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Cynthia M. Jones

Old Dominion University, [cjones@odu.edu](mailto:cjones@odu.edu)

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## Contribution to the Themed Section: 'The Value of Coastal Habitats for Exploited Species'

# Can we predict the future: juvenile finfish and their seagrass nurseries in the Chesapeake Bay

Cynthia M. Jones\*

Center for Quantitative Fisheries Ecology, Old Dominion University, 800 W 46th St, Norfolk, VA, USA

\*Corresponding author: tel: +757 683 4497; fax: 757 683 5293; e-mail: [cjones@odu.edu](mailto:cjones@odu.edu)

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The importance of estuarine seagrass beds as nurseries for juvenile fish has become a universal paradigm, especially for estuaries that are as important as the Chesapeake Bay. Yet, scientific tests of this hypothesis were equivocal depending on species, location, and metrics. Moreover, seagrasses themselves are under threat and one-third of seagrasses have disappeared worldwide with 65% of their losses occurring in estuaries. Although there have been extensive studies of seagrasses in the Chesapeake Bay, surprisingly few studies have quantified the relationship between seagrass as nurseries for finfish in the Bay. Of the few studies that have directly evaluated the use of seagrass nurseries, most have concentrated on single species or were of short duration. Few landscape-level or long-term studies have examined this relationship in the Bay or explored the potential effect of climate change. This review paper summarizes the seagrass habitat value as nurseries and presents recent juvenile fish studies that address the dearth of research at the long term and landscape level with an emphasis on the Chesapeake Bay. An important conclusion upon the review of these studies is that predicting the effects of climate change on fishery production remains uncertain.

**Keywords:** Chesapeake Bay, climate change, juvenile fish, seagrass nurseries.

### Status of seagrass nurseries in the Chesapeake Bay

The importance of estuarine seagrass beds as nurseries for juvenile fish worldwide was highlighted by Beck *et al.* (2001). They noted that these habitats have extremely high primary and secondary productivity. Moreover, these are among the most productive habitats in the world (Larkum *et al.*, 2006) and are especially important as nurseries for commercially and recreationally exploited fish and shellfish (Duarte, 2002; Schmidt *et al.*, 2011). In the Chesapeake Bay, many commercial and recreational fisheries species use these beds as nursery habitat (Olney and Boehlert, 1988; Able and Fahay, 1998; Murdy *et al.*, 2002) that provides a rich source of invertebrate prey (Elliot *et al.*, 2002), refuge from predation (Connell, 1975), and improved water quality (Gruber *et al.*, 2011).

However, seagrasses themselves are under threat and show many instances of localized extinction globally (Lotze *et al.*, 2006). One-third of seagrasses have disappeared worldwide (Waycott *et al.*, 2009), with 65% of their losses occurring in estuaries (Lotze *et al.*, 2006). A historical prospective of seagrass loss in the

Chesapeake Bay is illustrative. The Chesapeake Bay is the largest estuary in the United States with a watershed of 165 800 km<sup>2</sup>, and the third largest in the world. The Bay is ~10 000 years old and its watershed covers six states (Curtin, 2001). Before European settlement, the watershed was forested, in some place up to the water's edge (Brush, 2001), with extensive seagrass meadows and oyster reefs as noted in historical records (Orth and Moore, 1984). Subsequently, land was cleared for agriculture with 80% of the region cleared of forests in the 1800s. Since then, extensive reforestation has occurred as farms were abandoned. With the initial land clearing, sedimentation increased beginning the decline in seagrasses. Historically, seagrasses were widespread in shallow water habitats, but also with eelgrass (*Zostera marina*) undergoing periodic changes in abundance (Orth and Moore, 1984).

Among the vulnerable seagrass species that support fisheries productivity, eelgrass (*Z. marina*) has been the dominant submerged macrophyte in the mesohaline and polyhaline portions of the Bay (Orth and Moore, 1984; Moore *et al.*, 1996). In 1930,

eelgrass populations experienced a precipitous decline likely due to disease (Heck *et al.*, 2003). A further major decline occurred in 1972 when Hurricane Agnes (one of the largest June hurricanes on record) impacted the Bay. The Bay lost significant portions of its seagrass beds due to increased sediment mobility and freshwater inflows that lasted weeks and occurred during the height of eelgrass growth and reproduction. The loss of seagrass meadows was restricted to the Bay and not seen regionally (Orth and Moore, 1983). Subsequently, there was an increase in seagrasses that was largely attributable to a 71% increase in *Zostera* in the polyhaline portions of the Bay (Moore *et al.*, 2000). Eelgrass acreage has not appreciably rebounded, since the late 1990s despite replanting efforts (Orth *et al.*, 2010). Re-establishing seagrasses is often hindered by increased turbidity is one of the most important factors deterring seagrass restoration (Dennison and Alberte, 1985). Moore *et al.* (1996) determined that decreased light penetration caused by suspended sediments and phytoplankton blooms directly resulted in lost habitat for seagrass in the Chesapeake Bay.

Although there have been extensive studies of seagrasses in the Chesapeake Bay; surprisingly much less study has been directed to the relation between seagrasses as nurseries for finfish in the Bay. The objectives of this review are to: (i) summarize the value of seagrasses as fish nurseries worldwide, (ii) summarize knowledge specific to the Chesapeake Bay, and (iii) demonstrate the value of large-scale synoptic and process-oriented studies to evaluating potential impacts of climate change to the Bay.

### Seagrass nurseries and fisheries

Beck *et al.* (2001) provide a definition of a “nursery” as: “Generally, an area has been called a nursery if a juvenile fish or invertebrate species occurs at higher densities, avoids predation more successfully, or grows faster there than in a different habitat”. The literature largely defines a nursery as an area of high concentrations of juveniles or larvae (Jackson *et al.*, 2001). An important consideration is whether seagrass beds confer any survival advantage to juveniles subsequently as they migrate to their adult habitat (Heck *et al.*, 2003; Gillanders, 2006).

One way to estimate the value of seagrass nurseries is to compare the productivity of areas with and without seagrasses. However, the impact of seagrass losses on fisheries production is mixed depending on species and location (Jackson *et al.*, 2001). Orth *et al.* (2006) noted that the widespread decline in seagrasses in the western North Atlantic during the 1930’s was associated with a collapse in the scallop fishery and decline in the abundance of waterfowl. However, the eelgrass die off also occurred in the eastern North Atlantic Ocean and fisheries in there did not collapse (Heck *et al.*, 2003). These authors argue that alternative nursery habitats were available, such as those provided by macroalgae and rocky bottom, which subsequently filled the role of eelgrass and therefore fisheries did not collapse. When species can use alternative nurseries, effect of seagrass loss is ameliorated. However, alternative habitats may not be available, and for some species, juveniles may not survive as well in the alternative habitat. One alarming statistic is that 91% of species associated with seagrass worldwide were depleted because of eutrophication, exploitation, and decline in habitat quality (Lotze *et al.*, 2006). However, the relationship between seagrass beds and fish productivity is complex and may be mediated by the availability of alternate habitat. In a changing world, the fate of estuarine seagrass beds will undoubtedly impact the survival and recruitment of young fish, but in ways that are yet unknown (Najjar *et al.*, 2010). Nonetheless, knowledge addressing

the use of habitats and their effect on growth and survival may help us predict the impacts of climate change more accurately.

The use of seagrass habitats worldwide by important juvenile commercial fish and shellfish has been studied extensively. After conducting a major review of the published literature that evaluated seagrass habitats, Heck *et al.* (2003) noted that fish growth, abundance, and survival were greater in seagrass than in unstructured habitats. Moreover, fish and crabs are far more abundant in seagrass than in adjacent unvegetated areas (Thorhaug and Roessler, 1977; Orth and van Monfrans, 1987; Hughes *et al.*, 2002), although greater abundance is not universal for all species (Jackson *et al.*, 2001; Heck *et al.*, 2003). However, when contrasting among structured habitats, Gillanders (2006) noted that seagrass may not be more productive than other structured habitat; that structure itself provides the advantage. Other studies that evaluated seagrass beds and their faunal assemblages concluded that heterogeneous beds, with bare areas were interspersed with vegetated areas, provided the most favourable nursery habitats for mobile foraging juvenile fish (Orth *et al.*, 1984). Additional studies demonstrated increased juvenile growth because of more abundant food and lower predation in seagrass nurseries than in other habitats, although survival was more difficult to quantify (but see Koenig and Coleman, 1998). The majority of studies reviewed by Heck *et al.* (2003) and Gillanders (2006) measured density or abundance, fewer measured growth, only a handful measured survival, and none quantified the success of juveniles to migrate to their adult habitat (Gillanders *et al.*, 2003) or to return to natal habitats for spawning (Thorold *et al.*, 2001). Gillanders (2006) pointed out the dearth of landscape-level studies evaluating the importance of seagrasses to fish production.

### Studies of juvenile fish using seagrass as nurseries in the Chesapeake Bay

Surprisingly few studies have looked at the value of the Chesapeake Bay seagrass nurseries to the abundance, growth, or survival of juvenile finfish either synoptically or long term. Most published studies assess the impacts of abiotic and biotic factors on seagrass distribution (Moore *et al.*, 1996, 2000), interactions with lower trophic levels (Fredette *et al.*, 1990; Cordero *et al.*, 2012), and commercially exploited blue crab (*Callinectes sapidus*; Lipcius *et al.*, 2005; Seitz *et al.*, 2005). For example, although Grubbs and Musick (2007) determined the impact of salinity and oxygen concentrations on the habitat use of sandbar shark juveniles in the lower Bay, these studies were not specific to seagrass beds. Ontogenetic changes in the diet of summer flounder (*Paralichthys dentatus*) were conducted in another study on fish caught in trawls from the ChesMMAP Survey and VIMS Juvenile Finfish and Blue Crab Trawl Survey (Buchheister and Latour, 2011), which unfortunately are not conducted in the seagrass beds, although seagrass beds are an important habitat for summer flounder juveniles. Their study proposes similar interactions in seagrass nurseries that they found from fish in the trawl surveys, but provides no direct observation or confirmation for fish in seagrass.

Of the few studies that have directly evaluated the use of seagrass nurseries in the Chesapeake Bay by juvenile fish, most have concentrated on single species or were of short duration. Several compared the distribution of juvenile fish specifically in vegetated and unvegetated habitats (Orth and Heck, 1980; Heck *et al.*, 2003). Juvenile fish of harvested species, such as spot (*Leiostomus xanthurus*), and spotted seatrout (*Cynoscion nebulosus*) are found in far greater

abundance on seagrass beds that in unvegetated areas. Orth and Heck (1980) showed that juvenile fish were present in up to 2 orders of magnitude greater abundance in vegetated areas (eelgrass and widgeon grass, *Ruppia maritima*) in the Bay than in unvegetated areas. They found that predators captured near the seagrass beds, such as *Cynoscion* species and bluefish (*Pomatomus saltatrix*), fed on juvenile fish that take refuge in the seagrass beds. They also found a seasonal pattern with juvenile fish most abundant in summer and at minimally abundant in winter after the die-off of seagrasses.

### Physical properties of the seagrass nurseries in the Chesapeake Bay

Climate change will alter the physical properties of Chesapeake Bay waters, which in turn will alter the extent and possibly even existence of *Zostera* seagrass nurseries. Studies show that temperatures on the seagrass beds in the lower Bay are influenced strongly by air temperatures, follow a seasonal pattern, and vary little across the Bay (Dorval *et al.*, 2005b). These habitats are shallow and solar heating determines their temperature, which is consistent across the Bay. Thus, as temperatures rise by at least 5°C (Najjar *et al.*, 2010), the rise will affect all shallow-water beds alike. Unlike temperature, salinities differ across the Bay, with the eastern shore beds experiencing greater marine influence than the western shore due to Coriolis effects and riverine input to the western shore. These differences are also reflected in the chemistry of seagrass sediments (Hannigan *et al.*, 2010). Moreover, the chemistry of waters overlying the seagrass beds reflect their exposure to riverine or marine input, sediment resuspension and to the degree of tidal mixing that has occurred (Dorval *et al.*, 2005a). Although these habitats are usually well oxygenated, Breitburg *et al.* (2009, 2011) have found that transient anoxic events can occur at night, even though the beds are shallow and waters well-mixed and this will stress resident organisms. The resilience, or lack thereof, of seagrasses and juveniles that use them as nurseries portends their response to climate change.

### Relation between physical properties and juvenile finfish habitat use

To understand and anticipate how juvenile fish will respond to climate change, we refer to studies that capture how juvenile fish select habitats to optimize growth and survival. For example, mesocosm studies have shown that pinfish (*Lagodon rhomboides*) and Atlantic croaker (*Micropogonias undulatus*)—juveniles that are seen in Chesapeake Bay seagrass nurseries, choose habitats based dissolved oxygen, prey availability, and predation (Froeschke and Stunz, 2012). Habitat selection was structured predominantly by predation risk, with juveniles choosing low suitability habitat in the face of heavy predation. An advantage of seagrass beds is that they afford predator protection while meeting other habitat needs. However, *in situ* studies to ascertain predation risk in juvenile fish have not been done in the Bay's seagrasses.

Juveniles of commercially and recreationally harvested species use seagrass nurseries seasonally, occurring in greatest abundance from late spring through fall (Chesapeake Executive Council, 1990; Dorval *et al.*, 2005b). Although there have been studies of juvenile finfish and their community structure, these are limited in geographic scale (Orth and Heck, 1980; Heck and Thoman, 1981; Sobocinski *et al.*, 2013). Sobocinski *et al.* (2013) sampled over two nearby areas on the western shore from 2009 to 2011 and compared species richness, community composition, and density with a

parallel study done from 1976 to 1977. They observe similar seagrass coverage over time, but changes in community structure that they attribute to temperature rise.

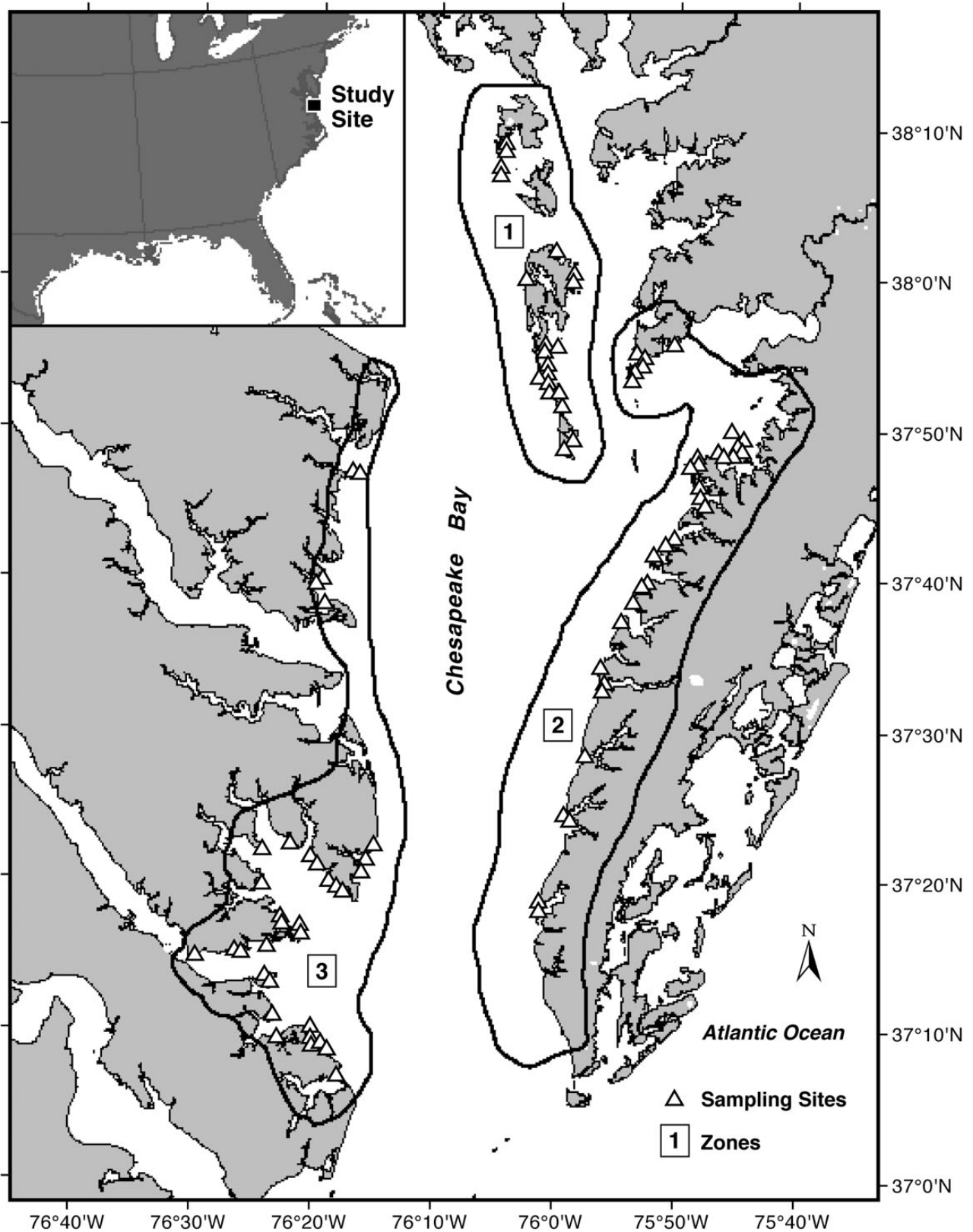
Only one recent study has evaluated the community structure of juveniles and its relation to biotic and abiotic factors synoptically throughout the seagrass beds in the mesohaline and polyhaline portions of the Bay (Schaffler *et al.*, 2013; Figure 1). That study sampled extensively in the seagrass beds across the Bay during summers from 1997 to 1999. They sampled over 20 000 juveniles of 51 species and tested the association between physical factors and the presence of species in three major seagrass habitats across the 3 years. Salinity was the most influential factor in guiding habitat choice, whereas seagrass species was not. Species associated with lower salinity seagrass habitats were weakfish (*Cynoscion regalis*), and Atlantic croaker (*M. undulatus*) for example, whereas spot (*L. xanthurus*) was strongly associated with higher salinity and temperature. Moreover spot, which has been the most abundant in historical studies (Orth and Heck, 1980), are no longer the dominant species, confirming the study by Sobocinski *et al.* (2013). It is difficult to attribute this solely to climate change because the change in abundance could be due to potential changes in spawning-stock biomass related to fishing. Because of the strong community structuring with salinity, Schaffler *et al.* (2013) note that climate related changes in precipitation (amount and rate) could lead to changes in the value of the seagrass habitats to juveniles of important finfish species.

### Case study of spotted seatrout (*C. nebulosus*) and seagrass nurseries in the Chesapeake Bay

One of the longest studies of the relationship between a species, spotted seatrout (*C. nebulosus*), and seagrass nurseries and spanned 1997–2008. Late larvae and juveniles are obligate residents of seagrass meadows across the mesohaline and polyhaline portions of the Bay (Dorval, 2003; Dorval *et al.*, 2005b). In accordance with recommendations laid in the Beck *et al.* (2001) review, the relative abundance, growth, and survival of spotted seatrout was evaluated in a series of studies. Moreover, these investigations were done at broad scale in one of the few landscape-level studies of finfish-seagrass nurseries in the Bay.

Spotted seatrout are found broadly distributed along the Atlantic East Coast and US Gulf of Mexico (Bortone, 2003) and Chesapeake Bay population is at the northern edge of the species' breeding range. Note that spotted seatrout are a separate genetic population in the Chesapeake Bay (Wiley and Chapman, 2003) and is the only population that undertakes inshore to offshore migrations in fall. Throughout its life, it is entirely estuarine dependent, has high site fidelity (Bortone, 2003), and "has the potential to serve as an important estuarine biological sentinel and monitor" (Bortone, 2003; preface). As such, it is an excellent species to portend the impact of climate change on the relationship between juvenile fish and seagrass nurseries. Elsewhere along its range, spotted seatrout uses several different types of structured habitat as nurseries. For example, in Louisiana, it is found in a variety of marsh habitats (Baltz *et al.*, 2003). However, in the Chesapeake Bay, juveniles are found tightly associated with seagrass nurseries and only rarely in other habitats, which makes them more vulnerable to seagrass loss in the Bay. To test the fidelity of juveniles to their settlement seagrass beds, Dorval *et al.* (2007) used the trace-element chemistry of otoliths (earstones that record age and growth; Jones, 2002) to demonstrate that juvenile spotted seatrout

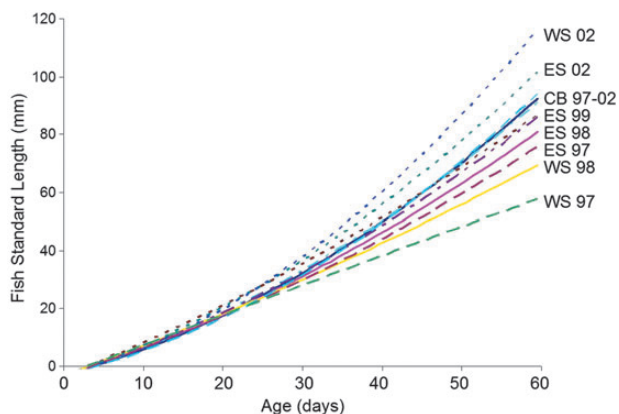




**Figure 1.** Map of the lower Chesapeake Bay with seagrass zones and a typical array of sampling stations indicated. These sampling zones have been consistently sampled from 1997 to 2008 in several studies investigating the value of seagrass nurseries to juvenile finfish.

settle into a seagrass nursery and only leave that specific habitat to migrate from the Bay in autumn as temperatures decline. Otolith chemistry showed near 100% correct classification to sampled habitat. Had juveniles moved between habitats, site-specific differences in their otolith chemistry would not have been observed. Dorval *et al.* (2007) demonstrated that juveniles incorporate into their otoliths the surface water chemistry of the habitat in which they live on a sufficiently fine scale that the juvenile's habitat use can be reconstructed in a defined chronology and their movement tracked thereafter (Dorval *et al.*, 2007; Hoover *et al.*, 2012). This specific habitat chemical signature was used to show that juvenile spotted seatrout growth, survival, and subsequent migration were tightly linked to the conditions in a specific seagrass meadow.

In synoptic, full season, and multiyear sampling, the habitat-specific growth of juvenile spotted seatrout was linked to regional weather events. Retrospective individual growth (Jones, 2000) of juveniles was determined between 1997–1999 and 2002 and these growth rates were compared across areas (Figure 1; same areas as used in Schaffler *et al.*, 2013). In normal years, fish grew fastest on the seagrass beds in the mid-Bay and slowest on the western shore (Smith *et al.*, 2008; Figure 2). However, in 2002 (a drought year), this growth pattern reversed and juveniles on the western shore grew fastest. Temperature and salinity drive these responses, as shown in other research for spotted seatrout in Texas (Holt and Holt, 2003). Typically, temperature is a strong determinant of growth, but this factor could be dismissed as causing the differences in growth in the Bay, because temperature in the seagrass beds across the Bay followed the same seasonal curve based on data from *in situ* recorders (Dorval *et al.*, 2005a). Moreover, because otolith microstructure records the age of juveniles in days, and because hatch-date distributions were similar, they dismissed temperature degree-day exposure as a cause of differences in growth resulting from variation in timing of post-larval settlement. The abiotic factor that differed dramatically between years was the salinity. Although salinity cannot be isolated as the sole causal agent in influencing growth—prey availability may also be altered. Holt and Holt (2003) have shown that spotted seatrout adapt well to a wide range of salinities in Texas estuaries, but that very low salinities impede the survival of early life stages. Anecdotally in 2003 (one of the wettest years in record), the freshwater inputs from Hurricane Isabel resulted in a complete loss of juvenile spotted seatrout from the Bay's seagrass



**Figure 2.** Growth rates of juvenile fish captured on seagrass beds on the eastern (ES) and western (WS) shores and the central Bay islands (CB) from 1997 to 2002. Taken from Smith *et al.* (2008).

beds subsequently. Clearly, more study is needed to determine if juveniles respond similarly at the northern edge of their range as they do in Texas estuaries. Taken together, these results suggest that altered precipitation patterns from storms or climatic changes affect not only the seagrass viability but also the potential fitness of juvenile fish closely tied to the beds, because faster growth is often reflected in better survival (Miller *et al.*, 1988).

To anticipate the impacts of changing climate on fisheries production, we need to be able to predict the survival of finfish using nursery habitats and measure their contribution to future reproduction. In work being readied for publication, S. K. Beharry and C. M. Jones (unpublished data) have tested the hypothesis that spotted seatrout return to the Bay in spring are a random sample from the seagrass beds or whether different seagrass nursery areas confer any survival advantage. Note again that spotted seatrout in the Chesapeake Bay are the most northerly population along the US East Coast and are unusual in that they undergo a seasonal migration into nearshore coastal waters each fall before returning to the Bay to spawn the following spring beginning at age one. Beharry and Jones used otolith chemistry as a habitat-specific natural tag to measure the nursery-specific proportions of 1-year olds that return in spring to spawn. As a natural tag, the chemistry embedded in the juvenile portion of the otolith can be examined in adults to classify them back to their seagrass nurseries with high reliability (Dorval *et al.*, 2005b; Beharry, 2011). Although the abundance of juveniles across the seagrass nurseries is equal, preliminary analyses show disproportionate survival in fish that use western shore nurseries across all years. Hence, there are differences in the survival advantage that seagrass beds confer. Further research is needed to investigate the causes of such disparate survival. The spotted seatrout nursery habitats in the Chesapeake Bay confer different growth and survival benefits, and some of these responses respond in complex ways to the climate change drivers that are occurring in the Bay.

### The seagrass nurseries, fish productivity, and potential impacts from climate change

Effects of climate change include not only temperature and CO<sub>2</sub> increase, sea-level rise, and changes in precipitations, but other impacts, such as ocean circulation that may change larval transport and alteration of community structure (Harley *et al.*, 2006). The rate of change is unparalleled, exceeding the natural cycles of the past (Harley *et al.*, 2006; IPCC, 2007). Thus, we anticipate distributional shifts that depend on species thermal tolerance and ability to adapt (Wuenschel *et al.*, 2012). For example, as temperatures have increased in the Barents Sea there is an apparent increase in capelin (*Mallotus villosus*) and cod (*Gadus morhua*; Dalpadado *et al.*, 2012). Increased abundance of juvenile finfish was observed in the Gironde estuary (France) that were correlated with an 18-year increase in temperature and seawater intrusion, but differed by species (Pasquaud *et al.*, 2012). In a 32-year study in the Hudson River estuary (New York, USA), the larval stages of striped bass (*Morone saxatilis*) were positively related to high freshwater flows, whereas American shad (*Alosa sapidissima*) were negatively related to warmer temperatures (O'Connor *et al.*, 2012). A 47-year study of finfish in Narragansett Bay (Rhode Island, USA) was sufficiently long to document a 1.6°C temperature increase that shifted community structure resulting in a decline in cold-water species and an increase in warm-water species (Collie *et al.*, 2008). We expect similar trends in the Chesapeake Bay.

Studies that have addressed the effect of climate change on the Chesapeake Bay are in general agreement that seagrass habitats will be at risk (e.g. Scavia *et al.*, 2002; Najjar *et al.*, 2010). Orth *et al.* (2006) point out that seagrasses evolved in the habitats of high temperature and CO<sub>2</sub>. Experimental studies indicate that high CO<sub>2</sub> will increase eelgrass productivity under high light conditions but have little effect under low light (Palacios and Zimmerman, 2007). But anthropogenically driven changes may be occurring at such a rapid rate that seagrasses may not adapt quickly enough. The *Z. marina* beds that serve as important habitats for exploited fish species could be decimated (Najjar *et al.*, 2010). The Chesapeake Bay is near the southernmost habitat for this species and high temperatures have already resulted in massive summer die-offs (Moore and Jarvis, 2008). This temperate species may be eliminated from the Bay as temperatures continue to rise (Carr *et al.*, 2012). *Ruppia maritima*, an alternate habitat for juvenile finfish in the lower Bay, will likely persist but in shallower waters. However, the functional ecology of *R. maritima* habitat differs considerably from those of *Z. marina* because of structural differences between these two species, the more ephemeral nature of *R. maritima* habitats and the fact that *R. maritima* does not have strong sediment binding rhizome system (Kemp *et al.*, 2005; Moore and Jarvis, 2008; Hengst *et al.*, 2010). Moreover, in 2012, NOAA announced that surface waters were 6°C and bottom waters 5°C higher than average in the Chesapeake Bay region, posing increased risk of *Zostera* decline ([http://www.nefsc.noaa.gov/press\\_release/2012/SciSpot/SS1209/](http://www.nefsc.noaa.gov/press_release/2012/SciSpot/SS1209/)).

Seagrass habitats are also vulnerable to freshwater run-off events. These events mobilize sediments, nutrients, and pollution, increase turbidity through suspension of fine particles, and promote phytoplankton production (Scavia *et al.*, 2002; Kemp *et al.*, 2005; Najjar *et al.*, 2010). These studies project wetter springs and drier summers. However, predicting the future flow regimes is made more difficult because climate models are inconsistent in their precipitation forecasts for the mid-Atlantic region. In the face of this uncertainty, implementation of the Chesapeake Bay Programme's Agricultural Best Management practices in the watershed's farms would help minimize nutrient and sediment run-off. Nonetheless, increased run-off from urbanization remains a problem that has not been addressed and may negatively impact seagrass nurseries (Roberts *et al.*, 2009; Freeborn *et al.*, 2012).

Predictions of warmer Chesapeake Bay temperatures generally favour the increased production of subtropical fish species (Hughes *et al.*, 2008; Najjar *et al.*, 2010; NOAA, 2010). This should favour sciaenid species such as Atlantic croaker, spot, and spotted seatrout, and perhaps even lutjanids. The sciaenids use seagrass nurseries extensively in the subtropics (Rooper *et al.*, 1998) and are sensitive to low winter temperatures (Lankford and Targett, 2001). Using temperature forecasts and three potential CO<sub>2</sub> scenarios, a coupled climate-population model predicted a 60–100% increase in Atlantic croaker populations and a shift in the population range of 50–100 km northward (Hare *et al.*, 2010). Nonetheless, these species also select nursery habitats based on salinity, type of vegetation, and other as yet unknown factors. Smith *et al.* (2008) have shown that changes in salinity impact growth rates of juvenile spotted seatrout using these nurseries, with drier, higher salinity conditions promoting their growth. Thus, if summers were drier, spotted seatrout juveniles might grow faster. Characteristics of the specific seagrass bed profoundly influence the growth and survival of spotted seatrout juveniles (Dorval, 2003; Beharry, 2011) and yet this species' response to *Zostera* loss is not quantified.

Change is coming to the Bay in ways that may alter the relationship between seagrass and their function as nurseries for juvenile fish. The effects of climate change are interdependent, non-linear, and difficult to predict. Climate models offer uncertain predictions about precipitation in the mid-Atlantic region, with some predicting increased severity of droughts and flood (Meehl *et al.*, 2007), and with widely varying streamflow projections (Najjar *et al.*, 2010). In an analysis of long-term trends in the mid-Atlantic Bight, Austin (2002) found that temperatures and precipitation were correlated, resulting in a shift between dry cool periods and warm wet periods. When fish populations are subject to harvest concurrent with environmental variability, their resilience to change can be unpredictable (Petitgas *et al.*, 2013). This paper has shown that few long-term and landscape-level studies have been done, upon which to base these predictions and it argues that there is more uncertainty than has been recognized in making such predictions.

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