmental conditions favorable for blooms of *A. tamarense* is not known.

NUTRIENT LOADING: CONCLUSIONS AND RESEARCH NEEDS

Some nearshore, heavily populated areas of the Gulf of Maine are experiencing or have the potential to experience eutrophication effects from human inputs of nutrients. There is no indication of either natural or human-induced eutrophication in the offshore areas. More specific assessments will require more specific data. The following research needs regarding nutrient loading were stated at the 1991 Gulf of Maine Scientific Workshop (Loder *et al.* 1991): (1) historical analyses and synthesis of existing data to determine past trends and predict future trends; (2) better budgets relative to inputs, especially in the coastal zone including rivers, sewage, atmosphere, benthic regeneration, *etc.*; (3) understanding the role of nutrient ratios, how they change in sources to and within the Gulf of Maine, and how they affect species composition; and (4) better understanding of coastal nutrient dynamics, including redistribution processes and regeneration.

POLLUTANT EFFECTS ON DEMERSAL FISHES

Edith Gould[49], Paul E. Clark[50], and Frederick P. Thurberg[51]

The following review focuses on effects of two major contaminant categories or groups--heavy metals and organic contaminants--on each of 13 commercially important demersal fish species found in the Gulf of Maine. No attempt has been made to review literature for any pollutant effects associated with ocean dumping or mining, eutrophication and consequent low DO, or sewer outfalls and the like, as most of these subjects are covered elsewhere in this document.

Of these 13 species, only two have been the subject of a number of contaminant-effect publications. The European literature abounds in both experimental and field work with the Atlantic cod, from which relevant reports have been abstracted for inclusion here with American and Canadian papers. We assume that research on the same species found in the Gulf of Maine should be applicable, wherever the work is performed. In the northeastern U.S., the winter flounder has been the focus of considerable experimental work. Field and laboratory reports on the remaining 11 species, whether cause-and-effect or circumstantial evidence, or even informed speculation, are sparse to nonexistent. The winter flounder and Atlantic cod are quite different in terms of habitats used, and may be "representative" of several other species based on habitat occupied.

Table 5 lists units of concentration used in the following descriptions of pollutant effects.

[49] National Marine Fisheries Service, Milford, CT.
[50] *id.*
[51] *id.*

Table 5. Units of concentration (and their equivalents) pertinent to understanding pollutant effects

LC_{50} =	concentration (of toxicant) at which 50% mortality of experimental organisms is observed
ppb =	parts per billion [= µg/l = µg/kg]
ppm =	parts per million [= mg/l = µg/g]

ATLANTIC COD

Heavy Metals

Observations of toxic effects of heavy metals on cod are comparatively few; most of the work has been with petroleum hydrocarbons. Cod from the Baltic Sea, however, 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); the cadmium content in cod liver and kidney increased from the western to the eastern sector of the study area. Areal differences in concentrations of zinc, copper, and cadmium in cod muscle reflected the areal concentrations in seawater (Perttila *et al.* 1982). Cod muscle tissue was also reported to have a higher storage capacity for mercury than liver tissue, of possible concern to the consumer (Julshamm *et al.* 1982). Threshold concentrations of toxic mixtures (copper plus zinc, for example, with an anionic surfactant) need not be high to produce toxic effects in cod (Swedmark and Granmo 1981). Limited water circulation in estuarine and coastal waters would be most likely to produce examples of such effects in young cod, which use those nursery areas.

Organic Contaminants

Pesticides

Cod larvae are far more sensitive to the chlorinated pesticide DDT and its breakdown product DDE than are 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). 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 days (Shparkovskii 1982).

Following a 1972 DDT ban in Norway, cod liver samples showed decreasing concentrations of DDT; 10 yr after the ban, highest levels of DDT in cod liver were about one-third of the corresponding 1972 residue level. The slow clearance of DDT in cod liver, as compared to other species examined, may be attributed to the substantially higher fat content of cod liver (Skåre *et al.* 1985). Chlorinated pesticides (DDT and its analogs) have the ability to accumulate in the lipid-rich tissues of cod liver in quantities far exceeding the amount present in the flesh (Sims *et al.* 1975), a fact more important to the fish than to the consumer.

Most organochlorine compounds in livers of Atlantic cod caught off the east coast of Canada in 1980 showed no change in concentrations over the previous 8 yr, with the exceptions of PCB and the DDT group where there was a general decline between 1972 and 1975 with no significant change thereafter (Freeman and Uthe 1984).

PCBs

Cod having ulcus syndrome (epidermal lesions) had significantly higher PCB residue concentrations in liver tissue than did cod without the syndrome. Ulcus is thought to be due to an imbalance in corticosteroid metabolism caused by elevated PCBs (Stork 1983).

Levels of PCB congeners in liver samples of male and female cod reflected a decreasing PCB pollution gradient away from the mouth of the Glomma River, Norway's largest. PCB levels in female cod varied significantly with seasonal sampling (*i.e.*, levels in September/October being higher than corresponding levels in June and November/December); no such effect was seen in male cod (Marthinsen *et al.* 1991). Different PCB congeners produce different effects in marine fish. Feeding Aroclor 1254 to juvenile cod to produce liver concentrations of *ca.* 900 ppm (wet weight) induced activity of the enzyme ethoxycoumarin O-de ethylase 30-fold, but had no effect on ethoxyresorufin O-de ethylase (EROD) activity. Feeding Aroclor 1016 to juvenile cod induced no enzyme activity of this P450 group (Hansen *et al.* 1983).

PAHs

When exposed to radiolabeled polycyclic aromatic hydrocarbons (PAHs) and a PCB for 24 hr, cod eggs and newly hatched larvae accumulated both from seawater. After being moved to uncontaminated seawater, yolk-sac larvae showed no apparent elimination of the PCB, although there was a clear elimination of some PAHs as determined by BáP-derived radioactivity measurements (Solbakken *et al.* 1984). Metabolic breakdown of PAHs has also been detected in adult cod (Bell 1983, cited in Davies *et al.* 1984) in parallel field and tank studies: fish exposed to nominal levels (50 ppb) of benzo-α-pyrene (BαP) in water showed liver aryl hydrocarbon hydroxylase (AHH) values 5-40 times those of controls.

Oil

Early Life Stages

Crude oil extracts can affect energetic processes of cod eggs and larvae in addition to causing structural and developmental damage. One such extract (8 ppm of 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, however, and starved larvae showed reduced oxygen consumption when placed in the extract; the starved larvae became narcotized (Davenport *et al.* 1979). Exposure of cod eggs to the water-soluble fraction (WSF) of crude oils (50-150 ppb) does not significantly affect surface membrane permeability, nor is osmoregulatory ability of the embryo affected by these ecologically realistic concentrations (Mangor-Jensen and Fyhn 1985). Another laboratory exposure to the WSF of North Sea crude oil (50 ppb) strongly suppressed oxygen consumption by cod larvae at the time of final yolk absorption when the larvae begin to feed (5-7 days post hatch); no effect on oxygen uptake was seen in the eggs (Serigstad and Adoff 1985; embryo age was not reported). Kühnhold (1972, cited in Longwell 1977) found through laboratory exposures that cod eggs were most sensitive to crude oil during the first few hours post fertilization, corroborating Davenport *et al.* (1979). He also reported that by 10 hr after exposure, mortalities were significant; oil retarded development and in some cases delayed or prevented hatch. Larvae that did hatch showed a high level of abnormal development or abnormal swimming movements, and died within a few days.

Effects found in cod eggs and larvae exposed to WSFs (6 ppm) and suspensions (30 ppm) of crude oil, oil distillation fractions, and some low-boiling aromatics included several morphological abnormalities: (1) delay and irregularities in cleavage and development, (2) poor differentiation of the head region, (3) protruding eye lenses, (4) abnormally bent notochord, and (5) various levels of inhibition of hatching and assimilation of yolk (Lønning 1977). Such morphological disturbances will result in ultimate death of larvae, which in turn can lead to serious effects on fish population in polluted areas. Further work with the same life stages confirmed the variety of adverse morphological changes, and noted a significant decrease in growth rate (Tilseth *et al.* 1981) and a toxicant-concentration-dependent reduction in feeding (Solberg, Barth, *et al.* 1982; Solberg, Mangor-Jensen, *et al.* 1982; Solberg, Tilseth, *et al.* 1982). Such oil-induced disturbance of physiological and behavioral patterns would reduce feeding capability at onset of feeding, with consequent high mortality in the field.

Exposure to high concentrations (1,000 ppm) of oil dispersants blocked fertilization of cod eggs, and induced rapid cytolysis of the developing eggs and larvae (Lønning and Falk-Petersen 1978). At lower concentrations, effects were noted at 10 ppm, but not at 1 ppm. Cod embryos were more sensitive to dispersants than were sea urchin embryos (Lønning and Falk-Petersen 1978); the strong effects of such dispersants may be correlated with their solubility in seawater.

Larval cod exposed to sublethal amounts of the WSF of crude oil had reduced growth, morphological changes (malformed upper jaw), lower specific weight (neutral buoyancy), reduced feeding ability and swimming speed, and a serious disturbance of the swimming pattern. Larvae exposed to 4.1 ppm or higher did not recover their feeding ability within 24 hr of transfer to clean water (Tilseth *et al.* 1984). Cod larvae less than 20 mm in length are the size most harmed by exposure to 50±20 ppb of the WSF of oil (Foyn and Serigstad 1988). Effects of a 1- to 2-hr exposure of cod eggs to the WSF are not acute but long-term, leading to starvation of cod larvae. Moreover, there was no recovery from effects of exposure to the oil WSF when cod eggs or larvae were placed in clean seawater (Foyn and Serigstad 1988).

Illumination of a microlayer of Ekofisk crude oil by artificial sunlight was clearly shown to induce formation of toxic photoproducts, both polar and nonpolar. In larval cod exposed to the water phase of the illuminated oil, the 24-hr LC_{50} was reached when concentrations of polar components reached 1.0-2.0 ppm; this concentration led to cessation of feeding by the larvae at all tested concentrations (Solberg, Barth, *et al.* 1982).

In direct field-related observations, developing cod embryos exposed to hexane extracts of the sea-surface microlayer from five marinas located in the North and Baltic Seas showed site-related effects; extracts from two locations produced significant embryo mortality as well as severe deformities in live-hatched larvae. The greatest biological effect was seen with petroleum hydrocarbon concentrations greater than 180 ppb. Most extensive responses were observed in samples collected when a sheen was visible (Kocan *et al.* 1987).

Genetic evaluation of Atlantic cod eggs collected from the site of the *Argo Merchant* oil spill (Longwell 1977) showed that: (1) oil droplets and tar were found to adhere to half of the cod eggs examined; (2) 20% of the eggs were dead or moribund; (3) all dead eggs had cytological abnormalities; and (4) cod eggs spawned in the laboratory which were at the same developmental stages as egg samples from the spill site contained only 4% dead or moribund. Thus, eggs from the oil spill area had an apparently higher mortality rate than usual.

Goksoyr *et al.* (1991) exposed cod eggs, larvae, and juveniles to a WSF of North Sea crude oil (for 1-6 wk at 40-300 ppb), and examined them for induction of P450 which is biosynthesized "on demand" to catalyze the breakdown of pollutant hydrocarbons. Although exposure began during the egg stage, induction response was delayed until after hatching. The P450 induction response was dose-dependent, and recovery in clean water resulted in normalization of P450 levels (Goksoyr *et al.* 1991). Immunochemical response (*i.e.*, specifically the induction of P450c) in liver of juvenile cod and in homogenates of whole larvae reflected the exposure of those life stages to a WSF of crude oil in a dose-dependent way. Larvae and juveniles that were allowed to recover in clean seawater showed a recovery 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 (*i.e.*, heart rate, respiration, and gill ventilation rate amplitude) that did not occur until pollutant concentrations were close to lethal levels (Johnstone and Hawkins 1980).

Adult Stage

Mature cod chronically exposed to WSFs of crude petroleum: (1) depleted their stored energy of neutral lipids (after 24 wk at 100-200 ppb of total hydrocarbons) (Dey *et al.* 1983); (2) showed reduced growth, gill hyperplasia, filament fusion, increased skin pigmentation, and hepatic granulation; and (3) had increased gall bladder size (after 13 wk at 150-300 ppb) (Khan *et al.* 1981). These fish produced an oil-inducible MFO activity that was elevated four times higher in liver, and three times higher in gills, than in control fish (after 4 mo at 300-600 ppb) (Payne and Fancey 1982). Kidney tissue did not appear to be affected significantly. Further histological examination of tissues from such chronically-exposed cod (for 13 wk at 50-300 ppb) showed increased numbers of mucus-producing epithelial cells, capillary dilation, delayed spermatogenesis, and an increase of melanomacrophage centers in spleen and kidney (Khan and Kiceniuk 1984). No mortalities occurred in such cod (after 21 wk at 30-500 ppb), but significant reductions in food consumption and body condition were seen, gall bladders were enlarged during summer-autumn when fish were feeding actively, the rate of gametogenesis was reduced in male cod exposed to oil fractions in summer-autumn, and spermiation (release of mature sperm from the Sertoli cells) was delayed in fish exposed in winter-spring (Kiceniuk and Khan 1987). Chronic exposure of cod to crude oils, it was concluded, results in severely disabling lesions and reproductive impairment.

Cod captured close to oil platforms showed significantly higher levels of oil-inducible AHH in their livers than did fish caught in areas well away from oil activity (Davies *et al.* 1984). In the laboratory, detection thresholds for behavioral changes in cod upon sudden exposure to oil compounds were observed at levels as low as 0.1-0.4 ppb (Hellstrøm and Døving 1983). Changes noted were snapping, darting, "coughing," and restless swimming activity. In another exposure, cod avoided concentrations of total petroleum hydrocarbons at levels as low as 50 ppb, either in solution or an emulsion; mortality due to oil spills among large free-swimming fish has hardly ever been recorded, Bøhle (1983) concluded, because fish are able to move away from contaminated areas.

A risk assessment model (Spaulding *et al.* 1983) designed to assess probable effects of an oil spill on cod predicts that: (1) 60 days after a spill, most of the hydrocarbon effects on cod would have taken place (using the number of spawned eggs per spawning cycle as reference); (2) 41.5% of the spawned cod would have been adversely affected by oil concentrations in excess of 50 ppb; (3) a cumulative loss to the population would peak at 23.9% in the seventh year after the spill; and (4) winter and spring spills would have the greatest effects. Another assessment model (Spaulding *et al.* 1985) predicts that timing of maximum effect from an oil spill would follow spawning activity: spills occurring in winter and spring (March, in particular), again, would have the greatest effects on cod.

Disease in Oil-exposed Cod

Of 100 live Atlantic cod collected from Halifax Harbor, Nova Scotia, 73 had histopathological liver lesions (Freeman *et al.* 1983). "Fatty change," a degenerative process in the liver, can be induced by chronically ambient organic contaminants or heavy metals (Freeman, Uthe, *et al.* 1981). Considered with other pathological signals, exceptional accumulation of liver lipids suggests that body burdens of PCBs, pesticides, or other organic contaminants constitute a substantial component of pathogenesis (Freeman *et al.* 1983). Pathological changes in cod liver seem to become progressively greater with increasing size of the fish, although temporal variations in contaminant levels in tissues of cod may continue to occur (Misra *et al.* 1988).

A comparison of fitness in cod collected from relatively clean and polluted sites showed slower-than-normal reaction times in fish from the polluted site (Olofsson and Lindhal 1979). In a laboratory exposure, hemoprotozoan-infected cod reacted more sensitively to petroleum hydrocarbons than did noninfected fish (Khan 1987), as measured by poor body condition, excessive mucus secretion by gills, retarded gonadal development, and greater mortality.

Finally, 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).

HADDOCK

Very little information was found for pollutant effects on haddock, whether experimental exposures or field studies.

Heavy Metals

A short acute exposure of 4-wk-old haddock larvae to copper (for 18 hr at 500 ppb) produced toxic effects, including increased mortality (50% as compared to 15% for controls) and severe olfactory lesions (Bodammer 1981).

The mean concentration factor for cesium-137 was 58 for haddock caught in the North Sea, well below the value recommended by the International Atomic Energy Agency as the limit for human dose assessment (Steele 1990).

Organic Contaminants

There appears to be little problem with organic contamination in haddock, other than events related to oil. Haddock caught in offshore fisheries have very low levels of PCBs

(Capuzzo *et al.* 1987). After Norway banned DDT and PCBs, haddock and cod from a Norwegian fjord had clearance rates for these compounds that were slower than those found for wolffish, sea scorpion, a European wrasse, and lemon sole (Skåre *et al.* 1985).

Levels of oil-inducible AHH activity were higher in haddock collected from areas around oil platforms than in fish from clean areas (Davies *et al.* 1984). These data are the first to indicate that oil in sediments around oil platforms may be biochemically available to fish in the area, probably *via* the food chain.

The greater the aromatic content of an oil dispersant, the greater its toxicity to unfed haddock larvae (Wilson 1977). Larvae were more susceptible to toxic effects of a dispersant immediately after first feeding. Numerous dispersants have been developed, however, that are 2-3 orders of magnitude less toxic than kerosene-based dispersants available earlier.

YELLOWTAIL FLOUNDER

Organic Contaminants

A fisheries assessment during the 1980s indicated minimum displacement of the yellowtail flounder fishery, along with plaice, cod, and redfish, by development of offshore oil-drilling platforms and by production of oil (Kulka 1991). The major fisheries were, in large part, located away from oil development areas off Newfoundland. Yellowtail flounder eggs collected during the first 3 days following a gasoline spill near Falmouth, Massachusetts, however, had an 81% mortality rate (13 of 16 eggs died) (Griswold 1981).

Unidentified Contaminants

The highest rate of finrot disease in yellowtail flounder was seen in fish collected from the New York Bight (Ziskowski *et al.* 1987). Of 15 fish species collected during 1979-82, 33% of all larval cestode cysts were found in yellowtail gut; liver lesions and skeletal anomalies could also be grossly observed (Despres-Patanjo *et al.* 1982; Murchelano *et al.* 1986). "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" (Murchelano *et al.* 1986). With the exception of lymphocystis and finrot in winter flounder, no distinct trends could be discerned in the distribution of diseases whose prevalence appears to be linked with pollution.

POLLOCK

Organic Contaminants

Although laboratory experiments showed that pollock were able at least partially to catabolize the PAH phenanthrene, and to excrete breakdown products from that PAH, no observations were made on physiological effects in the fish itself (Solbakken *et al.* 1980). In field work, pollock eggs from the area of the *Argo Merchant* oil spill, collected shortly thereafter (which was during the pollock spawning season), had oil adhering to the outer membrane, and showed evidence of cytogenetic damage (Longwell 1977).

AMERICAN PLAICE

Heavy Metals

American plaice collected off Newfoundland and Labrador had arsenic concentrations in muscle tissue that were similar to levels in sediments, but lower than concentrations in the local shrimp upon which they prey. The plaice arsenic levels were much higher than those found in other fish species (redfish, Atlantic cod, turbot) from the same area (Kennedy 1976).

Organic Contaminants

Elevated levels of polycyclic organochlorines (pesticides) found in livers of male American plaice sampled in the North Sea during winter were associated with their site of collection. Fish with high contaminant body burdens in that northern European habitat were found near areas of major riverine input and other sources of human-induced pollution (Knickmeyer and Steinhart 1990). In eastern Canadian waters, however, inducible mono-oxygenase 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.

WINTER FLOUNDER

Heavy Metals

Concentrations of heavy metals in muscle tissue of winter flounder caught in the New York Bight and Long Island Sound were relatively low, and did not reflect the high levels found in some sediments, namely those from Christiansen Basin and the "Mudhole" (30 km south-southeast of the basin, in the northern part of the Hudson Shelf Valley) (Reid *et al.* 1982). Concentrations of heavy metals in sediments of the New York Bight apex are 10-100 times greater near waste disposal areas than in uncontaminated sediments (Carmody 1973).

In a comparison of two sites in the west-to-east pollutant gradient in Long Island Sound, Hempstead Bay (westernmost) is considered heavily polluted relative to waters off Shoreham, New York (mid-sound), based on metals-in-sediment data, yet copper, manganese, and zinc concentrations in livers of winter flounder caught at those stations were twice as great for Shoreham as for Hempstead (Greig and Wenzloff 1977). There thus appears to be no general trend of heavy metal concentrations in liver in relation to heavy metals in sediment (MacDonald 1991). This observation also seems to be true for PCB levels in liver of windowpane (Greig *et al.* 1983) which were not related to an assumed pollution gradient. No data are available from fish moving from a polluted area to a cleaner one.

In related experimental studies, the order of sublethal heavy metal toxicity (for 2-5 mo at 10 ppb) for adult winter flounder was $CdCl_2 > HgCl_2 > AgNO_3$ (Calabrese *et al.* 1977). Heavy metal exposure either elevated (mercury) or depressed (cadmium) gill respiration. Mercury accumulated in tissues of winter flounder that were exposed to this heavy metal, whereas no statistically significant cadmium was taken up in tissues of fish in an analogous exposure. Mercury, but not cadmium or silver, provoked statistically significant hematological responses. Cadmium, however, is the most potent inducer of the transcription gene for metal-binding proteins (Chan *et al.* 1989), the principal function of which appears to be maintenance of homeostasis for the essential trace metal zinc (Garvey 1988). Exposure of winter flounder to low concentrations of this heavy metal induced several significant metabolic responses: (1) ligand sensitivity was lowered in kidney and heart enzymes; (2) enzyme induction occurred 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 also a loss of sensitivity to normal metabolic control (magnesium's enhancement of ligand affinities); and (4) in liver, glycolysis and shunt activity increased (Gould 1977). These same phenomena were observed in mercury-exposed flounder, but to a lesser extent, and silver-exposed flounder showed very little effect (Calabrese *et al.* 1975, 1977). No uptake of cadmium into either blood or gill tissue was detected, but mercury increased over control levels in both blood and gills in a dose-dependent manner (Calabrese *et al.* 1975). Recovery from hematological changes seen in winter flounder exposed to mercury alone was virtually complete in the 10-ppb-exposed group after a subsequent 60-day recovery period, and partial recovery was seen in the 20-ppb-exposed group (Dawson 1979).

In developing embryos of winter flounder, no effect was produced on percent total hatch by cadmium (0-1,000 ppb), nor by the interactions between cadmium and salinity, cadmium and silver, and silver and salinity (Voyer *et al.* 1982). Cadmium did, however, influence the viable hatch response, which decreased with increasing cadmium concentrations. The most toxic effect of cadmium on viable hatch was at 10‰ salinity, the lowest tested, and became less significant with increasing salinity. Addition of silver to cadmium solutions decreased cadmium toxicity; silver alone exerted no lethal effect over the concentrations used.

Absorption of leucine from food is diminished in both winter and summer flounders taken from the polluted water of Sandy Hook Bay, New Jersey, as compared to fish taken from cleaner areas (Farmanfarmaian *et al.* 1981). This effect was enhanced by low levels of mercury. Methylmercury, however, is a less effective inhibitor: inorganic mercury (at 0.4 m*M*, 108 ppm) completely abolished sodium transport ability in flounder urinary bladders *in vitro*, an effect that persisted during the 110-min period of experimentation (Farmanfarmaian *et al.* 1981). In an analogous experiment with methylmercury (at 0.4 m*M*, 100 ppm), Renfro *et al.* (1974) found only a transient inhibition of sodium transport. Mercury also blocks essential potassium movement in the urinary bladder (Venglarik and Dawson 1986). Exposure of winter flounder to methylmercury had no effect on osmoregulation, ion regulation, or blood volume regulation, nor was there any effect on transepithelial electrolyte transport in gill or intestine (Schmidt-Nielsen *et al.* 1977), but in bladder and kidney more energy was expended to perform that function in methylmercury-exposed winter flounder than in control fish. Musch *et al.* (1990) reported that in preparations of winter flounder intestine, inorganic mercury (100 μM), organic mercury (*p*-chloromercuribenzene at 1 μM), and organic arsenic (oxophenylarsine at 250 μM) all inhibited adenosine triphosphatase (ATPase) and decreased cell potassium, and inorganic mercury decreased uptake of tyrosine. The conclusion was that mercurial and arsenical effects on tyrosine absorption are due to inhibition of Na-K-ATPase, which thus decreases the driving force for cellular uptake of tyrosine.

Concentrations of silver above 54 ppb in a flow-through bioassay (for 18 days at 54-386 ppb) 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 ppb 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 exposure to 386 ppb were significantly smaller than measurements at lower concentrations.

Electron microscopic examination of olfactory organs of copper-exposed larval winter flounder (for 18 hr at 500 ppb) showed moderate-to-severe cytopathologic lesions (Bodammer 1981). Charney *et al.* (1988) reported that copper inhibited chloride absorption in *in vitro* preparations of winter flounder intestine, and damped the stimulatory effect of higher pH; concentrations as low as 50 μM were effective. Inhibition was not reversed by addition of a copper chelator. Zinc, on the other hand, stimulated chloride absorption at concentrations between 10 and 50 μM. Shears and Fletcher (1983) demonstrated that Cu^{++}, Ni^{++}, Fe^{+++}, Co^{++}, Cd^{++}, and Cr^{++} all significantly decreased the absorption of the essential trace element Zn^{++} across the wall of the intestine.

Organic Contaminants

PCBs

PCB concentrations in winter flounder are primarily derived from sediment. A food chain model demonstrates that uptake across the gill is exceeded by dietary uptake of PCBs; contaminated prey species provide most of the PCBs observed in flounder (80-95%). Assimilation efficiency of PCBs declines from high values for trichlorophenyl to low values for the more highly chlorinated homologues (Connolly 1991).

Fin erosion, seen most strikingly in winter flounder, appears to result from exposure of demersal fishes to contaminated sediments. PCB concentrations in muscle, liver, and brain tissues were greater in fishes with fin erosion from contaminated sites (primarily winter flounder) than in fish from reference sites (Sherwood 1982). The erosion pattern of 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. Sherwood (1982) speculates that increases in liver size and/or lipid content may be a response to DDT and/or PCB exposure.

PCB levels in winter flounder liver were significantly correlated with body fat content (Reid *et al.* 1982). "If fat content is inversely related to environmental stress, [no direct] relation between environmental contamination and PCB body burdens can be drawn, and may contribute to the observed absence of dramatically elevated burdens in the inner [New York] Bight." The greatest concentration of PCBs in winter flounder flesh (0.56 ppm wet weight) in the New York Bight was found 21 km south of the eastern end of Long Island. PAHs, predominantly phenanthrene, ranged from 28 to 246 ppb wet weight.

Winter flounder collected from three Southern New England areas with differing degrees of PCB and PAH contamination--New Bedford Harbor, Massachusetts, and Gaspee Point and Fox Island, Rhode Island--reflected site values in having analogous varying degrees of PCB-contaminated liver, with New Bedford Harbor greater than Gaspee Point greater than Fox Island. Liver EROD activity was the same at all three sites, but P450 was significantly higher in the New Bedford fish (Elskus *et al.* 1989). Data suggest that P450s catalytic activity (for EROD) is being competitively inhibited at New Bedford Harbor, possibly by some congeners of PCBs (Gooch *et al.* 1989).

Work in Long Island Sound showed concentrations of PCBs in winter flounder gonads to be greatest (0.73 ppm wet weight) in the months just before spawning, as compared to levels (0.056-0.36 ppm) in other months (Greig and Sennefelder 1987). After spawning, PCB concentrations in gonads decreased to very low levels (0.03-0.08 ppm). Liver PCB concentrations declined somewhat (0.33-0.60 ppm) before and during the spawning season, returning to higher levels (1.1-2.3 ppm) after the spawning season. Overall, PCB levels were higher in gonad and liver samples from more polluted sites in Long Island Sound.

Sexually mature female winter flounder showed lowered EROD activity and P-450E levels, thought to result from hormonal activity which suppresses induction of P450E, the activation catalyst for EROD (Förlin and Hansson 1982, cited in Elskus *et al.* 1989). Winter flounder collected in heavily urbanized areas of Long Island Sound having higher PCBs in sediments (New Haven and Norwalk, Connecticut, and Hempstead, New York) tended to have lower reproductive success when spawned in the laboratory, than did flounder collected from less urbanized sites (Nelson *et al.* 1991). In that same study which compared several different sites in the sound, winter flounder embryos from New Haven had the most abnormalities and the lowest percent viable hatch. Nelson *et al.* (1991) also found that flounder with high concentrations of PCBs in liver (Boston) had small larvae. Black *et al.* (1988) reported that eggs of winter flounder from New Bedford Harbor were significantly higher in PCB content (39.6 ppm dry weight) than those from Fox Island (a relatively clean area in Narragansett Bay), and that larvae hatched from New Bedford eggs were smaller in length and weight. There was an inverse relationship between the PCB content of the eggs and the length or weight at hatch.

Experimental work has shown that other contaminants can affect PCB assimilation. In winter flounder exposed to a PCB (for 24 hr at 1 ppm) in the presence or absence of added cadmium (200 ppb), fish dosed with added cadmium had significantly lower PCB in their liver and gills than did those dosed with PCB alone (Carr and Neff 1988). Weis *et al.* (1992) exposed (2 wk) developing winter flounder embryos to varying concentrations of treated wastewater effluent, or various extracts thereof. PCBs were present in all batches of effluent used. The highest mortality and a precocious but diminished hatch occurred in embryos treated with undiluted effluent; the organic-acid fraction and organic-bases fraction had reduced mortality, and the neutral-organics fraction had mortality rates less than that of the controls. Larvae grew faster in the organic-acid fraction; their increased growth rate is thought to be due to stimulatory metabolism in response to sublethal toxic challenge (*i.e.*, hormesis).

PAHs

In male and female winter flounder collected over a 2-yr period fron a relatively nonpolluted area in Nova Scotia waters, seasonal variation (about 10-fold) in hydrocarbon-inducible enzyme activity was less than that caused by environmentally realistic levels of pollutants (Addison *et al.* 1985; Edwards *et al.* 1988). Liver MFOs in this species, therefore, might be used to indicate environmental contamination.

High levels of PAHs have been found in Boston Harbor sediments (Malins *et al.* 1985). Pathological conditions seen in liver tissue excised from winter flounder collected from that area were construed as arising from PAH-induced genetic mutations leading to tumor formation (McMahon, Huber, Moore, *et al.* 1988; McMahon, Huber, Stegeman, *et al.* 1988). Experimental work led to substantiation: winter flounder fed chlordane- and BαP-contaminated food developed proliferative lesions similar to cholangiocellular carcinomas in winter flounder taken from Deer Island Flats in Boston Harbor (Moore 1991). Both BαP and BαP metabolites produced by polychaete worms accumulated in liver tissue when winter flounder were fed contaminated worms (McElroy and Sisson 1989). Scientists caution, therefore, that risk assessments for predators must account for metabolites produced by prey as well as the parent compound.

Cytochrome P450 was induced in winter flounder in the laboratory by injection of the PAH β-naphthoflavone; flounder collected from Boston Harbor and Buzzards Bay, Massachusetts, had the same enzyme. Other hydrocarbon-inducible enzyme activities (EROD and AHH) were elevated in both injected and field-collected fish (Stegeman *et al.* 1987). Immunohistochemical treatment of liver tissue from winter flounder further revealed evidence of histopathological conditions in liver, the hydrocarbon-inducible P450 enzyme. Presence of P450 was associated with liver disease in winter flounder taken from Boston Harbor (Smolowitz *et al.* 1989). The study shows that P450 could play a role in production of a mutagenic agent from environmental chemicals taken up by fish. Corroborative work on winter flounder collected near Mount Desert Island, Maine, indicated that elevated liver AHH and EROD activities were associated with a band in electropherograms of liver microsomal preparations corresponding to the position of cytochrome P450 (Foureman *et al.* 1983). Fish with low AHH and EROD activities showed the band only faintly.

Pesticides

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 amounts of DDT (1-2 ppb) (Smith and Cole 1973). No such effects were seen in eggs from flounder similarly exposed to dieldrin, nor were residues of either insecticide detected in milt of exposed or control male winter flounder.

The LC_{50} for chlordane, injected intraperitoneally in winter flounder, was 11,000 mg/kg (Moore 1991). Chlordane in high doses induced severe liver damage, and at subacute doses produced macrophage aggregation and a ersistence of necrogenic effects in liver. Dimethylsulfoxide increased toxic effects of chlordane almost 200-fold.

Pesticide levels were elevated in winter flounder from a tributary of Buzzards Bay, indicating significant levels of contamination in this area. Winter flounder collected near Lynn, Massachusetts, contained heptachlor in amounts close to the FDA limit for humans, and also contained elevated levels of DDT (0.117 ppm wet weight) (Connolly 1991). Greatest concentrations of chemical contaminants were found in coastal harbors and industrialized centers (MacLeod *et al.* 1981), whereas offshore areas had very low levels (Connolly 1991).

Oil

Winter flounder were exposed to oil-contaminated sediment either weathered for 1 yr or freshly deposited (1 l of oil in 45 kg of sand). After 4 mo, livers from fish which had been exposed to weathered oiled sediment had a cytochrome P450 induction rate that was seven-fold that of unexposed control fish; livers from fish which had been exposed to freshly oiled sediment had an induction rate that was 13-fold higher (Payne and Fancey 1982). BαP seemed to be a good inducer of cytochrome P450.

In 1984, examinations of winter flounder collected from the site of an oil spill and from a reference site showed that reliance on 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). Later, however, the same research team, using oiled sediments under a controlled laboratory exposure, found that biomarkers indicating exposure to oil were (in order of decreasing sensitivity): liver MFO activity, liver condition index (liver weight to total body weight), kidney MFO activity, spleen condition index, and muscle protein and water content. The liver lipid and glucose levels and the condition indices for gut, kidney, testis, and whole fish were not affected at any exposure level (Payne *et al.* 1988). *In vitro* work with skin of winter flounder exposed to crude oil suggests an added capacity for increased mucogenesis in direct response to stress associated with environmental conditions (Burton *et al.* 1984). 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 detoxication mechanisms are overwhelmed (Hutt 1985, cited in Payne *et al.* 1988).

High numbers of liver macrophage aggregates were reported as a response to exposure to stressful environmental stimuli, including chemical contamination (Wolke *et al.* 1985). Yet Payne and Fancey (1989) reported that exposure of winter flounder for 4 mo to varying concentrations of crude oil in sediments **reduced** numbers of melanomacrophage centers in livers, finding that liver hypertrophy increased with increasing oil exposure. Splenic atrophy occurred only at the two highest levels of exposure (250 or 500 ml of oil in 45 kg of sand). Exposure to oiled sand was shown to affect the immune respose in winter flounder by reducing the number of phagocytic cells available to ingest foreign particles (Payne and Fancey 1989).

Khan and Kiceniuk (1983) observed that there are fewer parasites in fish exposed to oil, and suggested that lowered parasitism might be attributed to toxicity induced by WSFs of crude oil and/or modification of the gut environment. 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). Juvenile and adult winter flounder, some infected with the blood parasite *Trypanosoma murmanensis*, were exposed to sediment contaminated with crude oil (for 6 wk at 2600-3200 ppm) 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. Deaths were associated with severe fin and tail necrosis, erratic swimming behavior, and failure to bury in the sediment. Moreover, blood chemistry changed: hemoglobin (Hb), hematocrit (Hct), plasma protein, and condition factor were all lowered. Clubbing of gill tissue, mucus hypersecretion, and congestion and enlargement of the spleen were also observed. Exposure to oil increased the incidence of infection and death, whether the exposure occurred before or after infection with *Trypanosoma* (Khan 1987). These results were corroborated 4 yr later, with additional data adding reproductive impairment and presumptive immunosuppression to the list of effects (Khan 1991). In plasma of sexually mature male winter flounder exposed to crude oil, total concentrations of the 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.

Pollution-linked Disease

Circumstantial evidence linking diseased fish to polluted habitats is abundant: proliferative lesions in endocrine, exocrine, respiratory, sensory, excretory, and digestive organs, and alteration of plasma protein were found to be characteristic of winter flounder from moderate to highly contaminated inshore areas. Concentrations of PAHs, PCBs, other organic compounds, and heavy metals associated with sediments were elevated in urban embayments. Liver disease was found to be absent in populations from uncontaminated offshore areas. Tissue concentrations of PCBs reflected those detected in sediments. Degree of sediment chemical contamination and disease suggest a causal interrelationship (Gardner *et al.* 1989).

Winter flounder collected from the inner New York Bight had statistically higher incidences of finrot disease when compared to fish collected either offshore of the bight or within Massachusetts Bay (Ziskowski *et al.* 1987). A Boston Harbor field study, in which tumors were not found in fish smaller than 32 cm in length, revealed a pattern in liver pathology: a progression from necrotic lesions to neoplasia, suggesting pollutants as likely inducers of the lesions (Murchelano and Wolke 1991). 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 gradient of pollution (Turgeon and O'Connor 1991).

Boston Harbor sediment extract, injected peritoneally, was acutely toxic to winter flounder; after 10 days, perivascular edema was observed in survivors (Moore 1991). Gardner and Yevich (1988) reported that fish exposed for 90-120 days to sediment from Black Rock Harbor, Bridgeport, Connecticut, developed neoplastic or proliferative lesions in kidney, olfactory, and lateral line sensory tissues, in gastro-intestinal tract, and in buccal (cheek) epithelium; cytopathology and cell necrosis were detected in the pituitary, all related to organic contamination. In the field, winter flounder collected from Black Rock Harbor and New Bedford Harbor had similar lesions.

In a long-term study of several sites in Long Island Sound, abnormalities observed in winter flounder embryos included evidence of cytotoxicity, chromosome damage, slower developmental rates, and cell necrosis (Perry *et al.* 1991). Embryos of fish from Morris Cove, New Haven, Connecticut, were usually the most aberrant, while embryos from Hempstead and Shoreham, New York, and from two Boston Harbor sites showed subtle indications of abnormality. New Haven sediments are contaminated with high levels of PCBs, aromatic hydrocarbons, chlorinated hydrocarbons, and heavy metals (Gronlund *et al.* 1991). High prevalences of liver lesions, blood cell abnormalities, liver DNA damage, and liver neoplasms, and high concentrations of organic compounds, heavy metals, and PCBs in gonads of winter flounder from New Haven have been found (Greig and Wenzloff 1977; Greig and Sennefelder 1987; Gronlund *et al.* 1991).

Incidences of small inclusions (micronuclei) in red blood cells were elevated six-fold in winter flounder from the New York Bight apex as compared to fish from the inshore Gulf of Maine and Block Island Sound, and two-fold in those from Georges Bank and Long Island Sound (Hughes and Hebert 1991). Inshore New Jersey and Virginia fish had significantly higher frequencies of micronuclei than fish from the Gulf of Maine and Block Island Sound. There were higher frequencies of micronuclei in flounder from Hempstead and Shoreham, New York, as compared to most other sites in Long Island Sound. Erythrocyte micronuclei 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 heavy metals and PAHs (Carmody *et al.* 1973; MacLeod *et al.* 1981; Environmental Protection Agency 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). 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. Higher incidences suggest a link with environmental pollution.

High mean blood lymphocyte counts in winter flounder were correlated with liver necrosis and suspected levels of sediment chemical contamination; winter flounder collected from Boston Harbor had greater 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.

Winter flounder collected from lower Narragansett Bay produced the smallest yolk-sac larvae with the lowest survival rate, while fish collected at Madison, Connecticut, produced the largest yolk-sac larvae with the highest survival rate (Buckley *et al.* 1991). Also, within Long Island Sound, survival was higher in the Madison group than in the Morris Cove group. These results are consistent with observations of embryonic development, suggesting that reproduction of winter flounder at the Morris Cove site has been compromised by high contaminant levels (Nelson *et al.* 1991).

Further circumstantial evidence associating pollutants with abnormalities has been reported for Massachusetts Bay. Biochemical variables such as hepatic and pectoral fin ascorbic acid concentrations, hepatic glycogen and lipid levels, plasma glucose concentrations, brain serotonin and norepinephrine concentrations, and the concentration and ratio of various free amino acids in muscle tissue were significantly lower in winter flounder from

Boston Harbor (Carr *et al.* 1991). The histological difference between Boston Harbor and Plymouth Beach (reference site) fish was the high prevalence of degenerating hepatic parenchymal cells in Boston Harbor fish. Low tissue concentrations of ascorbic acid and hepatic glycogen had significant statistical associations with the presence and severity of these hepatic lesions.

Mixed Contaminants

Halifax Harbor, Nova Scotia, sediments, which are enriched with heavy metals (arsenic, cadmium, copper, mercury, lead, and zinc), organic carbon, and PAHs, are considered to have caused the necrogenic effects (hepatocyte basophilia, macrophage aggregation, and hepatic epithelial vacuolation) observed in livers of winter flounder caught in the harbor (Tay *et al.* 1991).

Of three study sites in Long Island Sound, heavily urbanized New Haven had the greatest concentrations of aromatic hydrocarbons and PCBs in sediment, and the highest prevalences of histopathological changes and DNA alterations in livers of winter flounder (Gronlund *et al.* 1991). The liver macrophage aggregate index (Wolke *et al.* 1985) was significantly higher in winter flounder from New Haven as compared with those from Niantic, Connecticut. No differences in contaminant concentrations in fish, however, or in frequencies of red blood cell micronuclei, were found between sites (Gronlund *et al.* 1991). None of the sites sampled had contaminant levels or prevalences of lesions as high as were found at Boston Harbor or Raritan Bay, New York/New Jersey.

Levels of organic and inorganic contaminants in sediments and fish tissue were examined to evaluate correlations between biological effects and contamination (Zdanowicz *et al.* 1986). PAHs found in stomach contents of winter flounder appear to be sediment derived. Concentrations of PCBs were greater in the stomach than in sediments (with the exception of Salem and Boston Harbors), an indication that the chlorinated hydrocarbons were accumulated by prey organisms. Concentrations of PCBs and chlorinated hydrocarbons increased in flounder livers in relation to both stomach contents and sediment. Some pathological conditions had distributions similar to contaminant distribution. Distribution of certain contaminants and certain types of lesions appear to be related, but it is unclear whether the compounds or combination of compounds that were examined may be causing these maladies.

WITCH FLOUNDER

No references were found in the literature of any pollutant effects in witch flounder that are attributable to heavy metals or organic contaminants.

ACADIAN REDFISH

Heavy Metals

No adverse biological effects were found in Acadian redfish having a mean concentration of 800 ppb of arsenic in muscle tissue (Kennedy 1976).

Organic Contaminants

An 8-yr study of Acadian redfish showed that oil-exploration activity did not adversely affect the Acadian redfish stock (Kulka 1991). Another contaminant widely distributed in the environment is di-2-ethylhexyl phthalate (DEHP), a manmade organic compound. Concentrations of DEHP found in Acadian redfish were less than 0.001 ppm in both fatty and wet-weight tissues; no adverse effects were reported (Musial and Uthe 1980). The low levels found in Acadian redfish (and American plaice) from the deeper parts of the Gulf of Maine, as compared with much higher levels in Atlantic herring and Atlantic mackerel, suggest that DEHP contamination in fish is an inshore phenomenon.

WHITE HAKE

White hake collected from Boston Harbor during 1979-83 had a 1.89% prevalence of finrot disease; winter flounder collected from this area during the same time period had a prevalence of only 0.71% (Murchelano *et al.* 1986).

WINDOWPANE

Heavy Metals

In a 1975 field collection from Delaware Bay, windowpane had detectable levels of mercury in muscle tissue, but no mercury was found in water, detritus, algae, or shrimp from the environment of the fishes (Gerhart 1977). There was no correlation found between mercury body burdens and the food chain or habitat.

Laboratory exposure of windowpane to mercury (for 2 mo at 10 ppb) increased plasma sodium levels, and work with field-collected fish showed higher blood parameters (Hct and Hb) at the most polluted station (Dawson 1990). Field results indicate an increase in hematopoiesis. These observations suggest an attempt by the windowpane's system to compensate or even overcompensate for an imposed metabolic stress. The same laboratory exposure of windowpane to mercury produced abnormal localized swellings in gill, and some fragmentation of cellular membranes (Pereira 1988)--changes that may affect gill function.

Organic and Unidentified Contaminants

No relation was found between windowpane liver concentrations of PCB and a pollution gradient in Long Island Sound, nor was there any general trend of heavy metal concentrations in liver in relation to heavy metals in sediments (Greig *et al.* 1983).

A high incidence of fin erosion was observed in windowpane, winter, and yellowtail flounders exposed to materials dumped in the New York Bight (O'Connor 1976), with winter flounder having the highest incidence.

Numbers of red blood cell micronuclei in field-collected windowpane were 2-3 times higher in fish from the most polluted site in Long Island Sound than in fish from the reference station (Longwell *et al.* 1983).

Field collections of both planktonic eggs and adults of windowpane and winter flounders across a pollutant gradient in Long Island Sound showed mitotic abnormalities in specimens from more polluted coastal areas. Longwell *et al.* (1992) consider their findings to represent a "considerable...cumulative effect on...early-life survival."

SILVER HAKE

Heavy Metals

Detectable concentrations of mercury in muscle of silver hake could not be related to feeding habits or environment (Gerhart 1977).

Organic Contaminants

Although computer searches revealed no reports on effects of organic contaminants on silver hake either in the Gulf of Maine or elsewhere, some papers on the closely related European whiting (*Merlangius merlangus*) may be relevant. In particular, PCB, DDE, and dieldrin contamination in ovaries from North Sea whiting was significantly correlated with reduced viable hatch (von Westernhagen *et al.* 1989, cited in Dethlefsen 1989).

European whiting captured near offshore drilling platforms, which use oil-based muds for operation, showed no significant difference in AHH activity from whiting captured in control areas. Laboratory exposure of three fish species (whiting, Atlantic cod, and haddock) to 50 ppb of BαP showed that whiting had the least response, with AHH activity only 0-3 times that of control fish which had much greater variability (Davies *et al.* 1984).

RED HAKE

Heavy Metals

No correlation was found between detectable mercury concentrations in red hake muscle tissue and the fish's eating habits or its environment (Gerhart 1977).

Unidentified Mixed Contaminants

A significant fraction (7.5%) of red hake collected from the New York Bight had external ulcers, whereas those from outside the bight did not have lesions (Murchelano and Ziskowski 1979).

Although the general experience of fisheries scientists has been that finrot is almost never seen in red hake, collections of that species taken from Boston Harbor during 1979-83 had a significant (3.4%) prevalence of finrot disease (Murchelano *et al.* 1986). During this same survey, white hake had 1.89% and winter flounder only 0.71%, anomalously low for the latter species.

OCEAN POUT

No references have been found in the literature of pollutant effects in the ocean pout that are attributable to heavy metals or organic contaminants.

POLLUTANT EFFECTS SUMMARY

The highest concentrations of chemical contaminants are to be found in coastal, industrialized/heavily urbanized, and waste disposal areas. Such areas are also the spawning and nursery habitats for many important commercial fishes. The early life stages of these fishes are most susceptible to toxicants, 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 heavy metals or organic contaminants (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 (*Platichthys flesus*) did not. In winter flounder, PCB body burdens were accumulated from prey species in the sediments, rather than from the water column (Connolly 1991).

More immediately associated with pollutant exposure, abnormally high levels of detoxifying enzymes may signal (for a short time) exposure to organic contaminants (Addison and Edwards 1988), although this response varies with sex and gonadal maturation (Spies *et al.* 1988; Spies and Rice 1988; George 1989). Similarly, induction of metal-binding proteins may signal (for a short time) exposure to heavy metals (Overnell and McIntosh 1988; Garvey 1988; Fowler and Gould 1988; George 1989).

Different PCBs, however, have sometimes conflicting effects (Hansen *et al.* 1983), with some congeners inhibiting others (Gooch *et al.* 1989). 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 the PCB content of eggs and the length and weight of larvae at hatch. Goksoyr *et al.* (1991) have shown that in early life stages of a fish, the normal protective production of enzymes that break down organic contaminants 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 male, fish (Marthinsen *et al.* 1991). The fattier the liver, the slower the clearance rates of these fat-soluble toxicants (Skåre *et al.* 1985).

Biochemical synthesis of a specific enzyme system 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 Atlantic cod to crude oils produces severely disabling lesions and reproductive impairment (Khan and Kiceniuk 1984; Kiceniuk and Khan 1987).

In the case of organic contaminants, cadmium strongly depresses several detoxifying enzyme systems (George 1989), but appears to depress PCB uptake in winter flounder (Carr and Neff 1988). Moreover, in the European flounder, exposure to diesel oil produced no increase in detoxifying activity when copper was added to the oil (Addison and Edwards 1988). Overloads in marine animals of even essential heavy 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 bivalve mollusk, the sea scallop (Gould *et al.* 1988; Fowler and Gould 1988).

It is certain that pollutant stress affects recruitment to, and overall health of, fish stocks to an indeterminate degree that obviously varies erratically with time and site. Usually it is not possible to attribute specific effects observed in the field to specific pollutants (Wolfe *et al.* 1982), although abnormally active detoxifying systems in fish, including metal-binding proteins, suggest contamination by organic contaminants and/or heavy metals. Recruitment of fish stocks is also undoubtedly strongly influenced by the lowered spawning biomass brought about by overly heavy fishing pressure (Cohen *et al.* 1991).

Finally, human activities have altered the balance of our living marine resources for the worse; we must hope that the situation is not yet irremediable. Certainly, enhanced wild stocks and culturing of fishes and invertebrates must be predicated on clean, uncontaminated coastal waters.

SEWAGE SLUDGE AND INDUSTRIAL WASTE DUMPING

***Bradley W. Barr*[52] *and Stuart J. Wilk*[53]**

Offshore disposal of domestic (*i.e.*, sewage) and industrial (*e.g.*, acids) wastes normally involves barging sludge (*i.e.*, solids that settle during sewage treatment) and either containerized or noncontainerized industrial wastes to a designated, regulated site, and "dumping" them there. There are no designated sewage sludge dumpsites and only one designated industrial waste dumpsite in the Gulf of Maine--the Massachusetts Bay Industrial Waste Site (MBIWS), located approximately 19 nautical miles off Boston in 90 m of water. In all probability, though, there are other, nondesignated, unregulated sites in the Gulf of Maine where where industrial wastes have been dumped.

This section begins with a description of the MBIWS. For comparative purposes, two industrial waste dumpsites in the Middle Atlantic Bight--106-Mile Deepwater Industrial Waste Site (better known as Deepwater Dumpsite 106, or DWD-106) and New York Bight Acid Waste Site (AWS)--are also described.

Although there are no sewage sludge dumpsites in the Gulf of Maine, for decades sewage sludge was discharged into Boston Harbor from the Deer Island and Nut Island Waste Water Treatment Plants. According to the Massachusetts Water Resources Authority (1987), a daily average of 87 dry tons of anaerobically-digested sludge was discharged into the harbor. Such discharges were eliminated in December 1991.

While the fate of these materials was not definitively known, and given that they were discharged only on the outgoing tide, it is presumed that the majority, although not all,

[52] Stellwagen Bank National Marine Sanctuary, Plymouth, MA.
[53] National Marine Fisheries Service, Highlands, NJ.

ultimately was deposited in Massachusetts Bay and incorporated into sediments. Quality of the discharged sludge was variable, but generally contained additional concentrations of toxic heavy metals and of contaminants such as PAHs and PCBs. Given the multiple sources of such contaminants (including discharge from combined sewer overflows and liquid wastewater effluents, atmospheric deposition, and ocean dumping of industrial wastes, among others), it has been difficult to isolate a specific cause-and-effect relationship between discharge of sludge and effects to the Massachusetts Bay/Cape Cod Bay ecosystem and the contained living marine resources.

There has been one comprehensive study of the ability of living marine resources to recover from effects of sewage sludge dumping following the cessation of such dumping: the NEFSC's study of the New York Bight 12-Mile Sewage Sludge Site (better known as the 12-Mile Dumpsite, or 12-MDS). This section ends with a description of that site and the associated study.

MASSACHUSETTS BAY INDUSTRIAL WASTE SITE

The MBIWS, also called the "Foul Area" because material dumped there often tears fishermen's nets, was used for nearly 100 yr for disposal of industrial and commercial wastes. The principal site is circular in shape, with a radius of 1 nautical mile from the center at 42°27.7'N, 70°35.0'W. (Prior to 1975, the center was at 42°26.8'N, 70°35.0'W.) While records are incomplete or unavailable, it is clear that a wide variety of waste materials were disposed at the MBIWS. Among these materials were low-level radioactive wastes (LLRW) and toxic and hazardous chemicals.

According to Wiley *et al.* (1992), between 1953 and 1959 roughly 4,000 containers of LLRW were jettisoned in Massachusetts Bay at four designated sites, principally within or near the MBIWS. Some defense-related LLRW were deposited as early as 1946. Their resting places are unknown. Much of the material dumped in later years was encased in reinforced concrete. Unfortunately, it is suspected that not all material--particularly that disposed in earlier years--was protected in this fashion.

The toxic and hazardous wastes disposed at the MBIWS were generally held in 55-gal drums, many of which were punctured during disposal to insure sinking, dispersal, and dilution. While specific records of actions taken during disposal are virtually nonexistent, available cargo manifests list materials believed to have carcinogenic, mutagenic, neoplastic, or teratogenetic effects. Compounds containing heavy metals and halogenated organic contaminants are on the manifests. Overall, the full nature of toxic materials dumped at the MBIWS is unknown. Wiley *et al.* (1992) estimate, based on intensive sidescan sonar and remotely-operated vehicle surveys of the site, that approximately 21,000 containers are present in or near the site. In addition, it is likely, based on available records and eyewitness accounts of dumping activity, that many more containers were dumped in areas of Massachusetts Bay other than the MBIWS. Dumping at the MBIWS ceased around 1976; the site was de-designated by the EPA in the early 1980s.

The "Foul Area" is heavily fished by bottom trawlers and fixed-gear fishermen. In 1971, the U.S. Food and Drug Administration issued a "Notice to Harvesters" requesting fishermen to avoid the area as organisms harvested might be contaminated. Catches are dominated by American plaice and witch flounder, with Atlantic wolffish *(Anarhichas lupus)*, Acadian redfish, cusk *(Brosme brosme)*, haddock, and pollock caught in lesser amounts. The total number of fish species collected at the site was 41. The NMFS landings data for the 10'square encompassing the MBIWS (which also includes the productive Stellwagen Bank system) during 1982-84 showed annual landings averaging 6,316,000 kg.

The site is currently under study by the EPA in cooperation with NOAA and other resource agencies, to determine if materials dumped at the MBIWS, particularly LLRW, are affecting the Massachusetts Bay ecosystem, and to determine whether remediation should be undertaken.

ACID WASTE SITE

The AWS (40°16' to 40°20'N and 73°36' to 73°40'W) was established in 1948 for disposal of certain acid and alkaline wastes generated by metropolitan New York and New Jersey industries. The site received interim designation by the EPA in 1973, and was last used in 1988. The EPA determined that effects from dumping at the site would be localized and transient (Environmental Protection Agency 1980), and that long-term effects from past discharges at the site are unlikely. Therefore, monitoring has been terminated.

DEEPWATER DUMPSITE 106

The 106-Mile Deepwater Industrial Waste Site is within a larger 106-Mile Deepwater Sewage Sludge Site to allow effects of industrial waste and sludge dumping to be monitored cumulatively. The overall site, called Deepwater Dumpsite 106, is circular in shape with a radius of 3 nautical miles centered at 38°45'N, 72°20'W. Between 1961 and 1978, approximately 5.1 million metric tons (mt) of hazardous liquid chemical waste, in addition to approximately 380,000 mt of sludge and other municipal wastewater residues, were dumped at DWD-106. In 1991, it was closed by the EPA.

The Environmental Protection Agency (1989) indicates that commercially important species are not generally fished within DWD-106, but the overall area (*i.e.*, Middle Atlantic Bight) supports commercially important fisheries for yellowtail flounder, red hake, Atlantic mackerel, spiny dogfish (*Squalus acanthias*), tilefish (*Lopholatilus chamaeleonticeps*), Atlantic surfclam (*Spisula solidissima*), American lobster (*Homarus americanus*), and red deepsea crab (*Geryon quinquedens*). The study further mentions that landings of yellowfin tuna (*Thunnus albacares*), silver hake, and haddock come from the area.

Like the AWS, the EPA has determined that no lasting effects have been seen at the 106-Mile Deepwater Industrial Waste Site, and, therefore, continued monitoring is unnecessary. However, the larger 106-Mile Deepwater Sewage Sludge Site was extensively monitored pursuant to provisions of the Ocean Dumping Ban Act of 1988, under auspices of the 106-Mile Site Monitoring, Research, and Surveillance Program. Living marine resources were a principal focus of the monitoring effort.

INDUSTRIAL WASTE DISPOSAL EFFECTS ON FISHERIES RESOURCES

As mentioned, most industrial wastes were simply dumped into open ocean waters, as it was believed dilution and dispersion would minimize effects on living marine resources (Environmental Protection Agency 1980). Such a belief has guided the decision-making of federal agencies, but concern about containerized waste, such as at the MBIWS, has resulted in an initiation of new monitoring efforts to allow a better understanding of long-term effects of these disposals. Declining abundance of commercially important fish stocks, though, in areas remote from disposal sites suggests that overfishing or other population factors yet remain primary causes of such declines.

12-MILE DUMPSITE

South of Cape Cod, ocean dumping of sewage sludge occurred at a site 12 miles off Sandy Hook, New Jersey, in the northern New York Bight. From 1924 until 1987, when the site was closed, it was used by up to 200 sewage treatment plants, and though the number of municipalities using this site decreased over time, volumes of sludge increased as facilities improved (Environmental Processes Division 1988). Since 1960, when dumping rates were recorded, there had been a general increase, reaching a maximum annual amount of 7.6 million mt (wet weight) in 1983 (Suskowski and Santoro 1986). Sludge inputs in the early 1980s were, at the time, the largest known to any oceanic sludge disposal site (Norton and Champ 1989).

Composition of the sludge changed over time as improved sewage treatment and lowered industrial inputs reduced contaminant loadings and thus concentrations. A comparison of 1973 and 1987 loadings indicated decreases in total sludge solids, BOD, and most heavy metals; for example, amounts of cadmium, chromium, and mercury were reduced by at least 45% over the 14-yr period (HydroQual, Inc. 1989; Reid *et al.*, in press). Estimated annual pollutant loadings to the New York Bight apex from sewage sludge dumping and other major sources (*ca.* 1980) are given in Stanford and Young (1988).

Considerable effort has gone toward determining fates and effects of the dumped sewage sludge (*e.g.*, Gross 1976a,b; Mayer 1982), although an accurate determination of fates and effects has been confounded by other waste inputs to the area. Sludge disposal ranked only third in loadings of organic carbon and most contaminants to the inner bight; whereas dredged materials ranked first, and the Hudson-Raritan outflow second (Stanford and Young 1988).

A phase-out of sludge dumping in the inner bight between March 1986 and December 1987 prompted a multidisciplinary study by the NMFS and collaborators to examine ecosystem responses and thus infer what the fates and effects of the sludge had been (Environmental Processes Division 1988). The study included monthly sampling of a large suite of physical, chemical, and biological variables from July 1986 through September 1989. During the study, 991 otter trawl tows were made at 25 stations covering an area of 100 km^2. Abundance and composition were recorded for 75 species, representing 46 families of fish and "megainvertebrates." The New York Bight apex hosted a wide variety of migratory species, while the benthic/demersal biomass was dominated seasonally by a few. Atlantic rock crab (*Cancer irroratus*) was the most abundant such species in summer, replaced by little skate (*Raja erinacea*) and spiny dogfish during winter (Wilk *et al.*, in press).

Five species made up greater than 75% of the fish biomass: little skate, spiny dogfish, winter flounder, red hake, and ocean pout. Little skate and winter flounder occurred at greater than 80% of all stations. Four species made up greater than 90% of megainvertebrate biomass: Atlantic rock crab, horseshoe crab (*Limulus polyphemus*), American lobster, and longfin squid (*Loligo pealeii*). Atlantic rock crab occurred at greater than 85% of all stations (Wilk *et al.*, in press). In demersal fish catches at the three most intensively sampled stations, winter flounder and ocean pout made up 10 and 6.7%, respectively, of the weight per; winter flounder and silver hake made up 11.4 and 5.8%, respectively, of the numbers per tow. Atlantic cod and haddock were generally absent or merely traces in the catches.

The null hypothesis of no difference due to abatement of sludge disposal was tested for total fish and megainvertebrate biomass, as well as for frequently-occurring-species biomass (Pikanowski 1992). The null hypothesis could only be tested on species available throughout the year. Therefore, although catches included Atlantic cod, haddock, yellowtail flounder, windowpane, silver hake, and ocean pout, these species were seasonal, and only winter flounder was available all year. Total fish and megainvertebrate, Atlantic rock crab, little skate, and winter flounder biomasses showed no significant change ($P>0.05$) in relative abundance after cessation of dumping. American lobster did show a significant change ($P<0.05$) in relative abundance, increasing after dumping stopped. Sporadic occurrences of demersal fishes in the vicinity of the dumpsite suggest that effects of sewage sludge on these species are tenuous, particularly now that the site has been closed.

Diets have been examined for several demersal fishes (*e.g.*, yellowtail flounder, winter flounder, windowpane, silver hake, and red hake) collected in the sewage-sludge-affected area, although large samples were not always available (Steimle 1985; Steimle and Terranova 1991; Steimle, in press; Steimle *et al.*

1994). In general, diets reflected availability of common prey in the most sewage-affected areas of the bight apex, and basically were consistent with results of other diet studies for these species outside the sludge-affected area. The only indication of a significant shift in diet related to sewage sludge abatement was a decrease in frequency of *Capitella* spp. polychaetes used as prey by winter flounder. Corresponding to a decline in populations of *Capitella* following cessation of dumping, their frequency also decreased in winter flounder diets. Percentage of empty stomachs from winter flounder and red hake collected in the sludge-affected area during disposal was about 13%. This percentage did not decline within 2 yr after abatement, but instead increased. This increase appears to be related to a greater than 50% decrease in benthic macrofaunal biomass at the sludge-affected station after abatement (Reid *et al.*, in press).

Responses of benthic macrofauna to cessation of sludge dumping were somewhat greater than changes in fish diets, but were still limited (Reid *et al.*, in press). The area which had been thought to accumulate the greatest amounts of sludge had been dominated by *Capitella* spp. and nemerteans (ribbon worms), and had fewer species than found in other parts of the bight apex. After cessation, **total numbers** of species increased significantly relative to numbers at other sites sampled, as did total numbers of species of mollusks and crustaceans (including amphipods). As noted, there were significant decreases in abundances of *Capitella* spp. and ribbon worms. However, at 34 and 39 mo after cessation of dumping, there was little evidence of colonization of the sludge-affected area by species that were dominant at the other sites.

During a NOAA National Status and Trends Program study of demersal fishes--including winter flounder and spot (*Leiostomus xanthurus*)--in estuaries between Chesapeake Bay and Frenchman's Bay, Maine, Zdanowicz and Gadbois (1990) measured increased concentrations of synthetic organic contaminants in stomach contents relative to sediment, and increased concentrations in liver tissue relative to stomach contents. Similar bioaccumulation appears to have occurred at the 12-Mile Dumpsite. Sediments near the site are contaminated with organic compounds (PCBs, PAHs, and chlorinated pesticides) as well as heavy metals such as lead, copper, zinc, mercury, cadmium, and chromium (Boehm 1982; Zdanowicz 1982, 1991; Deshpande and Powell, in press). Concentrations of sewage-derived contaminants generally decrease with distance from the center of the site, except that depositional areas to the west--in the Hudson Shelf Valley--tend to be more contaminated than shallower, sandy sites located similar distances to the east. American lobster and winter flounder from the site had higher levels of PCBs than animals from adjacent reference areas (Draxler *et al.* 1991). Uptake of contaminants by prey and predators may result from exposure to contaminated water and sediment, or from transfer up the food chain, as suggested by findings of elevated heavy metal concentrations in benthic prey of demersal fishes in the dumpsite area (F.W. Steimle[54] and V.S. Zdanowicz[55], pers. comm.).

DREDGING AND DREDGED-MATERIAL DISPOSAL

Jonathan M. Kurland[56], F. Michael Ludwig[57], Stanley W. Gorski[58], and Chris Mantzaris[59]

The need for dredging and dredged-material disposal stems from expansion of waterborne commerce. Put simply, the draft of many commercial vessels (and even some recreational vessels) exceeds the natural depths of most harbors and rivers. Increased traffic by more efficient, deep-draft vessels is another factor.

Ocean disposal is commonly used because of the large number of navigable rivers and harbors, the vast amount of dredged material that is generated from their maintenance, and the inexpensive nature of at-sea disposal. Nationally, more than 65% of all dredged material is dumped in the ocean. Unfortunately, domestic and industrial wastes contaminate many harbor and river sediments, and dumping contaminated dredged materials in the ocean contributes to declining health and abundance of important aquatic habitats, as well as the living marine resources that rely on those habitats.

DREDGING

Dredging creates and maintains navigable waterways, turning basins, harbors, and marinas. Dredging projects in the coastal zone are diverse in purpose and in severity of effects. Potential adverse effects of dredging include: (1) increased turbidity; (2) altered sediment structure; (3) disruption and direct removal

[54] National Marine Fisheries Service, Highlands, NJ.
[55] *Id.*
[56] National Marine Fisheries Service, Gloucester, MA.
[57] National Marine Fisheries Service, Milford, CT.
[58] National Marine Fisheries Service, Highlands, NJ.
[59] National Marine Fisheries Service, Gloucester, MA.

(covering) of sensitive habitats (*e.g.*, eelgrass beds, bivalve mollusk beds, and spawning and nursery areas) and associated biological communities; (4) modification of natural water circulation patterns; (5) disruption of anadromous fish migrations; and (6) resuspension of trapped nutrients, organic matter, and contaminants (including toxicants) from within the substrate.

Short-term effects on marine organisms include clogging gills and digestive organs, reducing light penetration, facilitating eutrophication, depleting DO supplies, and making heavy metals, pesticides, and other pollutants bioavailable during dredging and disposal operations. Toxic contaminants can accumulate in tissues of marine organisms, contributing to long-term, chronic, debilitating effects. Also, animal and plant species can be displaced by excessive turbidity from dredging operations. Importantly, if dredging and disposal are curtailed or managed during sensitive life stages, adverse effects can often be minimized.

OCEAN DISPOSAL OF DREDGED MATERIALS

Disposal of dredged materials has environmental effects beyond those associated with actual dredging operations. The U.S. Army Corps of Engineers (COE) presently disposes of approximately 65% of its dredged material in open water; land disposal, however, may prevent adverse environmental effects that cannot be controlled in open-water disposal. Drawbacks to land disposal include difficulties in securing large tracts of land, handling problems, polluted water overflow and runoff, saltwater intrusion into groundwater, and costs of transporting materials to terrestrial disposal sites. These difficulties emphasize the relative advantages of open-water disposal. However, disposal of some types of polluted materials can pose a threat to estuarine and ocean ecosystems.

Concentrations of contaminants generally are associated with levels of organic enrichment of harbor silts and clays because of these contaminants' strong molecular affinity for fine particles, and because of the extent of domestic and industrial wastes released to urban or industrial harbors. Specifically, concerns with disposal of materials in open-estuarine or ocean waters can be summarized as follows:

1. Contaminants in industrial wastes, dredged materials, and sewage sludge can be assimilated into the marine food web via bioaccumulation and biomagnification. Accumulation of certain contaminants in tissues of marine organisms can affect physiological functions, resulting in compromised fish health, fecundity, and recruitment, and can represent a health hazard to human consumers.

2. Dredged materials dumped in the ocean can smother and eliminate populations of sessile or partially mobile benthic communities. Organisms living within these communities are forage for many species of predaceous fish and invertebrates. Loss of forage resources diminishes the value of an area for predaceous species, which must then seek food from other locales. This displacement consequently affects associated recreational and commercial fishing activities.

3. Organic sludges, associated with some dredged materials, elevate oxygen demand as they decompose. Ocean disposal of these materials may result in oxygen reduction (hypoxia) or even anaerobic conditions (anoxia) over portions of the ocean or estuarine bottoms and overlaying waters, particularly during periods when strong thermoclines are present. Such conditions can kill benthic organisms or even entire communities; they typically degrade habitats, leading to proliferation of stress-tolerant organisms of reduced value to the ecosystem. Dredged material disposal can also prompt closures of bivalve mollusk beds due to dispersion of pathogenic contaminants within sediments, water, and organisms.

4. Materials disposed of in the marine environment contain compounds that may promote growth of undesirable species of phytoplankton. In some cases, open-water disposal provides the very nutrients that are naturally limiting for planktonic growth. Any sudden availability of these nutrients can trigger plankton blooms.

Although voluminous historical data and other evidence exist equating specific contaminants with damage to the marine ecosystem and fisheries resources, long-term relationships between ocean dumping practices and direct cumulative effects on resources remain unresolved and controversial.

STATUTORY AND REGULATORY CONSIDERATIONS

In the early 1970s, concerns regarding the amount of sewage sludge requiring ocean disposal and the types of contaminants in such materials prompted a widespread examination of ocean dumping issues by regulatory agencies and other researchers. Scientists found that effects of ocean disposal were becoming recognizable at some of the deepest offshore dumpsites. These findings, combined with increased public awareness of ocean disposal problems, eventually led to passage of legislation prohibiting ocean disposal of sewage sludge, and mandating alternative processing and disposal methods.

The Marine Protection, Research and Sanctuaries Act of 1972 established criteria for evaluating dredged materials disposal. Suitability of dredged materials for ocean disposal is also governed, in part, by the London Dumping Convention, which identifies contaminants of concern and specifically prohibits disposal of selected toxic materials. Detailed sediment analyses to determine suitability for ocean disposal, however, are guided by the national "Green Book" and related operating procedures established by the EPA, COE, and other key agencies.

Under current ocean dumping regulations (40 CFR 220 *et seq.*), only dredged materials that have been evaluated and proven suitable for ocean disposal may be disposed of at sea. Criteria for this evaluation are intended to prevent further degradation of the marine environment. National and regional dredged material testing protocols have been developed by the EPA and COE in cooperation with the U.S. Fish & Wildlife Service, coastal states, and the NMFS. Materials that do not pass the analyses governed by these protocols are deemed unsuitable for unrestricted ocean disposal.

Confined ocean disposal, known as "capping," is one management alternative available for disposal of some types of materials that fail an initial assessment process. While capping is not a strategy that can routinely be applied to all ocean disposal activities, capping of certain toxins (*e.g.*, at the Mud Dumpsite in the New York Bight, and at the Central Long Island Sound Dumpsite) may be an acceptable option due to water depth, oceanographic conditions, sediment types, and availability of suitable capping material. These attributes and other factors are continuously monitored to insure effectiveness of capping.

The most recent guidelines for performing sediment tests on dredged materials are described in the revised draft, "Evaluation of Dredged Material Proposed for Ocean Disposal: Testing Manual" (Environmental Protection Agency and U.S. Army Corps of Engineers 1991), and in the regional "Guidance for Performing Tests on Dredged Materials to be Disposed of in Open Waters" (Environmental Protection Agency, Region I, and U.S. Army Corps of Engineers, New England Division 1989). Application and interpretation of test results performed under these guidance documents are discussed in more detail below.

Evaluation procedures for determining suitability of dredged material for ocean dumping emphasize the potential for biological effects rather than simple presence or absence of possible toxicants. Additionally, the four-tiered evaluation process creates a hierarchical sequence of assessment for determining suitability (to dump) or noncompliance with the ocean disposal regulations. Tiers I and II rely on existing information and relatively simple testing procedures for determining potential environmental effects of dredged materials under review. Those materials with nonexistent or minimum levels of toxicants, and with little likelihood of environmental effects, are candidates for ocean disposal. Materials failing initial review are advanced to subsequent tiers for further analysis of toxicant availability and potential environmental effects.

In tier III testing, sediment toxicity in the water column, bioassay, and bioaccumulation are assessed. Benthic community effects are evaluated by comparing bioaccumulation results with reference sediment uptake levels. If results show that open-water disposal is unacceptable, the materials must be evaluated under tier IV investigations should the project proponent wish to continue to seek ocean disposal. Tier IV considers steady-state bioaccumulation levels, biological evaluation of dredged materials, and evaluation of special management practices that might be employed to mitigate effects associated with placing materials in the open ocean.

Determination of suitability for ocean disposal is based, in part, on outcome of bioassay and bioaccumulation tests. Bioassay tests are designed to indicate the presence and toxicity of materials through limited but controlled exposure of specific indicator organisms to sediments of concern. The regulations consider as potentially undesirable: (1) statistically significant mortality, and (2) elevations of body burdens of contaminants above those found in reference animals.

The EPA has primary responsibility for the designation and management of open-water disposal sites. Those responsibilities integrate the tasks of permitting, enforcement, monitoring, and data interpretation. To comply with federal mandates, the EPA evaluates effects of ocean disposal by comparing monitoring data with predisposal baseline conditions, taking into account both short-term and long-term potential environmental effects. Effects occurring before, during, and following disposal activity, as well as long-term or cumulative environmental changes, are evaluated. The EPA conducts monitoring surveys at approved and potential ocean disposal sites to determine dredged materials distribution and movement (including resuspension and transport), benthic colonization of dredged material, sediment chemistry, food chain interactions between benthos and fish, and bioaccumulation of contaminants in organisms.

In addition to procedures noted above, Section 7(a)2 of the ESA requires agencies to ensure that proposed actions will not jeopardize the continued existence of listed species. The NMFS, which has jurisdiction over threatened and endangered marine species, has completed final recovery plans for humpback and northern right whales, and is developing a recovery plan for listed sea turtles as well. These plans describe actions deemed necessary to achieve recovery, potentially including dredged material management considerations. The plans establish implementation schedules that identify the federal agencies best suited to address each recovery action. The NMFS will continue to work with the EPA, COE, and other key participants to coordinate efforts toward achieving the goals of each plan.

Finally, there is the broad and recurring issue of determining the appropriate levels of contaminants that can be safely placed in the marine environment. Evidence is accumulating that amounts of persistent pollutants such as PCBs and cadmium are increasing in both sediments and biota of the continental shelf. Although presently it is impossible to equate any contaminant increases to any specific human action, it is impossible to exclude dredged material disposal in open water as a source of these contaminants. The charge of federal resource agencies is to protect and enhance the health and availability of our nation's public trust resources. To achieve this goal the above criteria for proper disposal of dredged material must be: (1) scrutinized closely and continuously, (2) interpreted so as to be ecologically conservative, and (3) updated as new information becomes available.

Table 6 lists and provides synoptic data on the five major and six minor (*i.e.*, infrequently used) sites for disposal of dredged materials in New England.

Table 6. Major and minor (infrequently used) dredged-material disposal sites in New England

Site	Description
	Major Sites
Rockland, Maine	Located approximately 3.3 nautical miles northeast of the Rockland Harbor breakwater. Receives approximately 193,000 yd^3/yr of dredged material. Has been used since 1973.
Portland, Maine	Located 3.5 nautical miles from Portland Harbor. Receives approximately 158,000 yd^3/yr of dredged material. Has been used since 1973.
Cape Arundal, Maine	Located approximately 2.75 nautical miles southeast of Cape Arundal. Receives approximately 123,000 yd^3/yr of dredged material. Has been used since 1985.
Massachusetts Bay, Massachusetts	Used to be known as "Foul Area." Recently designated by EPA as permanent disposal site for relatively "clean" material. Receives approximately 293,000 yd^3/yr of dredged material.
Buzzards Bay, Massachusetts	Located 1.4 nautical miles from Chappaquoit Point, West Falmouth. Used infrequently as disposal site for dredged materials. Since 1979, 92,000 yd^3/yr of material have been disposed of at site.
	Minor Sites
Wellfleet, Massachusetts	Located adjacent to Wellfleet Harbor entrance. Site was last used in 1983.
St. Helena, Maine	Located outside St. Helena Harbor entrance. Site was last used in 1988 when only 385 yd^3 of dredged material were dumped. In 1984, approximately 61,000 yd^3 of dredged material were disposed of at site.
Frenchman's Bay, Maine	Used infrequently.
Saco Bay, Maine	Used once, in 1989, for disposal of approximately 51,000 yd^3 of dredged material.
Sandy Bay, Maine	Used once, in 1987, for disposal of approximately 6,000 yd^3 of dredged material.
Sheep Island, Maine	Used twice, in 1987 and 1988, for disposal of 2,000 and 103,000 yd^3, respectively, of dredged material.

MINING OF SEABED AGGREGATES

John B. Pearce[60]

Effects of coastal and deep ocean mining for aggregates can best be elucidated by reports from the ICES Working Group on Effects of Extraction of Marine Sediments on Fisheries (International Council for the Exploration of the Sea 1991b, 1992a, 1992c, 1995) and the Minerals Management Service (Hammer *et al.* 1993). These reports, developed over the past two decades, include reference to North American case studies and results. Other sources include reviews and environmental impact statements pertaining to sand and gravel mining, dredging, and trawling as such activities relate to industry within urban areas including Boston, New York, and Baltimore Harbors, and within Canadian waters (New England Marine Resources Information Program 1973; Schlee and Sanko 1975; Messieh *et al.* 1991).

NOAA, the COE, and the U.S. Geological Survey (USGS) have extensively researched these issues. For instance, the USGS has studied colloidal material and its possible roles in transport of sediments and contaminants (Rees 1991). The aforementioned ICES reports remain the best information on fisheries effects of ocean mining. A NOAA paper by Orlando *et al.* (1988) stresses the cumulative effects of shoreline modification, including dredging and mining.

Locally, Duane *et al.* (1988) identified two areas in Massachusetts Bay and ancillary waters where sand and mixed aggregates occur in amounts sufficient for mining. One of these areas is off Boston Harbor, between Hull and Plymouth; the other is Stellwagen Bank, although it has now been designated a national marine sanctuary, where, under the terms of designation, mining is prohibited. Aggregate beds occur in other areas in the Gulf of Maine (*e.g.*, off Cape Ann), but in many instances the beds are limited in size or occur in waters too deep to be mined easily. Various reviews (*e.g.*, Cruickshank and Hess 1975) have speculated on potential sources of industrial amounts of sand and gravels.

Mining is usually done by surficial scrapping or point excavation of materials to some greater depth. Hydraulic dredges generally are used to lift materials to receiving barges. Environmental effects of such removal have been listed as: (1) "destruction" of existing benthic biota; (2) resuspension of fine sediments with subsequent effects on larval, juvenile, and/or adult fish or juveniles and larvae; (3) changes in profiles or surfaces of mined areas; and (4) consequences of entrained fine materials being carried tens or scores of kilometers from the dredging site. Development of deep excavation or "borrow" pits can become anaerobic during certain periods of the year, with attendant direct effects on fish (Pacheco 1983). In many areas of the Gulf of Maine and in the New York Bight, there is concern about how mining in one area may affect fish eggs, demersal larvae, juveniles, and adults some distance away. Mechanisms discussed by Rees (1991) are of interest in these regards.

Ancillary information from studies on environmental effects of trawling may be pertinent to assessments of mining. Jones (1992) noted that effects can be related to weight of towed gear, towing speed, characteristics of sediments, and strengths of tides and currents in the trawling area. Valentine and Lough (1992), in a study of Georges Bank, report that in untrawled or lightly trawled areas there exists "a biologically diverse community dominated by abundant attached organisms." In heavily trawled areas, "this community is poorly represented, and the bottom is relatively smoother." Finally, Messieh *et al.* (1991) discussed effects of trawling, dredging, and dumping, and reported on inevitable effects and changes and on the need for further research and assessments.

There is little likelihood that, regionally, individual adult fish would be affected drastically by specific mining operations, since active adults of most species are usually able to avoid such local activities. Beyond this, however, observations in the English Channel, off eastern England, and along Holland and other low countries, suggest that there may be effects ranging from minor local ones to more significant widespread ones. DeGroot (1979) noted that dredging and construction of a sand island would cause long-lasting damage and permanent economic loss to fisheries in a particular area; in 1990, $5.3 million (U.S.) would have been lost due to this activity. Also, DeGroot and others have expressed concern about specific effects on Baltic herring, and have noted that catches in trap nets set near certain dredging activities off Finland were affected by a mining operation. However, there are no data that suggest declines in the trapnet catches could be related directly to mining activities.

Data on possible effects on marine mammals are almost nonexistent. In a review, Hammer *et al.* (1993) suggested that while suspended sediments might occur due to mining, elevated levels should have little effect on whales since they can avoid plumes of entrained sediments. Such judgements were speculative, however, since occurrence of whales is highly seasonal and not predictable. There was, and continues to be, concern about secondary consequences on marine mammals since direct effects of mining on benthic or demersal forage species have been measured, which in turn may affect both the fisheries and mammals. Finally, this review stressed possible sonic effects, but noted that generally mammals have not been measured for effects by various noises.

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[60] National Marine Fisheries Service, Woods Hole, MA.

The most recent comprehensive reviews of effects of extraction of marine sediments on fisheries are in a draft report prepared by the ICES Working Group on the Effects of Extraction of Marine Sediments on Fisheries (International Council for the Exploration of the Sea 1992a). This report contains considerable background and an extensive bibliography on ocean mining, as well as references to possible effects. Section 3 of the draft report, "Effects of Activities on Living Resources and Fisheries," contains detailed information on physical effects on the seabed and water column, as well as on specific biological effects. In discussing removal of substrata from coastal and shelf environments, the authors noted that where "pits" are developed, there is relatively little possibility for transport of gravel from adjacent areas into such depressions. Consequently, infill occurs slowly, and, with reduced strength of bottom currents within the pits, there often is deposition of finer organic sediments resulting in organic enrichment and concomitant oxygen depletion (see Pacheco 1983; Messieh *et al.* 1991). Recent studies using severely contaminated sediments show that "typical" marine invertebrates may respond behaviorally and physiologically in ways that are detrimental to individuals as well as to organisms higher in the food web (Olla *et al.* 1988).

A principal concern of the ICES (1992a) report is dredging in spawning areas of commercial fish species. Secondarily, there is concern about removal of important benthic forage species used by a wide range of fish species and various life history stages. The ICES report also touches on how primary productivity might be affected by increased turbidity resulting from dredging in inshore areas such as Boston Harbor or Raritan Bay. Finally, in considering physical/chemical effects, it was noted that if screening of sediments is done at sea, materials returned to the seabed may change adversely the nature of the surficial sediments, and, hence, the ability of marine organisms to recruit in sediments when specific sediments types are required for this process. From a chemical point of view, the report noted that the bulk of mined aggregates are "clean" sands and gravels, and that because of their relatively large particle size, low surface area relative to total bulk, and low surface activity (*i.e.*, few clays or organic materials to interact chemically), there is usually relatively little chemical interaction with the water column. The report also pointed out that when sediments are disturbed in estuaries or deep channels where fine materials have accumulated, there is a possibility for release of harmful chemicals from sediments to overlying waters.

There are well documented instances where dredging has changed biological communities. The scale or magnitude of dredging operations is a prime factor in the severity of effects upon benthic and associated demersal fish communities. Similar to "block cutting" of forests, dredge cuts or folds have been created in the seabed which are separated by undisturbed "hummocks" of sediment containing adult populations of benthic species. Such species are capable of repopulating surrounding dredged furrows or pits. McCauley *et al.* (1977) documented such occurrences.

In addition to size of area and time of year of dredging (*i.e.*, it should be avoided during spawning and early life history), another important factor is type of dredging. Effects of mechanical dredging upon the seabed and biota are usually severe but localized, whereas effects of hydraulic dredging are less profound but much more widespread. Numerous agencies and organizations are now sponsoring research on such effects. For instance, Burton *et al.* (1992) report on entrainment effects of hydraulic dredging in estuaries on striped bass eggs and larvae, and Banks and Alexander (1994) discuss modified dredge dragheads to avoid entrainment of and damage to sea turtles and other fauna.

DeGroot (1979) reported on effects of inshore dredging on benthic communities in a small area of Seine Bay, France, where sand and gravel that overlaid a rocky substratum were removed. No redeposition of sediment upon the exposed rocky ground occurred, and, as a result, a hard-bottom fauna developed over time which had less food value for fish species than did the preexisting soft-bottom fauna.

Studies of recovery of benthic communities in highly stressed areas have noted that infaunal communities exposed to strong tidal currents or periodic storm disturbances are often dominated by short-lived (opportunistic) species, at least in surficial sediments (McCauley *et al.* 1977). The ICES summary of benthic sensitivity to sediment removal and sediment redeposition noted that most benthic organisms are destroyed in the immediate area of dredging activity (International Council for the Exploration of the Sea 1992a). At the periphery, "edges" (ecotones) will often occur where benthic species are particularly abundant. As noted previously, dredge pits may become anaerobic at certain times of the year, and thus represent traps for various larval and juvenile fishes and benthic invertebrate forage species. The draft ICES Report (International Council for the Exploration of the Sea 1992a) recognized that certain species were especially sensitive. These include sand lance and herring that lay demersal eggs which adhere to stones and gravel. In the case of herring, when aggregates are removed, eggs are taken with them; in the case of sand lance, an important prey species for many gadoids, eggs may become surrounded by sand and finer particles, and killed. Finally, various species of crab and lobster use certain aggregates as habitat, or as migratory and feeding pathways. Often, crabs burrow in sand, and, obviously, when dredging occurs, individuals are taken up and physically destroyed or injured so they eventually die. Thus, again, decisions on dredging should consider those species that might be present within a particular habitat at specific times of the year.

As a basis for future management of dredging in areas that represent spawning grounds or habitat for important marine resources, ICES and its aforementioned working group highly recommended that such areas be mapped so that resources can be avoided or managed during removal of sands and gravels (International Council for the Exploration of the Sea 1992a). They suggested that marine mining activities should be monitored on a regular if not continuous basis to provide records for regulatory authorities and the scientific community. If dredging for aggregates were so monitored, it would be possible to understand better what the consequences of such activities might be within specific areas.

The draft ICES report (International Council for the Exploration of the Sea 1992a) concluded that in most member nations, the at-sea mining industries are "well established and growing." Given this, it is obvious that greater attention must be given to compiling existing data and information and organizing long-term monitoring, research, and assessments. The physical effects must be understood, including: (1) change in topography and sediment types, (2) turbidity and sedimentation, and (3) burial (of fauna and sediments) due to at-sea processing. Further, it was suggested that while most aggregate mining involved "clean" sediments, even these can have certain chemical effects, **albeit** minor ones. As noted, extent and timing of aggregate extraction are important in regard to biological effects as are: (1) types of dredges, (2) overside disposal of "wastes," (3) depth of water and sediments, (4) types of sediments, and (5) nature of indigenous species and benthic community structure.

The draft ICES report (International Council for the Exploration of the Sea 1992a) noted that of 12 countries surveyed, 11 have specific legislation governing submarine extraction of minerals. Seven have specific terms and conditions relative to environmental and fisheries management. The report also noted that monitoring must consider "sensitive" as well as endangered species, and should be conducted with specific minimum requirements for electronic data gathering, including: (1) position(s) of mining vessels, (2) time of deployment of gear, and (3) unusual events. Monitoring could include underwater video and photography, sediment sampling, sidescan sonar, and assessments of benthos and suspended sediments and plankton. The latter would include data on: (1) change(s) in benthos, including invertebrate and demersal fish species; (2) recolonization rates; and (3) long-distance transport of sediments via plumes.

Finally, in regard to ICES concerns for mapping, the NMFS and USGS are conducting a joint program to map large areas such as Georges Bank (Valentine and Lough 1992). This mapping includes geologic and oceanographic environmental factors as well as distribution and abundance of fishes and invertebrates.

Studies and assessments of mining will be important for managing both habitats and resources in a sound manner, emphasizing measurements and documentation as opposed to assumed effects. This will become increasingly important as land-based sources of aggregates and minerals become evermore limited, and as we turn to the seas as sources. Moreover, states and localities are already looking to uses of dredged materials and mined aggregates as ways to: (1) replenish or restore beaches; (2) construct recreational and wildlife islands; (3) restore wetlands; (4) build oyster bars, fish reefs, and underwater berms; and (5) modify benthic habitats (Virginia Institute of Marine Science 1993). In addition to these beneficial uses of marine sediments, researchers are now considering how seabed areas heavily burdened with organic wastes can be manipulated to avoid buildup of bacteria, reducing conditions, and contaminants inimical to wild and cultured fishes (O'Connor *et al.* 1993). For decades, marine scientists have noted that such organic loadings from sewage discharges (Pearce *et al.* 1991), pulp mill discharges (Pearson and Rosenberg 1978), and mariculture facilities (Gowen *et al.* 1988; Mattison and Linden 1983) can affect benthos and demersal fisheries greatly. O'Connor *et al.* (1993) demonstrated that farrowing, harrowing, or raking contaminated sediments can aerate these sediments, eliminating reducing conditions and pathogens. Beyond these treatments, such enriched sediments could be mined for seabed applications or for terrestrial fertilizers.

Thus, marine and coastal zone managers must remain aware of future multiple uses of the seas, seabeds, and their biota. In recent decades, the ICES has annually asked its member nations to submit reports on amounts of marine aggregates extracted from coastal and estuarine waters; these data and the documented reductions in terrestrial sand and gravel beds indicate that nations must be better prepared to manage ocean mining at a time when there are increased demands for both aggregates (Pearce 1979) and sustainable fisheries.

MULTIPLE-USE ISSUES IN ESTUARINE AND COASTAL HABITAT LOSS

Stuart J. Wilk[61] *and Bradley W. Barr*[62]

As stated by Chambers (1991), habitat degradation and loss are adversely affecting coastal, estuarine, and riverine ecosystems which are essential for spawning, feeding, growth, and/or migratory routes for a majority of the nation's living marine resources. A review of multiple-use issues in such habitat degradation and loss is especially warranted in light of the depressed state of the Northeast's demersal fish stocks, and of the ever-increasing estuarine and coastal development in New England. The following discussion identifies and describes multiple-use issues identified by NOAA (National Oceanic and Atmospheric Administration 1986).

WASTE DISPOSAL

Disposal of dredged materials has been and continues to be practiced in estuarine and coastal waters off the Northeast. The once-common practice of disposal of sewage sludge, chemical wastes, and radioactive materials at ocean dumpsites, though, is now prohibited. Effects of these past and present disposal activities on Northeast demersal fish stocks are described in a number of monitoring study reports already mentioned. Although few of these stocks are estuarine dependent (notable exceptions being pollock and some flounders), heavy metals, chlorinated hydrocarbons, and petroleum products emanating from disposal activities nonetheless degrade some New England estuarine and coastal waters (National Oceanic and Atmospheric Administration 1986).

Effluent discharge from sewage treatment plants--either directly into receiving waters or into rivers emptying into those waters--contributes significantly to organic loading of estuarine and coastal habitats. Such loading can produce: (1) excessive algal blooms, (2) shifts in abundance of algal species, (3) BOD increases in sediments of heavily affected sites, and (4) anoxic events. Pathogens and parasites harmful to humans can be found in biota and sediments at organically loaded sites, including ocean dumpsites for sewage sludge. Presence of excessive concentrations of pathogenic bacteria and of indicator species of such bacteria has triggered closures of bivalve mollusk harvesting grounds (Environmental Processes Division 1988).

Dumping of wastes from fish and invertebrate processing operations generates much the same concern as sewage treatment effluent discharge and sewage sludge dumping. This concern includes increased BOD, algal blooms, and increased concentrations of pathogenic bacteria. Closure of land-based processing plants, because of their inability to meet federal (*i.e.*, National Pollution Discharge Elimination System, or NPDES) or state pollution discharge effluent requirements, serves to enhance the appeal of at-sea disposal. While at-sea disposal of these wastes is exempt from regulation under the Ocean Dumping Act, the onus of proof of no environmental harm rests with the entity pursuing at-sea disposal.

COASTAL URBANIZATION

Tremendous development pressures exist throughout the coastal Northeast. The NMFS reviews annually more than 2,000 permit applications for commercial, industrial, and private marine construction projects. These projects range from those which would have few effects to those (*e.g.*, major dredging and filling projects) which would eliminate significant habitats. Even projects with few demonstrable effects can have significant cumulative effects.

Construction in and adjacent to waterways often involves dredging and/or filling activities which elevate suspended sediment levels. Excessive turbidities can abrade epithelial tissue in marine organisms, clog gills, decrease egg buoyancy, and reduce light penetration, thereby affecting photosynthesis and causing localized oxygen depressions. Suspended sediments may subsequently settle to the benthos and destroy or degrade productive bivalve mollusk beds, forage areas, and spawning sites. Often, but not always, effects of turbidity and siltation are temporary and short-term (National Oceanic and Atmospheric Administration 1986).

Accompanying the increased development of estuarine and coastal areas is the demand for potable, industrial, receiving, and cooling waters necessary for ever-increasing wastewater treatment and disposal, community development, industrialization, and electric power. Demands increase as groundwater becomes depleted or contaminated, and as freshwater is diverted via dam and reservoir construction, canals, or other methods. Reducing flows to estuaries can reduce nutrient levels and increase salinity, and thus decrease overall productivity of estuarine systems. Moreover, reduction of nutrient-rich oxygenated water in a large estuary can lower significantly the biological productivity of large areas of coastal water normally exposed to the seaward-flowing estuarine plume.

Water not lost through domestic and industrial consumption is returned to rivers as point-source wastewater discharges. Although generally treated, domestic discharges often contain suspended organic and inorganic compounds (including chlorine compounds), heavy metals, nutrients, and bacteria. Sewage treatment effluent may produce nearfield changes in biological communities due to chlorination and increased contaminant loading. In addition to creating thermal plumes, industrial

[61] National Marine Fisheries Service, Highlands, NJ.
[62] Stellwagen Bank National Marine Sanctuary, Plymouth, MA.

discharges may contain dissolved and suspended contaminants, including nutrients, heavy metals, halogenated hydrocarbons, petroleum products, and other organic substances. The EPA regulates industrial wastewater effluent through NPDES permits as a means of identifying, defining, and, where necessary, controlling **all** point-source discharges. The problem remains, however, that it is difficult, perhaps impossible, to estimate the singular, additive, and synergistic effects of industrial and domestic wastewater discharges on estuarine and coastal habitats in general, and on fisheries resources specifically. Effects are cumulative and are the hallmark of intensively or extensively developed urban communities.

Associated with urban development are inevitable increases in nonpoint-source contamination of estuarine and coastal waters. Highways, parking lots, and removal of terrestrial vegetation and fringe marshes facilitate runoff of soil, fertilizers, biocides, heavy metals, grease, oils, PCBs, and other harmful materials to marine biota and their habitats. Atmospheric emissions from industrial processes may contain sulphurous and nitrogenous compounds that contribute to acid precipitation, a concern in some freshwater sections of tidal streams. Moreover, components of these nonpoint pollutants accumulate in water, sediments, and marine biota (see National Oceanic and Atmospheric Administration 1986).

ENERGY PRODUCTION AND TRANSPORT

Energy production facilities are widespread along Northeast coastal areas, and include land-based nuclear power plants, hydroelectric plants, and fossil fuel stations. Effects of these facilities on estuarine and coastal habitats include water consumption, heated-water discharges, thermal shock, entrainment (in cooling systems) and impingement (on intake screens) of organisms (especially of larvae and juveniles), discharges of heavy metals and biocides, destruction and elimination of habitat, and disposal of dredged materials and fly ash (National Oceanic and Atmospheric Administration 1986).

Although Congressional action has precluded outer continental shelf energy exploration in the most productive fishing grounds in the Northeast, future drilling, transport, and production facilities could affect biota and their habitats through deposition of drilling muds, cuttings, and other materials. Oil spills resulting from well blowouts, pipeline breaks, and tanker accidents remain a major concern. Seismic testing operations can interfere with fishing operations and damage fishing gear.

PORT DEVELOPMENT AND UTILIZATION

All ports require shoreside infrastructure, mooring facilities, and channels sufficiently deep for operations. Upgrading of facilities may include dredging and disposal of dredged materials, filling aquatic habitats to create land for port improvement, and expanded use and degradation of water quality. All of these activities can adversely affect marine biota and their habitats.

AGRICULTURAL OPERATIONS

Agricultural operations can affect fish habitats directly through physical alterations, and indirectly through chemical contamination and erosion and transport of suspended matter. Fertilizers, herbicides, insecticides, and other chemicals are carried into the aquatic environment via nonpoint-source runoff from agricultural lands. Such runoff can affect aquatic vegetation directly which will, in turn, affect the food web or chain. Agricultural runoff also transports sediments and animal wastes which can affect spawning and nursery areas, and degrade overall water quality and benthic substrates.

One of the most serious consequences of erosional runoff is that it necessitates frequent dredging of navigational channels. The resulting dredged material requires disposal, often in areas important to marine biota (National Oceanic and Atmospheric Administration 1986).

MARINE AGGREGATE AND MINERAL EXTRACTION

Mining for sand, gravel, shell stock, and other aggregates in estuarine and coastal waters can directly and indirectly affect benthic organisms, modify substrates, change circulation patterns, and decrease DO concentrations at deeply excavated sites. Sand and gravel mining resuspends various materials at mining sites, with the possibility of turbidity plumes moving many kilometers from those sites.

In some mining endeavors for potash and other minerals, affected habitats cover hundreds of square kilometers.

COASTAL AND WETLAND USE AND MODIFICATION

Increased demand for land suitable for homesites, resorts, marinas, beach clubs, and industry has destroyed or altered large areas of New England's estuarine and coastal wetlands and subtidal habitats through dredging, filling, diking, bulkheading, ditching, erosion, and other forms of shoreline modification. As residential and commercial uses of estuarine and coastal lands increase, so does the recreational use of adjacent waters. Marinas, public access landings, private piers, boat ramps, and beaches all vie with fish and other wildlife for space, and encroach upon essential, sensitive estuarine and coastal habitats.

Competing uses further contribute to destruction or modification of wetlands. Agricultural development, including wetland drainage to increase tillable acreage, can significantly affect wetlands. Flood control measures in low-lying coastal areas, including dikes, ditches, and stream channelization can also significantly affect wetlands. Wildlife management techniques that modify wetland habitats, such as construction of dredged ponds and low-level impoundments, can harm marine fishes since such freshwater habitats do not replicate the brackish or saltwater habitats they replace.

Each coastal state, as well as the COE, regulates projects proposed for wetlands. Although these regulations have to some extent ameliorated wetland modification and destruction, construction that is judged to be in the public interest or to be water-dependent continues, as does illegal, unauthorized construction. Primary threats associated with such construction activities (*e.g.*, agricultural runoff) have been discussed earlier in this section.

SUMMARY

John B. Pearce[63] *and Jon A. Gibson*[64]

The preceding 10 sections of this document summarize the state of our knowledge of the principal living resources, habitat conditions, and human perturbations in the Gulf of Maine. That knowledge is considerable, as evidenced, in part, by the following section of references cited. Gaps do exist, though, in our knowledge.

Our knowledge of the gulf's biota is generally good. We largely know the species composition, distribution, and abundance of the phytoplankton, zooplankton, benthos, fishes, marine mammals, and other biotic components of the ecosystem. We know little, however, about the trophic linkages among these components. While we have good data on food habits of commercially important fish species, we have only a fair understanding of the effects of competition for food within the ecosystem as such competition ultimately influences the composition, distribution, and abundance of fishes and marine mammals.

Our knowledge of the gulf's habitats is also generally good. We broadly know the topography, sediments, water masses, currents, temperatures, salinities, nutrients, and other abiotic components of the ecosystem. We know little, however, about the local interactions among these components. In particular, we have only a fair understanding of the localized effects of rivers, basins, seamounts, and ledges on major water masses and their hydrographic regimes.

Our knowledge of human perturbations in the gulf is generally fair. We largely know the fishing and whale watching activities, the point-source discharge and dumping of wastes, and the myriad of other industrial and commercial activities actually or potentially occurring in the ecosystem (transportation, mining, energy, etc.). We know little, however, about non-point-source discharge of wastes (especially by agricultural operations and from river outflows), and (except for a few well-studied sites and species) about locations, fates, and effects of pollution.

The biggest gap in our knowledge is, however, an understanding of the collective, cumulative, and synergistic (CC&S) effects of food chains, environmental variability, contaminant effects, *etc.*, all operating at the same time. Some 30 major workshops and conferences on the Gulf of Maine have been held in the last two decades; none have reached consensus on the effects of what we might call this "CC&S syndrome." Not suprisingly, most of the questions to yet be answered--from a fishery management standpoint--for the Gulf of Maine involve the intersection of biological processes, environmental conditions, and human activities. Examples of these questions are: (1) What are the effects of otter trawling and shellfish dredging on benthic habitats, which produce much of the prey for commercially important demersal fishes?; (2) What would be the effects of long-term climate change on the ability of fishery managers to restore overfished stocks to "normal" population levels?; and (3) What is the relationship between nutrient pollution, hydrographic conditions, and the occurrence and severity of toxic algal blooms? There are many such unanswered questions.

The preceding sections and the following references cited section provide a good summary of the principal living resources, habitat conditions, and human perturbations in the Gulf of Maine. More importantly, they provide a solid foundation for further study and research planning. Future research must increasingly address the CC&S aspects of the ecosystem's components and processes. Until such questions have been answered, fishery managers will continue to operate reactively, responding to what has occurred and what has been done. As answers to these questions accumulate, managers will begin to operate predictively, preparing for what will likely occur and providing options for what can be done. That is the future of fishery management in the Gulf of Maine and elsewhere. How soon the future arrives depends on society's support of such research and on our diligence in our research efforts.

[63] National Marine Fisheries Service, Woods Hole, MA.
[64] *id.*

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Jon A. Gibson, Technical Editor
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166 Water Street
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