

Chapter 5. Nekton (Dave Borkman, Jefferson Turner and Rodney Rountree)

Introduction

Except for the fishes, the nektonic community of Mt. Hope Bay has not been well described. Relatively little is known of the distribution and temporal abundance patterns of important decapod crustaceans such as the shrimps *Crangon septemspinosa*, *Paleomonetes intermedius*, *P. pugio*, and *P. vulgaris*, the crabs *Callinectes sapidus*, *Ovalipes ocellatus*, *Libinia* spp., *Cancer* spp., and the hermit crabs *Pagurus* spp., despite their great importance to the ecosystem. The lack of information on *Crangon septemspinosa* is particularly important as there is credible evidence that it is an important predator of winter flounder and other larval fishes (see discussion under the winter flounder section below). More information is available on the fishes, but attempts to investigate temporal and spatial abundance patterns has been limited to a few economically important species susceptible to trawl sampling.

There are approximately 30 years (1971 to present) of monthly to twice-monthly trawl data from various locations in Mt. Hope Bay (source: MRI, October 2001 presentation by M. Scherer) that provide information on the Bay's fish abundance and community composition. A standardized trawl has been made at least once per month at 5 locations (Spar Island, Cole River, Lee River, near BPPS intake, and a 'crossleg' transect south of BPPS) from late 1971 to the present (Figure 5.1). An additional monthly trawl, a Wilcox trawl with a smaller mesh (125 mm), has been implemented since 1993 to census the abundance of smaller fish, especially small winter flounder.

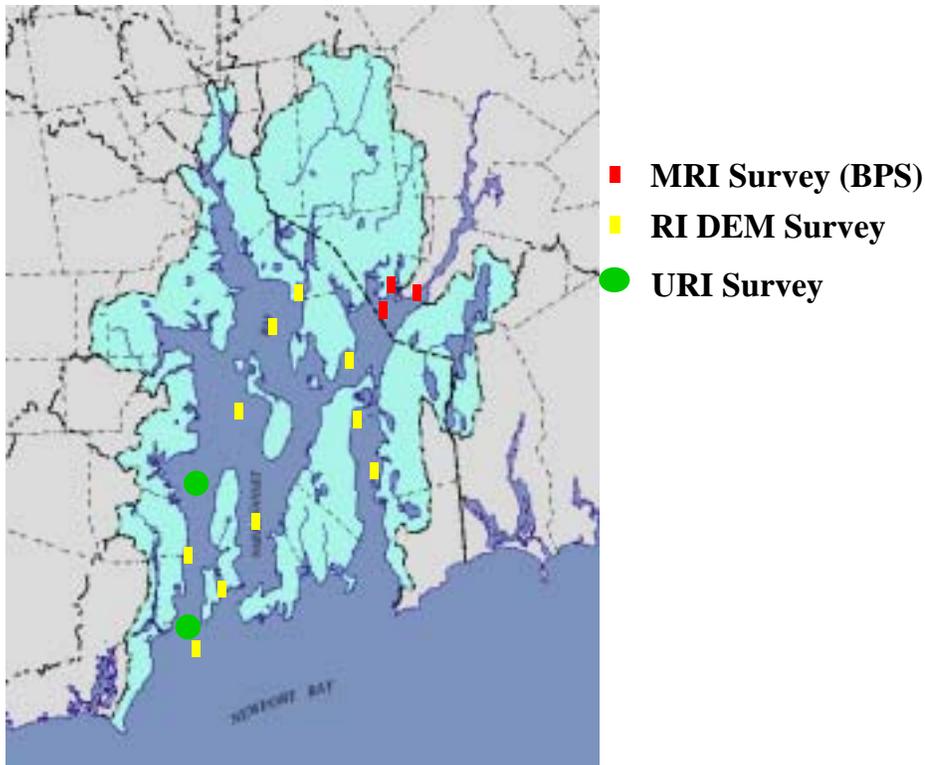


Figure 5.1. Locations of MRI, RIDEM and GSO/URI standard trawl sampling.

Additional sources of Mt. Hope Bay area fish abundance data include the weekly fish trawls taken near the mouth of the West Passage and Rhode Island Sound by GSO personnel since 1959 through the present (Figure 5.1, Jeffries and Johnson 1974, Jeffries and Terceiro 1985, Jeffries et al. 1989). The RIDEM has done a census of juvenile fishes in Narragansett Bay, including one station in Mt. Hope Bay, in the summer/early autumn of 1988 through 1996 (Figure 5.1, Meng and Powell 1999). The Narragansett Bay fish community was surveyed for a single year (June 1971 to May 1972) by Oviatt and Nixon (1973). NMFS provides species and stock-specific catch data for the southern New England area (for example, see Nitschke et al. 2000).

Fishery trends in Mt. Hope and Narragansett Bays

The fish community of lower Narragansett Bay has been monitored via weekly trawls since 1959 and has been more extensively analyzed (Jeffries and Johnson 1974, Jeffries and Terceiro 1985, Jeffries et al. 1988). Additionally, the Rhode Island Department of Fish and Wildlife (RIDFW) has recently summarized RI landings and stock assessment data for several commercially important species (DeAlteris et al. 2000). The above analyses all indicate a recent (post ca. 1983 to 1999) trend of declining abundance of demersal fish and increasing abundance of pelagic fish species in Narragansett Bay. For example, RI landings of demersal winter flounder show dramatic post-1983 declines (discussed in detail below), while RI landings of pelagic longfin squid (*Loligo pealei*) has displayed a nearly 4-fold 1983 to 1999 increase (DeAlteris et al. 2000; see Figure 11 of DeAlteris et al. 2000). This demersal-pelagic dominance swap has also been observed in fishery-independent surveys in lower Narragansett Bay and RI Sound URI/GSO (Figure 5.2, Jeffries and Terceiro 1985).

Jeffries and Terceiro (1985) attributed much of this demersal/pelagic swap to increased availability of food resources due to declines in the abundance of resident and formerly abundant winter flounder. The underutilized food resource is hypothesized to now be used by transient fish (such as scup, butterfish and squid) such that there is a negative correlation between abundance of resident vs. migrant fish in Narragansett Bay. This statistically significant correlation was found by Jeffries and Terceiro (1985) between abundance of resident winter flounder, on the one hand, and migrant ocean pout ($r = -0.55$), red hake ($r = -$

0.47), and silver hake ($r = -0.62$), on the other. This relationship between abundances of transient and Narragansett Bay resident fish has been summarized as a power curve (Jeffries and Terceiro 1985; Figure 5.3).

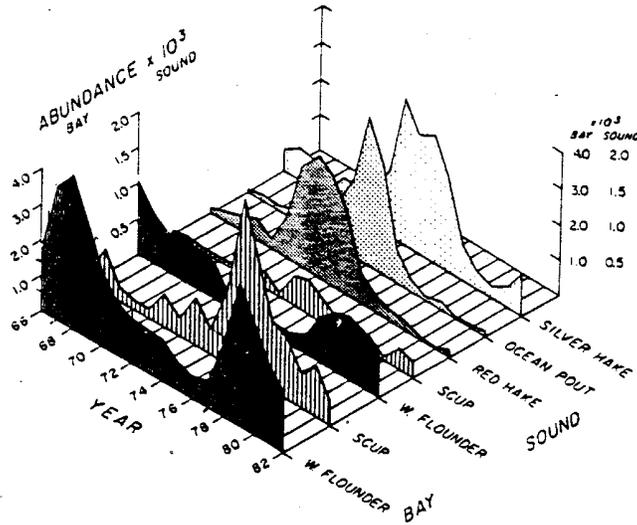


Figure 5.2. Comparison of annual totals of mean monthly trawl catches for demersal fishes in Narragansett Bay and Rhode Island Sound from 1966-1982 (reprinted from Jeffries and Terceiro 1985; permission pending). Note different scales among y-axes.

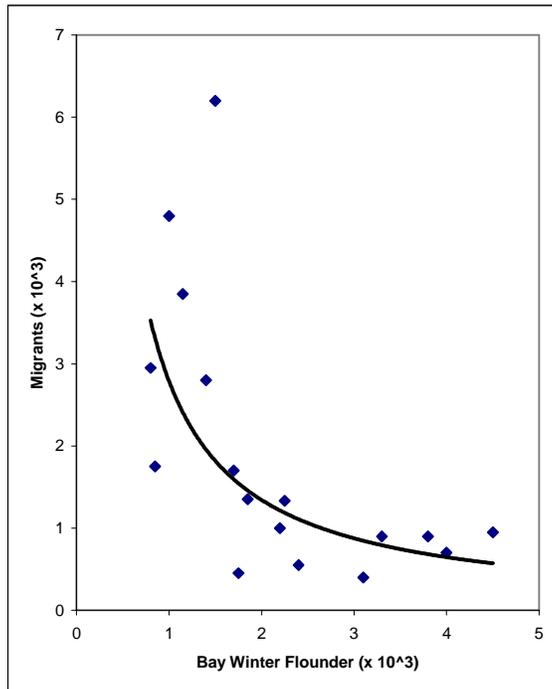


Figure 5.3. Power curve for relation of total annual abundance of Narragansett Bay winter flounder and total migrant species abundance. (After Figure 6 of Jeffries and Terceiro 1985; permission pending.)

More recent assessment of RI and Narragansett Bay fish abundance confirms the trends and pattern, initially identified by Jeffries and Terceiro (1985), of declining numbers of demersal fish and increasing abundance of pelagic species. For example, RIDFW trawl data has shown that Narragansett Bay pelagic species abundance has increased ca. 3-fold, while demersal species abundance has decreased ca. 3-fold between 1980 and 1999 (DeAlteris et al. 2000). During this period, overall fish abundance (biomass) appears to have remained relatively constant, suggesting that Narragansett Bay fish carrying capacity has remained stable, but the partitioning of fish biomass into demersal and pelagic species has changed (DeAlteris et al. 2000). Mt. Hope Bay trawl data (from MRI surveys) are available to determine whether or not similar changes in the proportions of Mt. Hope Bay demersal and pelagic fishes have occurred over the same period.

Trends in Rhode Island landings and stock assessment of commercially important fishery species—namely winter flounder, summer flounder, striped bass, and longfin squid, as well as quahog, oyster and lobster—have recently been reviewed (DeAlteris et al. 2000). Rhode Island winter flounder catch data compiled by DeAlteris et al. (2000) has been presented above. The overall pattern of Rhode Islands fisheries presented by DeAlteris et al. (2000) is consistent with that reported by Jeffries and Terceiro (1985): an overall 1980-1999 decline in the abundance of demersal species and a simultaneous pattern of increase in pelagic species. For example, between 1980 and 1999 the

Narragansett Bay longfin squid index of abundance (based on URI/GSO and RIDEM trawls) has increased ca. 4- to 5-fold, and the Narragansett Bay striped bass index of abundance (based on MRI trawls and recreational CPUE) has increased ca. 4- to 5-fold, while the winter flounder abundance index (based on URI/GSO and RIDEM trawls) has declined ca. 6-fold (DeAlteris et al. 2000). Jeffries and Terceiro (1985) have indicated that the long-term shift from demersal to pelagic species may be the result of subtle temperature-related changes in Narragansett Bay. What roles overfishing and habitat degradation play in this shift is not well known. However, it is important to realize that this phenomenon is not restricted to Narragansett Bay, but rather has occurred on a broad geographic scale throughout the northwest Atlantic fisheries and is generally thought to be a direct result of overfishing (Sinclair and Murawski 1997). For example, over the same 30-year time period, U.S. groundfish stocks declined strongly while those of elasmobranchs and pelagic fishes increased dramatically (Figure 5.4).

Long-term (1972-1997) abundance indices of several Mt. Hope Bay fishes have been compiled from monthly standard trawl surveys (work of Marine Research, Inc., summarized in NEPC and ASA 1998). Eight species (windowpane, winter flounder, scup, weakfish, butterfish, bay anchovy, little skate, hogchoker) account for most (88% in 1997) of the standard trawl catch in the open waters of Mt. Hope Bay (see NEPC 1996 for chart of trawl locations). Trends in these fish abundance indices are summarized here. Window-pane (*Scophthalmus aquosus*) abundance was highly positively correlated with that of

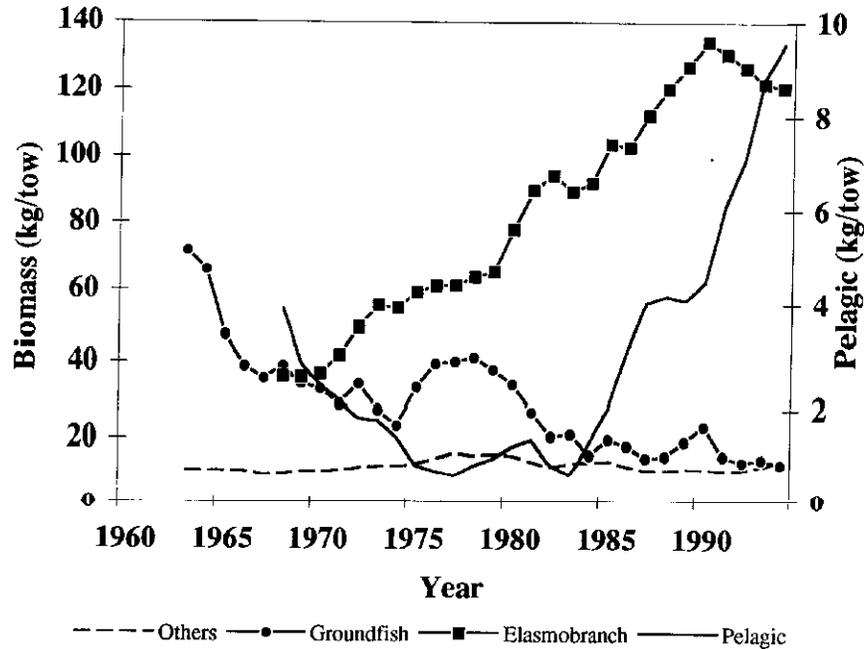


Figure 5.4. Shift in dominance of groundfish species to elasmobranch and pelagic fishes observed in the NW Atlantic. (Reprinted from Sinclair and Murawski 1997; permission pending.)

winter flounder, and displayed a decline from near 11 fish per tow in 1979 to near 0.07 fish per tow in 1991 and 1992. Since the early 1990's, Mt. Hope Bay windowpane abundance, unlike that of winter flounder, has increased slightly to ca. 0.7 to 0.9 fish per tow in 1996 and 1997. Comparison of Mt. Hope Bay windowpane abundance indices to those of various sites in Connecticut (DEP-Niantic River), lower Narragansett Bay Rhode Island (URI/GSO trawl survey), and Massachusetts (Massachusetts Division of Marine Fisheries and Plymouth Nuclear Power Station) generally revealed similar patterns and trends in windowpane abundance, indicative of regional synchrony in this species' abundance patterns. Another flatfish, the hogchoker, has shown post-1985 declines in Mt. Hope Bay. The hogchoker is a small flatfish (usually <6 inches) for which there is no commercial or recreational fishery, so fishing mortality is

not likely to play a role in the abundance cycles of this fish. The Mt. Hope Bay hogchoker abundance index declined from near 1 to 2 fish per tow in 1972-1985 to a level of 0.1 fish per tow in 1986-1997 (NEPC and ASA 1998). In the late 1990's, modest gains in Mt. Hope Bay hogchoker abundance were observed, with abundance indices of 0.35 (in 1997) to 0.45 (in 1996) fish per tow. As with windowpane, patterns of Mt. Hope Bay hogchoker abundance were similar to those of lower Narragansett Bay (URI/GSO trawl survey) and Connecticut estuaries (CT DEP).

Mt. Hope Bay tautog abundance displayed a decline over 1972 to 1997, with a sharp decline post-1985 (see Figure F-14 of NEPC and ASA 1998). For the standardized MRI trawl, the tautog abundance index declined from 1 to 2 fish per trawl in 1986 to <0.1 fish per trawl in 1997. In 1997, for the first time in the 1972 to 1997 time series, no tautog were caught at the long-term fixed trawl location in Mt. Hope Bay. The ca. 10-fold decline in the Mt. Hope Bay standard trawl tautog catch represents a recent (post-1985) decline in this species that has also been observed in lower Narragansett Bay (URI/GSO trawl data) and in Connecticut estuaries (see NEPC and ASA 1998). However, Dorf and Powell (1997) recently reported that Mt. Hope Bay is an important spawning area for tautog and that macroalgae is an important nursery habitat for the species.

Scup are Narragansett Bay summer migrants, present in Mt. Hope Bay in May through October. Unlike winter flounder, windowpane, hogchoker and tautog, there have been no recent abrupt declines in Mt. Hope Bay scup abundance. The Mt. Hope Bay scup abundance index has remained near 5 to 10

fish per tow—with peaks of 20+ fish per tow in 1976 and 1980—from 1972 to ca. 1992, with a slight decline evident in the late 1990's (NEPC and ASA 1998). The recent decline in Mt. Hope Bay scup is not consistent with the inverse pattern of winter flounder and scup abundance seen in the 1970's to 1980's in Narragansett Bay (Jeffries and Terceiro 1985). Butterfish also showed an inverse pattern of abundance to that of winter flounder in lower Narragansett Bay (Jeffries and Terceiro 1985). This inverse pattern was not seen in Mt. Hope Bay, however, where butterfish abundance indices have shown no trend over the period from 1972 to 1997 (NEPC and ASA 1998). The abundance of little skate displayed a gradual increase in Mt. Hope Bay (NEP and ASA 1998). Although never abundant (usually <0.5 fish per tow, except ca. 1 fish per tow in 1992) in Mt. Hope Bay, little skate numbers there have risen significantly (sign test $p=0.046$, NEPC and ASA 1998) from an index near 0-0.1 fish per trawl in the 1970's, to 0.1-0.9 fish per trawl in the 1990's. This increase is consistent with observed increases in lower Narragansett Bay (URI/GSO trawls) little skate abundance (NEPC and ASA 1998).

Winter Flounder

Because of their former abundance and recent precipitous decline, winter flounder have been a focus of fishery research in Mt. Hope and Narragansett Bays. Historically, winter flounder (*Pseudopleuronectes americanus*) have been a dominant component of the Narragansett Bay fish community, comprising >50% of the fish in lower Narragansett Bay (Oviatt and Nixon 1973, Jeffries and

Johnson 1974). The winter flounder breeds in winter and early spring, spawning between January and May in southern New England (Bigelow and Schroeder 1953). In the post-yolk-sac stages of development, larval winter flounder first feed on diatoms (Bigelow and Schroeder 1953), and in later stages on a diet of crustaceans, especially isopods (Bigelow and Schroeder 1953) and benthic invertebrates (Jeffries and Johnson 1974). The first year of winter flounder life, during which a length of ca. 100 mm is attained, is complicated by the physiological and behavioral changes associated with the metamorphosis from symmetrical planktonic larvae to dorso-ventrally flattened benthic juveniles. These changes, especially in the first three months of development, have been described by Jearld et al. (1992).

There is a tendency for the fry to be concentrated in nearshore areas (bays, estuaries) and move offshore as they grow older (Bigelow and Schroeder 1953, Percy 1962). Winter flounder tend to migrate into estuarine embayments in winter (spawning season), followed by migration to deeper offshore (RI Sound) waters in summer (Nitschke et al. 2000). There is evidence that some RI winter flounder migrate easterly as they move offshore, with RI-tagged fish being recovered in Vineyard and Nantucket Sounds, but most tagged RI winter flounder have been recovered in Narragansett Bay and RI Sound (Powell 1989). Natal fidelity, or homing instinct, the return to the same spawning locations, has been reported for winter flounder by Bigelow and Schroeder (1953). In mark and recapture studies, up to 90% of fish tagged in bays and estuaries of New England and Long Island were later recaptured in the same bay or estuary (Perlmutter

1947, Bigelow and Schroeder 1953). Saila (1961), in a 2-year mark/recapture study, found that winter flounder from the south coast of Rhode Island dispersed as far as 54 miles from the original tag location in summer, but returned to nearshore bays and salt ponds in winter. The 20-fathom isobath has been suggested as a limit to the seaward distribution of winter flounder (Saila 1961). Winter flounder migrations of tens of miles, such as a Block Island to Georges Bank migration, have also been documented (Bigelow and Schroeder 1953). A recent tag-recapture study in the Mid-Atlantic Bight also found high fidelity of winter flounder to specific estuarine systems (Phelan 1992). A similar pattern of overall natal fidelity mixed with occasional migration has recently been documented, via genetic analysis, for another common coastal fish (weakfish, *Cynoscion regalis*) metapopulation (Thorrold et al. 2001). The life history of winter flounder has been reviewed by Grimes et al. (1989), while Pereira et al. (1999) have recently reviewed the habitat requirements of various winter flounder life stages.

Narragansett Bay winter flounder appear to be part of the southern New England/Mid-Atlantic population group that is one of three (north of Cape Cod, south of Cape Cod, Georges Bank) separately recognized populations groups (Nitschke et al. 2000). Dramatic declines in winter flounder abundance have been reported in all three areas. Rhode Island winter flounder catch followed a pattern of elevated abundance in the 1970's and early 1980's followed by a drastic decline beginning in ca. 1982 (DeAlteris et al. 2000). During this period, RI

winter flounder landings declined from a peak of 9.3 million pounds in 1981 to a low of 1.3 million pounds in 1998 (DeAlteris et al. 2000; see Figure 5.5).

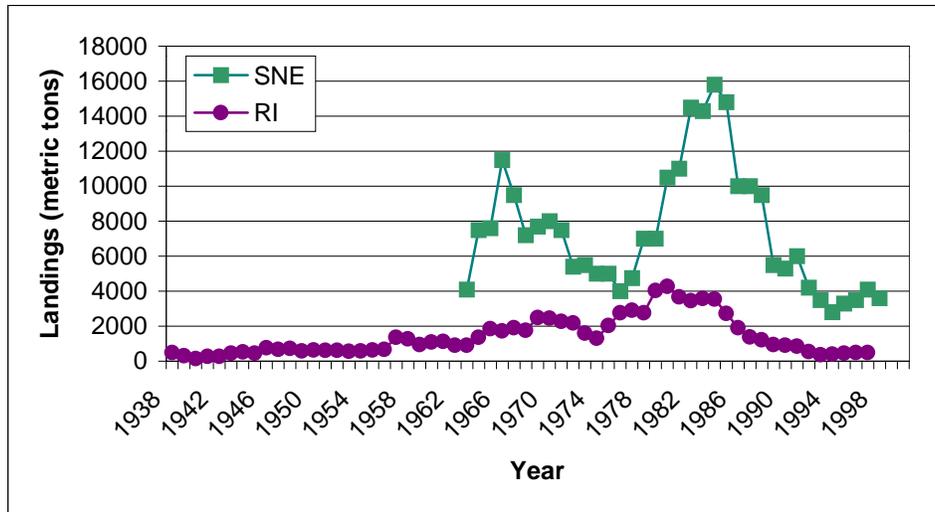


Figure 5.5. Rhode Island (RI) and southern New England/Mid-Atlantic (SNE) winter flounder landings time series. (Rhode Island data after DeAlteris et al. 2000; southern New England data after NMFS data summarized by Nitschke et al. 2000.)

Very little recovery in RI winter flounder abundance or catch is apparent in the data (to 1998) presented by DeAlteris et al. (2000). NMFS data indicate that Gulf of Maine total landings follow a similar pattern, having declined from ca. 3,500-5,000 metric tons per year in 1980-1982 to <100 metric tons per year in 1995-1998 (Nitschke et al. 2000; see Figure 5.6). Similarly, a pattern of decline was also seen in the Georges Bank population, with winter flounder commercial landings falling from sustained levels of 2,000-4,000 metric tons per year in the 1970's and early 1980's to levels of <2,000 metric tons per year in 1990-1998 (Nitschke et al. 2000; Figure 5.7).

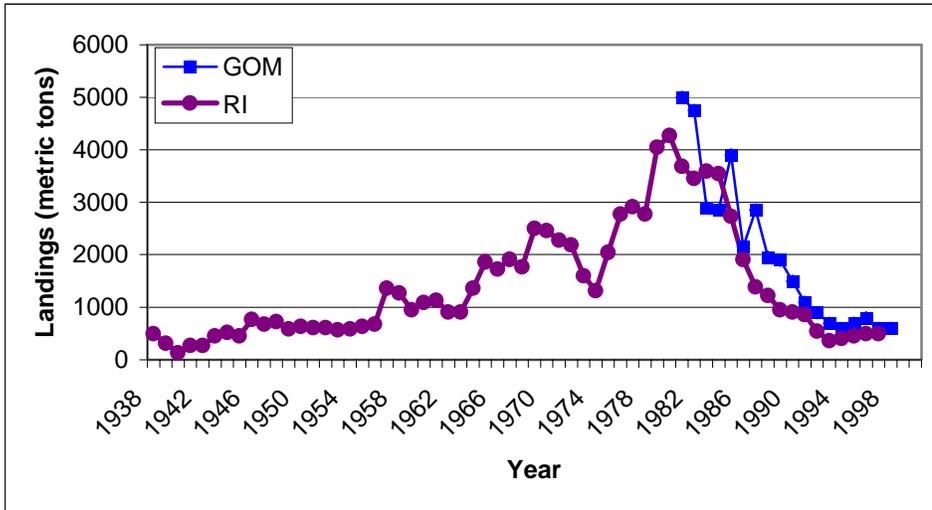


Figure 5.6. Rhode Island (RI) and Gulf of Maine (GOM) winter flounder landings time series. (Rhode Island landing data after DeAlteris et al. 2000; GOM landings after NMGS data summarized by Nitschke et al. 2000.)

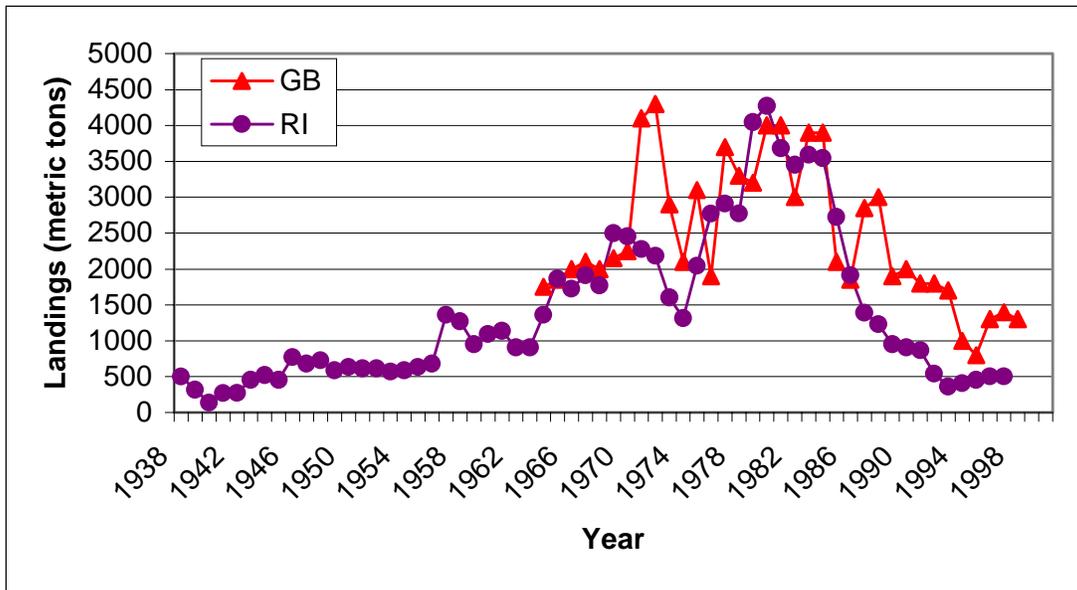


Figure 5.7. Rhode Island (RI) and Georges Bank (GB) winter flounder landings time series. Rhode Island data after DeAlteris et al. 2000; Georges Bank data after NMFS data summarized by Nitschke et al. 2000.)

For both Gulf of Maine and Georges Bank winter flounder, overfishing has been implicated as the cause for recent population declines (Nitschke et al. 2000). In the southern New England/Mid-Atlantic group, total landings declined from a peak of ca. 12,000 to 15,000 metric tons per year in 1980 to 1985 to levels of 5,000 metric tons per year or less in 1993 through 1998 (Nitschke et al. 2000; see Figure 5.8). In the southern New England/Mid-Atlantic group, fishing mortality

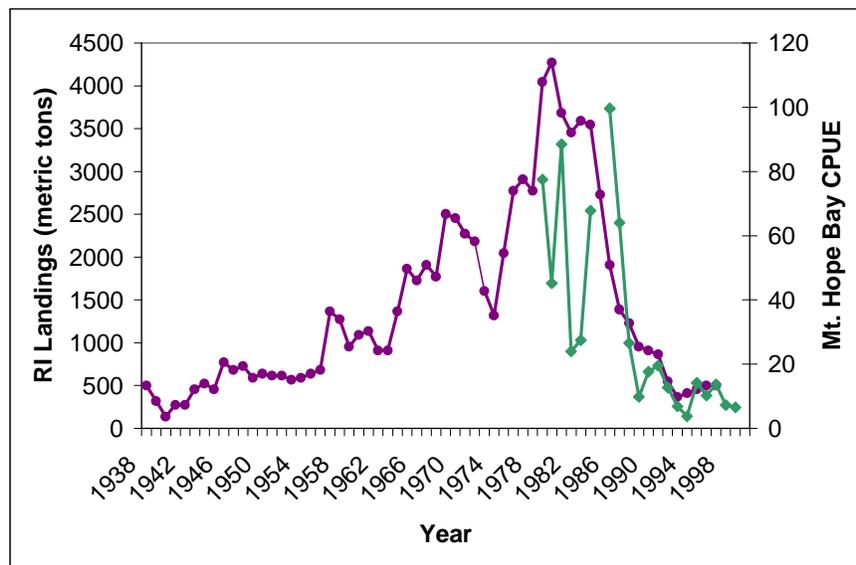


Figure 5.8. Rhode Island (RI) winter flounder landings (filled circles, after DeAlteris et al. 2000) and Mt. Hope Bay winter flounder catch per unit effort (CPUE; filled diamonds). (CPUE data is based on MRI trawl surveys as analyzed and presented by T. Englert at the October 2001 Mt. Hope Bay Workshop.)

was extremely high in 1981-1993, resulting in population declines (Nitschke et al. 2000). Since 1993, reductions in fishing mortality have led to rebuilding of winter flounder biomass, from a low estimate of 8,000 metric tons in 1992 to an estimated 22,300 metric tons in the southern New England group in 1998 (Nitschke et al. 2000; Figure 5.8). The recent (post-1981) decline in RI winter flounder landings, as well as the late 1970's to early 1980's increase, appears to be consistent with the population fluctuations experienced in neighboring (Gulf of

Maine, Georges Bank) population groups. A single pattern of flounder abundance appears to be applicable to all of southern New England; similar patterns of population fall and rise are observed in lower Narragansett Bay, Providence River (Manchester Street Electric Plant), Mt. Hope Bay (BPPS), and offshore populations (Jeffries et al. 1989). However, recent signs of stock rebuilding in winter flounder stocks, including in Rhode Island Sound, have not been mirrored in Narragansett and Mt. Hope Bays, where populations remain depressed (DeLong and Collier 2001).

From mid-1971 to the present, Mt. Hope Bay winter flounder abundance has been assessed via monthly to twice-monthly trawls carried out by MRI. The raw catch numbers have been compiled into an annual catch per unit effort (CPUE) index of Mt. Hope Bay winter flounder abundance for 1979 through 1999 (T. Englert, personal communication). This Mt. Hope Bay winter flounder CPUE index displayed a pattern of elevated CPUE in the early to mid-1980's followed by a precipitous decline in the late-1980's that is similar to the pattern of RI winter flounder landings during the same time period (Figure 5.8).

Narragansett Bay winter flounder population fluctuations have been linked to fluctuations in winter water temperature (Jeffries and Johnson 1974, Jeffries and Terceiro 1985, Keller and Klein-MacPhee 2000). Air temperatures along the Atlantic coast of the United States have displayed a recent increase (Ford 1996), and Narragansett Bay winter water temperatures have increased by 2-3°C between ca. 1960 and 1996 (Cook et al. 1998, Keller et al. 1999, Keller and Klein-MacPhee 2000). Winter flounder abundance has been statistically linked to

variations in winter water temperature. However, these variations, on the order of $\pm 2\text{-}3^{\circ}\text{C}$, are well within the physiological thermal tolerance of winter flounder, suggesting that some secondary, rather than direct, temperature effects are the controlling mechanism (Jeffries and Johnson 1974, Jeffries and Terceiro 1985, Jeffries et al. 1989). Mortality at early life-stages (i.e., larvae through young-of-year) is likely the main source of variation in determination of fish year-class strength (Houde 1987). This mortality appears to be dependent on food availability at time of first feeding, oceanographic conditions (i.e., larval retention mechanisms, etc.) and predation (Cowan and Houde 1990). Predation on winter flounder larvae at metamorphosis, which would be expected to be elevated in warmer winters due to increased activity of benthic predators, has been suggested as a potential mechanism behind winter flounder declines in Narragansett Bay (Jeffries and Terceiro 1985).

This hypothesis has been tested in a series of controlled Narragansett Bay (MERL) mesocosm studies (Keller and Klein-MacPhee 2000). The MERL-simulated physical and biological marine environment is similar to that of lower Narragansett Bay (see Sullivan and McManus 1986 and references therein). In their study, Keller and Klein-MacPhee manipulated the temperature of the mesocosms to compare winter flounder development and survival rates in February and April of 1997 at cooler than mean (ca. 1.86°C mean temperature) and at warmer than mean (ca. 5.11°C) water temperatures. There was experimental replication, with three cool treatment tanks and three warm treatment tanks. Temperatures were not held fixed at these levels, but tracked the

vernal warming of Narragansett Bay, with the warm treatment being raised ca. 0.7°C warmer than the Bay and the cool treatment being cooled ca. 2.5°C cooler than the Bay. Cold tanks ranged from near 0°C in early February to near 2.5°C in early April, while warm tanks warmed from 3°C in early February to near 6°C in early April (see Figure 1 of Keller and Klein-MacPhee 2000).

Several significant differences were found between the two experimental treatments manipulated by Keller and Klein-MacPhee (2000). Phytoplankton biomass was significantly higher in warm tanks, although diatoms (especially *Skeletonema costatum* and *Chaetoceros* spp.) were more abundant in the cold tanks. Abundance of copepod nauplii, copepodites and adults (mainly *Acartia hudsonica*) was higher in the warm treatment. Flounder hatching success was significantly higher in cold (91 to 94 %) than warm (65 to 82 %) treatments, and hatching-size larvae in cold tanks were significantly larger than those in warm tanks. However, growth rates of those larvae that did hatch were higher (0.016 to 0.018 day⁻¹) in warm versus cold (0.011 to 0.013 day⁻¹) treatments. Thus, there was greater hatching success and larger first-hatch larvae in the cold treatment, but greater rates of larval growth in the warm tank. Greatest larval survival was found in the cold treatment with lowest food availability, suggesting that winter flounder larval survival is dependent on larval mortality (predation) rather than food availability (Keller and Klein-MacPhee 2000). The sand shrimp *Crangon septemspinosa* was present in both treatments, but more active in the warm treatment (in which *Crangon* was seen actively swimming and feeding in the benthos) than in the cold treatment (in which *Crangon* was inactive and buried in

the sediment). There was a highly significant positive correlation between benthic predator activity and larval flounder mortality (see Figure 10 of Keller and Klein-MacPhee 2000), leading Keller and Klein-MacPhee to suggest that warm winter benthic predation may be a primary mechanism behind recent winter flounder year class failure and the decline in Narragansett Bay winter flounder.

Several lines of evidence support Keller and Klein-MacPhee's conclusion. Winter flounder larvae that developed in cool temperatures (2°C) had higher RNA and protein content, indicative of good health, than did those raised in warm (7°C) temperatures (Buckley et al. 1990). Water temperature also significantly affects winter flounder age at metamorphosis and size at metamorphosis, with fish raised at warmer temperatures (10°C) metamorphosing ca. 30 days earlier and being ca. 15% longer than those reared at cool temperature (5°C; Chambers and Legett 1992). The relation between winter flounder juvenile body size and predation has been examined by Whitting (1993). Predation of recently settled larvae of a similar demersal fish (plaice, *Pleuronectes platessa*) by *Crangon* has also been documented in the Wadden Sea (Van der Veer and Bergman 1987). *Crangon* can be much more abundant in Narragansett Bay (to 80 animals m⁻²; Thornton-Whitehouse 1994) than the low abundance (ca. 5 m⁻²) used in Keller and Klein-MacPhee's experiment, suggesting that the impact of *Crangon*-related winter flounder mortality in Narragansett Bay may be even greater than that observed in the above-described mesocosm experiments.

At all life stages, winter flounder are a prey species for many common Mt. Hope Bay fish and invertebrates. Larval winter flounder are preyed upon by

small medusae of *Sarsia tubulosa* (Pearcy 1962). The mud anemone (*Ceriantheopsis americana*) has been suggested as a predator on winter flounder larvae in Narragansett Bay (Klein-MacPhee et al. 1993). Summer flounder were found to prey heavily on juvenile winter flounder in a New Jersey estuary, and in fact preferred winter flounder over alternative prey (Manderson et al. 1999). The importance of habitat type was underscored by the finding that predation was lower in eelgrass than in macroalgae. The striped searobin was also found to be a significant predator of juvenile winter flounder (15-70 mm TL) in New Jersey estuaries (Manderson et al. 2000). Summer flounder and searobin are both common in Mt. Hope Bay, with the former accounting for 6% of the catch during 2000 (USGen 2001). Bluefish, *Pomatomus saltatrix*, common in Mt. Hope Bay in summer, prey upon juvenile winter flounder according to Howe et al. (1976), who also suggest that birds (especially gulls and cormorants) are an important source of juvenile winter flounder mortality. The seven-spine shrimp (*Crangon septemspinosa*, mentioned above in conjunction with the work of Keller and Klein-MacPhee 2000) was found to prey on young-of-the-year winter flounder of 10 to 20 mm length (Witting and Able 1995). A compilation of known fish predators of juvenile winter flounder, compiled by Pereira et al. (1999), included summer flounder (*Paralichthys dentatus*), windowpane (*Scopthalmus aquosa*), and sea robin (*Prionotus evolans*). Changing relative abundance of different fishes (i.e., increases in searobin abundance; Manderson et al. 1999) may lead to increased predation of young winter flounder. According to a summary by Pereira et al. (1999), adult winter flounder are known to be prey items for striped

bass (*Morone saxatilis*), spiny dogfish (*Squalus acanthus*), goosefish (*Lophius americanus*), oyster toadfish (*Opsanus tau*), and sea raven (*Hemitripterus americanus*). Like juveniles, adult winter flounder are preyed upon by birds, including cormorants, blue herons and osprey (Pearcy 1962). Harbor seals (*Phoca vitulina concolor*) are also known to prey upon adult winter flounder, and Harbor seals are often observed in Mt. Hope Bay in winter (Post 1998). However, flounder (as a group of five species) made up less than 10% of the harbor seal diet in a study by Payne and Selzer (1989).

The decline in Narragansett Bay and Mt. Hope Bay winter flounder abundance has recently been examined by Collie and DeLong (2001), who analyzed time series of winter flounder abundance at several locations in Narragansett and Mt. Hope Bays to identify potential mechanisms for winter flounder decline. Previous work (Keller and Klein-MacPhee 2000) had identified the potential for climate-related increases in predation and mortality at the egg-larvae stages of winter flounder in Narragansett Bay. In their analyses of 1973-1999 MRI Mt. Hope Bay winter flounder data and 1979-1999 RIDEM data, Collie and DeLong (2001) identified two "bottlenecks," or periods of increased mortality, that are related to total winter flounder mortality: the egg-larvae period and the age-1 fall to age-2 spring period. The correlation coefficient of age-1 fall to age-2 spring mortality with total mortality ($r = 0.80$) was much larger than that of egg-larvae period ($r = 0.37$), or any other period examined, indicating that the age-1 fall to age-2 spring period is the period to which much of the decline in Mt. Hope Bay winter flounder abundance can be attributed (Collie and DeLong 2001).

Further, mortality rate at various stages (age-1 spring to age-1 fall, juveniles June to October) displayed significant positive correlation with bottom temperature, suggesting temperature-dependent mortality of Narragansett Bay winter flounder (see Collie and DeLong 2001, DeLong et al. 2001). Temperature-dependent juvenile winter flounder mortality may be either a direct physiological response or an indirect response to other temperature-related changes such as availability of prey (DeLong et al. 2001). Juvenile winter flounder prefer warmer water than do adults (Casterlin and Reynolds 1982) and are known to feed in shallow, nearshore areas such as tidal creeks (Rountree and Able 1992a,b, 1993) and intertidal mud flats (Tyler 1971, Wells et al. 1973). The distribution of preferred juvenile winter flounder habitat (shallow waters) in Mt. Hope Bay is shown in Figure 5.9. Note that this area is also the most impacted by BPPS heated water effluent (see Chapter 2 herein). Such shallow regions are also expected to be more susceptible to the habitat-degrading effects of such factors as warming, development, and eutrophication than are the deeper regions of Mt. Hope Bay. Loss of suitable habitat in these shallow regions used by age-1 to age-2 winter flounder may be a mechanism contributing to elevated mortality at this developmental period and the overall decline in winter flounder in Narragansett and Mt. Hope Bays.

Other findings of Collie and DeLong's (2001) analyses include a recent (post-1988) trend of declining adult female winter flounder in Narragansett Bay relative to their abundance in Rhode Island Sound (although this analysis is based on a single station in each area; see Figure 5.1), suggesting a relative depletion of Narragansett Bay winter flounder spawning stock. Spatial-temporal analyses of

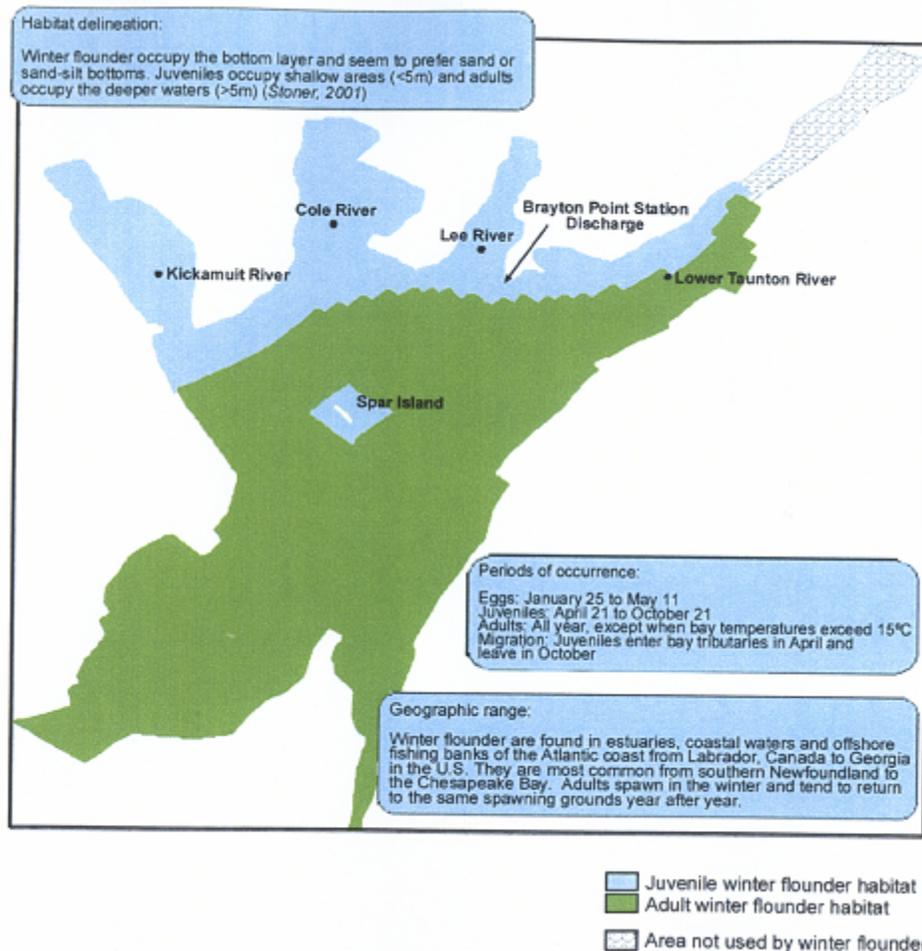


Figure 5.9. Habitat delineation of juvenile winter flounder habitat as determined by the BBPP (USGen 2001).

available winter flounder data (Collie and DeLong 2001) indicate that the habitat suitability of Mt. Hope Bay ("sector 3" of their analyses) has increased for large and small winter flounder during the spring, but that fall habitat suitability has decreased.

Water temperature is a critical component of winter flounder habitat (Pearcy 1962, Sogard 1992, Pereira et al. 1999, Stoner et al. 2001), and autumn has been identified as the season in which Mt. Hope Bay water temperature is most affected by anthropogenic thermal modification (Carney 1997, Mustard et al.

1999). While anthropogenic temperature modification of Mt. Hope Bay (on the order of 0.8°C; Mustard et al. 1999) is within the physiological range of winter flounder (see Table 5.1), subtle temperature-dependent changes in habitat (e.g., changes in food availability), behavior, etc., may be influencing winter flounder mortality at key developmental stages such as the age-1 fall to age-2 spring period identified by Collie and DeLong (2001). Sediment type has also been identified as of critical importance to juvenile winter flounder (Phelan et al. 2001). Small juveniles prefer fine grain sediments, while larger juveniles prefer larger grain sediments due to size-related changes to burrowing ability. However, prey availability was found to override sediment type preference under experimental conditions.

Winter Flounder and Habitat Change

Watershed development and related nutrient loading often result in degraded estuarine water quality and degraded fish habitat, with an ultimate expression as reduced fish stocks (Polgar et al. 1985, Valiela et al. 1992). Assessments of Rhode Island coastal water habitat quality on winter flounder growth rate have indicated that dissolved oxygen concentration and benthic prey abundance and type are important determinants of winter flounder distribution (Meng and Powell 1999, Meng et al. 2000, Meng et al. 2001). Similar results were observed in Mid-Atlantic Bight estuaries (Bejda et al. 1992). Additionally, recent warm Narragansett Bay winter water temperatures may represent a degradation of the winter flounder's ideal spawning habitat (Jeffries 1994, Keller

Table 5.1. Adult winter flounder thermal tolerance data.

Exposure Temp(C)	Acclimation Temp(C)	Exposure Time	Response	Source
<u>Heat exposure:</u>				
19	4 -	-	upper lethal T	McCracken 1963
12-15	-	-	preferred temperature	McCracken 1963
13.5	- -	-	preferred temperature	Reynolds 1977
18.5	- -	-	preferred T (yearlings)	Casterlin & Reynolds 1982
22	5	4-64m	'low' mortality	Itzckowitz & Schubel 1983
22	7 -	-	critical thermal maximum	Duffy and Luders 1978
22.5	17-23	-	burial and inactivity	Olla et al. 1969
23.7	14	-	critical thermal maximum	Duffy & Luders 1978
24.5	14	-	avoidance behavior	Duffy & Luders 1978
24.5	22	-	avoidance behavior	Duffy & Luders 1978
24.9	20	-	avoidance behavior	Duffy & Luders 1978
26-32	4-23	-	critical thermal maximum	Everich & Gonzalez 1977
26.5	20	-	upper lethal temperature	McCracken 1963
27	21	-	critical thermal maximum	Duffy & Luders 1978
27.5	24	-	avoidance behavior	Duffy & Luders 1978
28	22-28	48h	upper lethal temperature	Hoff & Westman 1966
28-30	5	4m	100% mortality	Itzckowitz & Schubel 1983
29.1	28	-	critical thermal	Duffy & Luders 1978
30.6-31.4	20	1h	upper lethal temperature	LI Light Co. 1977
<u>Cold exposure:</u>				
1.0	21	48h	lower lethal temperature	Hoff & Westman 1966
5.4	28	48h	lower lethal temperature	Hoff & Westman 1966

and Klein-MacPhee 2000). First settling winter flounder (ca. 25 mm long or less) prefer the fine-grained, high-organic-content sediments typical of depositional areas where there is increased availability of suitable prey items (calanoid and harpacticoid copepods and small spionid polychaetes; see Meng and Powell

1999, Phelen et al. 2001, Stoner et al. 2001). As the juvenile fish grow to ca. 25 to 55 mm length, their diet still includes these same prey items, but the dietary proportion of larger polychaetes increases. In upper Narragansett Bay, the polychaete *Polydora cornuta* seems to be an especially important food for ca. 30mm winter flounder: when present, *P. cornuta* comprised a mean of 70% (by volume) of young winter flounder stomach contents (Meng et al. 2001). Delta N-15 analyses of Narragansett Bay young winter flounder (ca. 30 mm length) show that they feed 1-2 trophic levels above primary producers, suggesting a macroalgae-harpacticoid copepod-polychaete food chain (Meng and Powell 1999). With attainment of ca. 55 mm length, the winter flounder diet shifts from copepods to larger crustaceans like amphipods and mysids and includes larger prey items such as large polychaetes (Stoner et al. 2001). The trend towards the proportion of polychaetes in the diet increasing with winter flounder age continues until adulthood; adult winter flounder have been placed in a benthivore guild (Garrison and Link 2000), with a diet composed of 43% (by volume) polychaetes, followed in volume by amphipods and isopods. Similarly, in a review of Cape Hatteras to Nova Scotia winter flounder stomach contents, Langton and Bowman (1981) found an adult winter flounder diet dominated by polychaetes. Changes in abundance or species composition of the Mt. Hope Bay benthic community, which in turn may affect winter flounder feeding, have not been noted between the mid 1970's and 1997/1998 at four open-bay monitoring locations (MRI 1999).

Winter flounder prefer relatively open sand/silt barren bottom, but may prefer such areas that are adjacent to eelgrass (*Zostera marina*) beds that provide a refuge from predators (Baltz et al. 1993, Meng and Powell 1999). Declines in the extent of coverage of Narragansett Bay eelgrass beds are well documented. Eelgrass beds, which were historically present throughout Narragansett Bay, including the upper bay in Greenwich Cove and the mouth of the Palmer River, are now only found in the lower bay, i.e., south of Jamestown (RIDEM 2000; compare Figures 3.1 and 3.2 herein). Recent reports indicate that there are no eelgrass beds remaining in Mt. Hope Bay (Rines 2001 cited in USGen 2001). Loss of this potentially important micro-habitat may be a factor in recent winter flounder declines. Additionally, profusion of macroalgal mats (especially *Ulva lactuca*), enhanced by N-loading in the upper reaches of Narragansett Bay, may represent a significant degradation in winter flounder habitat. In the Raritan/Navesink estuary (northern New Jersey), *Ulva* cover of up to ca. 20 g wet wt. m⁻² increased the probability of finding 55 mm or greater winter flounder, while the fish avoided *Ulva* densities of >20 g wet wt. m⁻² (Stoner et al. 2001). The heavy *Ulva* cover observed by Stoner et al. (2001) may degrade winter flounder habitat by impeding motility, limiting access to food items in/on the sediment and, when the *Ulva* mat dies and decomposes, leading to reduced dissolved oxygen concentration. These field findings are consistent with the experimental work of Timmons (1995), who found that young winter flounder avoided high concentrations of *Ulva*.

The feeding strategies of winter flounder—i.e., association with shallow, typically nutrient-enriched and depositional areas of estuaries—place them in jeopardy when estuaries become over-enriched (eutrophic). The same features of the estuarine habitat that provide an organically enriched, depositional sediment where deposit-feeding polychaetes thrive, also make that habitat susceptible, when subject to excessive nutrient loading, to low dissolved oxygen concentration and heavy macroalgal cover. For example, in a comparison of winter flounder growth rate in Connecticut, Miese et al. (1999) found that winter flounder growth at the relatively polluted New Haven Harbor was greater than that at the relatively clean Connecticut River estuary. Similarly, Meng et al. (2001) found that winter flounder growth at three sites along the estuarine gradient in Narragansett Bay—from near Providence (estuarine) south to Jamestown (coastal water)—was highest at the mid-bay site. Growth rate was initially greatest at the estuarine site (Gaspee Point, near Providence) but declined in response to decreasing dissolved oxygen (as low as 2.8 mg L^{-1}) concentration, which eventually reduced benthic prey availability. At the lower bay site (Sheffield Cove, Jamestown, RI), oxygen concentration was not growth-inhibiting at 19 mg L^{-1} (winter flounder growth declines at ca. $2.5 \text{ mg O}_2 \text{ L}^{-1}$; Bejeda 1992), but food availability may have reduced growth rate. The mid-bay site (Prudence Island, RI), with intermediate dissolved oxygen concentration (ca. $18 \text{ mg O}_2 \text{ L}^{-1}$) and greatest prey availability, sustained highest winter flounder growth rates (Meng et al. 2001). Areas of upper Narragansett Bay, such as near Gaspee Point, previously identified as primary juvenile winter flounder habitat areas (Meng and Powell 1999), may now be

suffering habitat degradation through the dissolved-oxygen-decreasing effects of eutrophication (Granger et al. 2000). Mt. Hope Bay is thought to be a secondary juvenile winter flounder habitat to Narragansett Bay (Meng and Powell 1999). If eutrophication-induced declines in Mt. Hope Bay bottom dissolved oxygen concentration are reaching the growth-limiting levels (ca. 2 to 3 mg O₂ L⁻¹) observed at Gaspee Pt. (Meng et al. 2001), they represent a significant degradation of winter flounder habitat. Diel pattern in dissolved oxygen concentration observed in Mt. Hope Bay (Howes, unpublished data) may also represent habitat degradation, as winter flounder have been shown to grow more slowly when exposed to diel fluctuation in DO (Bejda et al. 1992).

Summary

It is clear that much is known about the population dynamics and life history of winter flounder in general, and specifically about Mt. Hope Bay populations. However, important gaps in our knowledge are also evident. Important phases in the life history of winter flounder, illustrated in Figure 5.10, can be used to summarize data gaps for the species. Numbers in the following sections refer to critical phases of the movements and behavior of adult winter flounder as illustrated in Figure 5.10a.

1) Migration into the estuary/estuarine fidelity: Once of the most crucial questions is how Mt. Hope Bay contributes to the regional stocks. We've seen that population trends for Mt. Hope Bay mirror those for Narragansett Bay, Rhode Island Sound, and in fact, for the entire NW Atlantic to a large degree. In

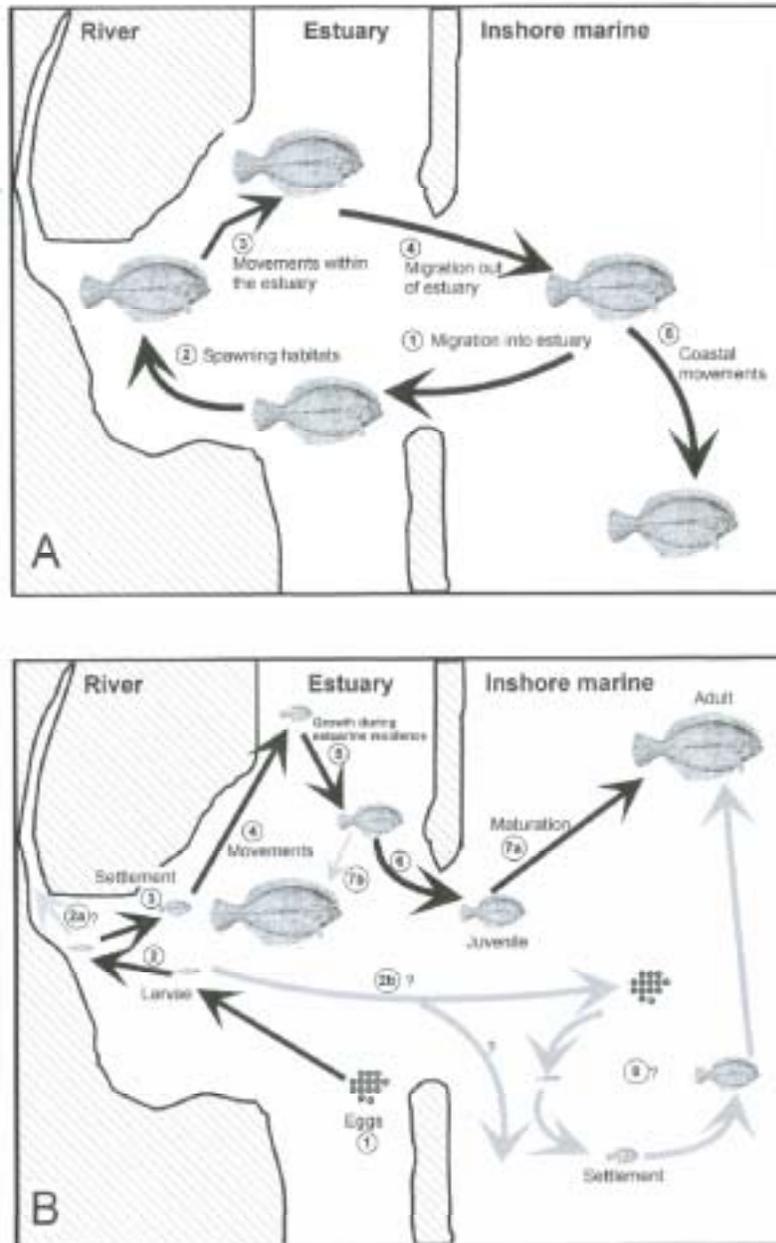


Figure 5.10. Schematic of the life history pattern of winter flounder. Movement patterns of adults are shown separately at the top for clarity. (See text for explanation.)

order to tease apart local from regional sources of temporal trends, we need to determine to what extent adults return to the same estuary each year. As we have reviewed, there is mounting evidence that winter flounder exhibit a high degree of

fidelity to particular estuaries. What percent of surviving adults return to the estuary? Just as importantly, we need to define what constitutes the “home” estuary. Is it Mt. Hope Bay or the greater Narragansett Bay—i.e., what is the spatial scale on which homing operates? These are critical questions because they in effect define population boundaries for the modeling efforts. In addition, it would be important to quantify sources of adult mortality for Mt. Hope Bay relative to Narragansett Bay, the Sakonnet River and Rhode Island Sound. What is the fishing mortality on MHB winter flounder while they summer in Rhode Island Sound? Another important issue is the mechanism(s) regulating the timing of immigration into the estuary from coastal waters. Although we have good data on the general seasonal migration pattern, identification of environmental triggers for migration would greatly enhance modeling efforts and allow us to understand annual variations.

2) Spawning habitats: Identification of specific spawning areas/habitats is currently uncertain, but there is evidence for spatially distinct sites in Rhode Island estuaries (Crawford and Carey 1985). The location of distinct spawning sites in MHB would be of great importance to modeling the impact of various natural and anthropogenic factors on spawning success. Protection of the habitat quality of these sites would of course be of special concern. Although it is known that spawning occurs at night (Stoner et al. 1999), a more precise determination of the timing of spawning events would enhance efforts to model egg dispersal though the bay. Is spawning synchronized with tidal and/or lunar cycles?

3) Estuarine movements: Quantification of the movement patterns of winter flounder within MHB over daily, weekly and seasonal time scales would also be useful. Limited information from conventional tagging studies suggests relatively little movement, but telemetry studies would be required to elucidate finer-scale movement patterns. Of particular importance is quantification of movements of adults among sections of greater Narragansett Bay, including passage into Mt. Hope Bay via the East Passage and Sakonnet River. Since individual winter flounder spawn repeatedly over a two-month period (Stoner et al. 1999), it is possible that individuals may spawn in many different locations throughout the Mt. Hope Bay. The relative success of eggs spawned in various locations would be an important model input (see below).

4) Migration out of the estuary: A better understanding of factors affecting emigration is also important. Estuarine mortality can be estimated by comparing emigration and immigration numbers, assuming the percentage of adults that remain in the estuary is known. And again, identification of the environmental trigger(s) for emigration would be useful to the modeling efforts. The condition of emigrating individuals might also be used to provide an indication of survival likelihood. The geography of Mt. Hope Bay (the East Passage is less than 800 m wide and the Sakonnet River passage less than 100 m wide) makes it feasible to conduct telemetry studies to quantify these parameters.

5) Coastal movements: The movements of winter flounder within coastal waters are generally known, but studies that identify the natal source of winter flounder fish stocks taken in different coastal areas would certainly be valuable.

Conventional tagging and otolith chemical composition methods—e.g., as has been done for weakfish (Thorrold et al. 2001)—could provide that information. This would provide quantification of fishing mortality specific to Mt. Hope Bay populations as well as other estuarine populations.

Numbers in the following sections refer to critical phases in the early life history of winter flounder as illustrated in Figure 5.10b.

1) Egg dispersal and hatching success: Winter flounder eggs are demersal and stick together and are most common in shallow shoal waters of Narragansett and Mt. Hope Bays (Bourne and Govoni 1988). This suggests limited dispersal; however, since it takes about 2-3 weeks for hatching, significant dispersal from spawning sites is possible. The degree of dispersal from the spawning site is unknown and obviously a function of hydrography and temperature. The impact of dispersal on hatching success is also unknown. Are spawning sites chosen to enhance egg-hatching success? As discussed under the adult spawning phase, identification of optimal spawning habitats is an important factor for consideration in the MHBNL program. Identification of the environmental characteristics of spawning sites is particularly important. The fate of eggs once spawned is likely to be an important modeling focus. The duration of the incubation period is highly variable and dependent on temperature, occurring in as few as 11 and many as 63 days (see review in Able and Fahay 1998). Further quantification of the effect of temperature and duration of incubation on hatching success would be helpful. An examination of the relationship between spawning and riverine discharge may also be of interest, as

peak egg densities in Mt. Hope Bay (Figure 4.6) occur at the onset of the period of peak river discharge in March (Figure 3.19).

2. Larval dispersal: Dispersal of larval winter flounder is highly amenable to modeling efforts and is strongly influenced by tidal, gravitational, and wind-driven circulation patterns (Bourne and Govoni 1988). Because duration of the larval stage is highly physiologically plastic in winter flounder (Chambers and Leggett 1987), larvae can remain in the plankton for extended periods if suitable settlement habitats are not encountered. The timing of peak abundance of winter flounder is known to be highly variable between years (Able and Fahay 1998) and is likely related to annual variations in water temperature. Matching of winter flounder larval peak abundance and planktonic food resources is an important unknown in Mt. Hope Bay. If spawning is delayed and larval abundances are mismatched with plankton food resources, greater mortality can be expected; hence efforts to incorporate plankton population dynamics into winter flounder models are clearly warranted. The influence of water quality parameters (temperature, salinity, turbidity, light, pollutants, etc.) on larval behavior is poorly understood and can have a strong impact on larval retention and dispersal patterns. Larval winter flounder may use tidal stream transport mechanisms to maintain position in the estuary, and/or to move into favorable upper estuarine and shallow tidal habitats. In addition, environmental parameters can have a strong effect on larval survival by affecting their energetics. Physiological responses to even small changes in temperature and salinity can have an important energetic cost to larval fishes, and given that mortality of larval

fishes is thought to be strongly a function of starvation, any increased demand on energy reserves is of concern. Loss of Mt. Hope Bay larval populations to upstream dispersal (3a) and downstream transport out of the estuary (3b) are important input and output parameters for modeling efforts. Downstream transport of winter flounder larvae is possibly significant, as loss rates of 3% per day have been estimated for other estuaries (Pearcy 1962).

3) Settlement: Metamorphosis and settlement to the bottom is an important bottleneck in the life history of winter flounder and other flatfishes (Able and Fahay 1998, Pereira et al. 1999). During this stage winter flounder must adapt to a new suite of predators and prey. Predation by benthic crustaceans such as *Crangon septemspinosa* is an important source of mortality (Witting 1995, Witting and Able 1995). Identification of settlement sites and quantification of settlement site suitability should be high priorities. Beam trawl sampling is preferable to seine and trawl sampling for this type of study. The timing of settlement is also important, as it affects predator and prey availability. As mentioned above, winter flounder appear to be able to delay settlement for considerable periods if suitable habitats are not found.

4) Post-settlement movements: Winter flounder are thought to settle in limited areas and then disperse throughout the estuary. Growing evidence suggests that early settlement stages prefer very shallow tidal waters and develop tidal migration behaviors as they grow. We know little about the movements and habitat use patterns of early post-settlement winter flounder in Mt. Hope Bay. Movement patterns of older juveniles have been examined in some detail over

short time scales; however, seasonal movements are less well studied.

Environmental triggers for these movements are currently unknown.

5) Estuarine growth: Estuarine growth of juvenile winter flounder has been relatively well studied (see reviews in Able and Fahay 1998 and Pereira et al. 1999). However, quantification of growth and mortality rates of juveniles specific to Mt. Hope Bay habitats would be helpful. In particular, the effect of nutrient enrichment on growth is unknown. The impact of predation by summer flounder, searobins, bluefish and other piscivores during this phase require closer examination.

6) Estuarine emigration: Juvenile winter flounder do not always remain resident within the estuaries, but in fact a large component can migrate into coastal marine waters by January of the second winter (age-1; Able and Fahay 1998). Because there is evidence (Collie and DeLong 2001) of an increase in the mortality of winter flounder in Mt. Hope Bay at this age, and in greater Narragansett Bay at age-1 to age-2, a closer look at juvenile movements is warranted. Quantification of the percent of resident versus migrating individuals at each life stage is needed. A comparison of growth and mortality rates for coastal and estuarine juveniles may provide an important indication of sources of population changes for the species. Again, as with other migratory stages, identification of important environmental and biotic triggers for migration is also needed.

7) Coastal residence/growth: Growth and movements of juveniles within the coastal waters should be compared to those in estuarine waters.

Habitat use patterns are unknown. Is there a difference in mortality and contribution to adult stocks from juveniles that migrate from the estuary at different times versus those that remain resident in the estuary until the onset of maturity?