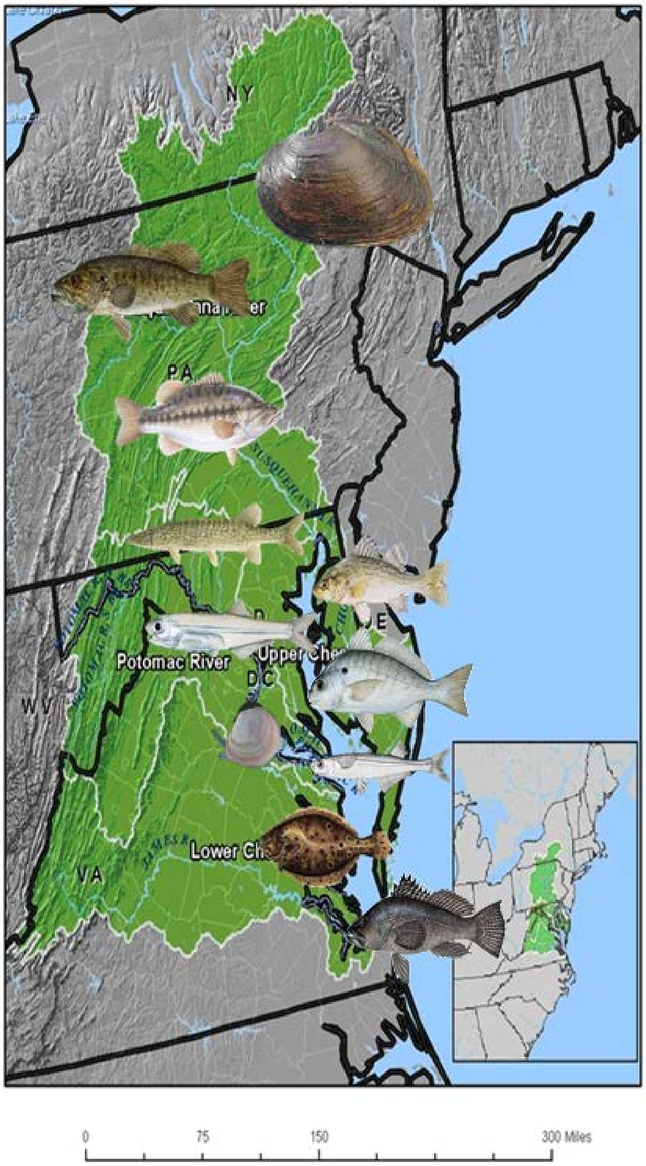
Chesapeake Bay Species Habitat Literature Review



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Bob Murphy **Sam Stribling**

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Purpose:

The Sustainable Fisheries Goal Implementation Team (Fisheries GIT) of the Chesapeake Bay Program was allocated Tetra Tech (Tt) time to support Management Strategies under the 2014 Chesapeake Bay Program (CBP) Agreement. The Fish Habitat Action Team under the Fisheries GIT requested Tt develop a detailed literature review for lesser-studied species across the Chesapeake Bay.

Fish and shellfish in the Chesapeake Bay and its watershed rely on a variety of important habitats throughout the watershed. These habitats, which are key to sustaining fisheries, are being threatened by a suite of stressors and sources such as increased urbanization, poor water quality and climate change. Successful fisheries management depends on knowing what and where these important habitats are and addressing the potential and realized threats to their integrity. Tetra Tech will provide staff time and resources to develop products in support of the Fish Habitat Outcome and Management Strategy (http://www.chesapeakebay.net/managementstrategies/strategy/fish_habitat). As a first step the Management Strategy calls for compiling information on key species, their habitat requirements, habitat functions (generally spawning, nursery, and forage), and the threats or stressors to these habitats.

Project Organization:

Adriana Berk – Work Assignment Leader
Bob Murphy – Fisheries Ecologist
Sam Stribling – Director of Ecological Sciences

Atlantic Silverside (*Menidia menidia*)

The Atlantic silverside is one of the most abundant fish species in Chesapeake Bay and along the mid-Atlantic coast (Able and Fahay 1998; Murdy et al. 1997) and is ubiquitous in surface waters along the coast (Murphy and Secor 2005).

Atlantic silversides tolerate a wide range of salinities (Fay et al. 1983) so salinity is not considered a critical habitat component. However, larval growth rate peaks at 30 ppt (Fay et al. 1983), though they will suffer little mortality at lower salinities. Larval Atlantic silversides favor a temperature range of 15^o-20^o C, though they are tolerant of a wider range of temperatures (Austin et al. 1975). Adults tolerate similar temperatures ranging from 5^o-30^o C, with a preference for water temperatures of 15^oC (Fay et al. 1983). Atlantic silverside larvae show a preferred dissolved oxygen level of 7.9 mg l^{-1} and exhibit decreased fitness when DO < 3.0 mg l^{-1} (DePasquale et al. 2015) Several studies indicate that adult Atlantic silversides prefer seagrass habitat and can therefore be considered critical habitat (Orth and Heck Jr. 1980; Schein et al. 2012) while it also shows affinity to sea lettuce (*Ulva* sp.) habitat in the northern portion of its range (Schein et al. 2012). There is evidence that a portion of Atlantic silversides seek refuge in the deeper waters of the Chesapeake Bay over the winter (Conover and Murawski 1982). Atlantic silversides are pelagic predators on mesozooplanktion and therefore require fairly clear water to find prey (Gilmurray and Daborn 1981) although no clarity criteria has been established. Recent research has confirmed that *Spartina alternaflora*, in association with *Enteromorpha* is the preferred spawning habitat (Balouskus and Targett 2012).

Silversides require fairly robust DO levels in the water column and would therefore be sensitive to any stressors increase the spatial and temporal scale of hypoxia (Batiuk et al. 2009; Breitburg 2002). Increased eutrophication is the primary cause of hypoxic conditions and represents a threat to Atlantic silverside habitats. Increased eutrophication coupled with increased sediment runoff will increase turbidity, and therefore potentially affect feeding success (Gilmurray and Daborn 1981). As the volume of coastal waters become hypoxic, there is less habitat available, and consequently fish are “squeezed” from the water column into less productive areas (Breitburg 2002; Campbell and Rice 2014; Eby et al. 2005). Atlantic silversides utilize seagrass meadows (for foraging and refuge); this habitat is also threatened by eutrophication and reductions in water clarity (Wazniak et al. 2007). In addition, *S. alternaflora* habitat (for spawning) is under direct threat from shoreline modification and development (Balouskus and Targett 2012; Bilkovic and Roggero 2008), thereby reducing reproductive success.

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Bay Anchovy (*Anchoa mitchilli*)

Bay anchovy is often considered the most abundant fish species found in Chesapeake Bay as well as the Atlantic Coast (Able and Fahay 1998; Jung and Houde 2003; Szedlmayer and Able 1996). Because of the high abundances of bay anchovy, they are frequently found in a wide variety of estuarine habitats (Szedlmayer and Able 1996), including oyster reefs (Peterson et al. 2003) and seagrass beds (Orth and Heck Jr. 1980).

Bay anchovy utilize a wide range of salinities during spawning, ranging from 0-32 ppt (North and Houde 2004; Olney 1983). Additionally, North & Houde (2004) demonstrated the importance of the pycnocline to larval survival, primarily because of the greater concentration of prey items at this frontal boundary, although this is not always the case if there is greater mixing due to wind events (Rilling and Houde 1999). Bay anchovy require greater than 3.0 mg l^{-1} dissolved oxygen at larval and adult stages (Houde and Zastrow 1991). Due to their ubiquity in Chesapeake Bay and Atlantic coast inner continental shelf waters (Woodland et al. 2012), structural habitat components (e.g. reefs, seagrass beds) are not as critical compared to water quality attributes that affect a greater proportion of available habitat for bay anchovy (Houde and Zastrow 1991). Adult bay anchovy tolerate a wide range of temperatures, from 5-30^{o} C while larvae are more restricted to optimal temperatures of 23-27^{o} C (Houde and Zastrow 1991).

The greatest threats to water quality, particularly dissolved oxygen minimum requirements, are stressors that reduce oxygen levels. Eutrophication is the major cause of hypoxia in coastal systems (Breitburg 2002) and has a significant influence on available habitat for bay anchovy (Lee and Jones 1991). As the volume of coastal waters become hypoxic, there is less habitat available, and consequently fish are “squeezed” from the water column into less productive areas (Breitburg 2002; Campbell and Rice 2014; Eby et al. 2005). In addition, hypoxia will potentially have negative effects on egg and larval survival, although due to the complicated structure of the Chesapeake Bay, this may be an oversimplification (Adamack et al. 2012) that is true in some instances (e.g. the Patuxent River vs the influence of the mainstem Bay on riverine systems). One mechanism by which hypoxia has shown to be deleterious on bay anchovy growth is thru displaced foraging opportunities on mesozooplankton due to hypoxia (Ludsin et al. 2009). Although structural habitats are less critical for bay anchovy growth and reproduction, these habitats are exploited by adults and therefore the population will be negatively affected by the loss of these habitat types (Sobocinski et al. 2013). In addition to eutrophication, the effects of climate change are likely to have negative effects on bay anchovy by warming coastal waters and thereby reducing amount of dissolved oxygen the water can hold (Roessig et al. 2004). Moreover, coastal temperatures are susceptible to warming above the optimal temperatures that bay anchovy are most adapted to, thereby imposing deleterious effects on fishes through physiological stress, reduced growth and fitness, and mortality (Roessig et al. 2004).

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Black Sea Bass (*Centropristis striata*)

Black sea bass occur along the Atlantic seaboard from Nova Scotia to southern Florida and into the Gulf of Mexico (Able et al. 1995). They are primarily a marine species that occasionally inhabit the middle and lower Chesapeake Bay, remaining in polyhaline waters (Murdy et al. 1997).

Black sea bass spawn along the Atlantic continental shelf where eggs are retained deep in the water column at depths of 10-375m (Drohan et al. 2007). Larval growth peaks 22^{o} C (Berlinsky et al. 2000) as they drift along the continental shelf waters. Juveniles frequently enter estuaries where they are strongly associated with structural complexity, whether it be from seagrass (Weinstein and Brooks 1983), wrecks (Stephan and Lindquist 1989) or oyster reefs (Arve 1960; Lehnert and Allen 2002). As a matter of fact, the decline in black sea bass stocks in Maryland coastal lagoons was attributed to oyster reef removal by the 1950's (Arve 1960; Coen et al. 1999). Adult sea bass rely on structure similar to the juveniles, though they are most frequently found at greater depths than juveniles (Drohan et al. 2007). Typically in Chesapeake Bay, oyster reefs are restricted to the shallower waters < 10m (McCormick-Ray 2005).

Threats (stressors) to black sea bass critical habitats (seagrasses, reefs, deep water) are many and varied, with respect to spatial and temporal variability. Seagrasses, as photosynthetic organisms, are susceptible to decreased light availability associated with poor water quality stemming from increased algal blooms (stimulated by eutrophication) and suspended solids (runoff from upland erosion) (Batiuk et al. 2000; Dennison 1987). In addition to the effects of eutrophication on seagrass habitat, increased nutrient loads stimulate algal blooms, ultimately resulting in areas with decreased dissolved oxygen. These areas have been shown to "squeeze" water column habitat typically occupied by black sea bass and forcing them into less productive habitats or to avoid estuarine systems altogether. (Campbell and Rice 2014; Eby et al. 2005; Froeschke and Stunz 2012). Oyster reefs have been decimated over the past century (McCormick-Ray 1998), primarily due to overfishing, disease (Dermo and MSX), and siltation (Smith et al. 2005; Woods et al. 2005). Therefore, valuable foraging habitat is lost when oyster reefs cannot be restored or maintained (Coen et al. 2007).

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Chain pickerel (*Esox niger)*

Chain pickerel is a common resident in all tributaries, tidal and non-tidal, of the Chesapeake Bay. It is most often found in freshwater, but can tolerate salinities up to 22 ppt (Murdy et al. 1997) and in some systems may prefer oligohaline conditions (Meyers and Muncy 1962).

Larval and juvenile pickerel utilize grass beds where they attach themselves, presumably for protection against predators (Underhill 1949). Adult chain pickerel are frequently found in submerged aquatic vegetation (SAV) beds (Armbruster 1959; Murdy et al. 1997) in slow moving water, typical of reservoirs, ponds, and tidal embayments (Moring and Nicholson 1994; Murdy et al. 1997; Underhill 1949). Although vegetation appears to be the primary habitat type, structure of any sort, including snags and woody debris, contribute to successful feeding in chain pickerel (Benke et al. 1985; Jenkins and Burkhead 1994). In addition, chain pickerel tend to prefer shallow water < 3m (Scott and Crossman 1973). Spawning takes place in water temperatures of 2-22^{o} C although adults can tolerate temperatures up to 30^{o} C (Coffie 1998). Primarily a freshwater species, chain pickerel are known to move into brackish waters with salinities < 5 ppt (Jenkins and Burkhead 1994). Little information is available on the oxygen requirements of chain pickerel, but it can be inferred from its aggressive predatory nature that minimum oxygen requirements of 5 mg l^{-1} are required (Batiuk et al. 2009).

Since structure, particularly aquatic vegetation, is the critical habitat component for chain pickerel, threats and stressors that affect the abundance of SAV are the most important consideration for management of this species. As photosynthetic organisms, aquatic vegetation is susceptible to poor water clarity conditions, particularly those resulting from eutrophication and turbidity arising from sediment runoff (Batiuk et al. 2000; Dennison 1987). Although specific oxygen requirements from chain pickerel were not found, decreases in dissolved oxygen are likely to occur coincidental with eutrophication, whether in impoundments (Michaletz et al. 2012) or in coastal waters (Breitburg 2002).

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Eastern Elliptio (*Elliptio complanata* [Lightfoot, 1786])

(Bivalvia: Unionidae)

The Eastern elliptio (*Elliptio complanata* [Lightfoot, 1786]) is a widespread and common freshwater mussel distributed throughout most of eastern North America, from Georgia to New Brunswick, and westward through southern Quebec and Ontario to Minnesota (Cummings and Cordeiro 2011). That dominance is reflected in the Chesapeake Bay drainage, with its being the most frequently encountered mussel species during a statewide stream survey in Maryland (Ashton 2009; Harbold et al. 2014). The species is recognized as being a habitat generalist, requiring little more than a permanent waterbody, having been documented in large rivers, small streams to headwaters, canals, reservoirs, lakes, ponds, and wetlands with bottom materials ranging from clay, mud, sand, gravel, and cobble (Balfour and Smock 1995; Bogan and Proch 1997; Cummings and Cordeiro 2011; Strayer 1993). Although the characterization of *E. complanata* as a generalist is accurate, several researchers have found some affinity for different stream and watershed characteristics. Strayer (1993) and Ashton (2009) documented a somewhat higher frequency of occurrence in larger streams, draining >~300km^2, relative to smaller streams; further, Kat (1982) found reduced growth rates associated with soft, muddy stream bottoms, which he postulated as resulting from elevated energy requirements needed to maintain depth position and ingest nutritional resources. Studies of Eastern elliptio in lakes have demonstrated endobenthic versus epibenthic substrate positions to be seasonal, with the former more common during winter months (Amyot and Downing 1991), with similar behavior confirmed in lotic systems (Balfour and Smock 1995). Other non-*Elliptio* taxa of unionid bivalves have been evaluated for the relationship among thermal sensitivity, dissolved oxygen, and vertical migration, results indicating some capacity for withstanding the effects of drought or dewatering (Archambault et al. 2014; Hagg and Warren 2008). Strayer and Malcom (2012) investigated the effects on successful recruitment of crayfish predation, presence or absence of host fish (American eels), fine sediments, interstitial dissolved oxygen, and unionized ammonia; they found un-ionized ammonia to easily have the most distinct relationship, with 0.2 mg/L the apparent threshold. Additional threats to populations of freshwater mussels that may arise from nutrient enrichment include diseases and exposure to algal toxins. Examination of *E. complanata* collected from freshwater streams, and using standardized health assessments, Chittick et al. (2001) found gastrointestinal infections with anaerobic bacteria and trematodes. Histological evaluation of those individuals demonstrated atrophy of digestive glands and overall inflammation, which suggest a potential for diminished survival and reproduction. Harmful algal blooms and exposure of the Eastern elliptio to algal toxins was shown to compromise the immune system (Gelinas et al. 2014), which would also effect reductions in overall fitness. Glochidial access to fish hosts is critical to survival and reproduction of *E. complanata*, and so, presence/absence of suitable fish species is a limiting factor. Through highly focused and specific laboratory tests, Lellis et al. (2013) demonstrated suitability of five fish species as hosts for eastern elliptio glochidia, from mussels collected in the Susquehanna River (PA) and the Chester River (MD). Successful host infections were obtained with the American eel (*Anguilla rostrata*), Brook trout (*Salvelinus fontinalis*), Lake trout (*S. namaycush*), Slimy sculpin (*Cottus cognatus*), and Mottled sculpin (*C. bairdii*). Even minimal barriers such as lowhead structures prevent fish species from moving upstream, and thus, eliminating a critical portion of the mussel life cycle (Watters 1996).

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Eastern floater (*Pyganodon cataracta* [Say, 1817])
(Bivalvia: Unionidae)

(Synonyms: *Anodonta cataracta, A. dariensis, A. doliarus, A. marginata, and A. teres*)

The Eastern floater (*Pyganodon cataracta* [Say, 1817]) is common and broadly ranging throughout eastern North America, from Alabama, Georgia, and Florida (Apalachicola River basin), to the Lower St. Lawrence River basin (Burch 1973; Cummings and Cordeiro 2012; Strayer 1993; Strayer and Jirka 1997; Walsh et al. 2007). Ashton (2009) found it to be fairly widespread throughout the Maryland portion of the Chesapeake Bay drainage, but usually in low numbers. The species seems to prefer protected waters, quiet, low gradient, off the main channel, and has been observed in a variety of habitats including streams and rivers, marshes, lakes, and ponds (Bogan and Proch 1997; Strayer and Jirka 1997; Walsh et al. 2007). The Eastern floater is occasionally more common in lowland or piedmont streams (Ashton 2009; Strayer 1993), which are more typically low gradient and stable. Such streams would have a general tendency to have smaller-sized particles in suspension just above the more fixed substrate, with hydrologic disturbances potentially resulting in accelerated rates of erosion and increases in size dominance of suspended inorganic and organic particles. Tankersley and Dimock (1993) investigated the effect of brooding on water filtration and particle retention in the Eastern floater, demonstrating that adult individuals maintaining broods of glochidial larvae are less efficient at retaining particles <5µm in diameter, a size range that would be inclusive of many freshwater algae, including diatoms. Although the exact effects are unclear, the fact that there is such specificity in filtering efficiency suggests that substantial alterations in sizes of suspended sediment and algal particles could potentially lead to decreased fitness. Other research demonstrated that juveniles of *P. cataracta* are sensitive to anoxic conditions, as well as water having pH > ~4.5 (96h LC50), and temperature >33°C (96hr LT50) (Dimock and Wright 1993) Wright 1993). Many species of freshwater mussels rely on fish species for completion of their life cycles, and any factors compromising glochidial access to host fish are limiting survival and reproduction (Watters 1996). Same as for habitats, the Eastern floater is a host generalist, and fish species thought to be host include *Amploplites rupestris* (Rock bass), *Catostomus commersoni* (White sucker), *Cyprinus carpio* (Common carp), *Gasterosteus aculeatus* (Threespine stickleback), *Lepomis gibbosus* (Pumpkinseed), *Lepomis macrochirus* (Bluegill), *Perca flavescens* (Yellow perch) (NatureServe 2015; van Snik Gray et al. 1999).

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Largemouth Bass (*Micropterus salmoides*)

Largemouth bass are common inhabitants of lacustrine, riverine, stream, and brackish water habitats (Murdy et al. 1997; Stuber et al. 1982). As one of the most sought after game fishes in North America, there is substantial research into critical habitats at various life stages. The native range of largemouth bass included the Mississippi River drainage and the east coast from Florida to southeastern Virginia but the species has been introduced worldwide, including within the Chesapeake Bay watershed (Brown et al. 2009; Love 2011).

Largemouth bass's preferred habitat is lacustrine (Brown et al. 2009) and, although no natural geological lakes occur within the Chesapeake watershed, this habitat is found in reservoirs and impoundments throughout the region. The preference for sluggish water is evident in the Chesapeake drainage (Love 2011) as largemouth bass are typically encountered in greater numbers in larger riverine systems and sluggish backwaters (Murdy et al. 1997), while abundances are low to absent in first and second order streams (Stuber et al. 1982). Both juvenile (fry) and adult largemouth bass display a strong preference for vegetated environments (Brown et al. 2009; Stuber et al. 1982) and is thus considered a critical habitat for growth and feeding (Brown and Maceina 2002; Rintamaki 1986). Largemouth bass are tolerant of low salinities, although it appears that as salinities approach 10 ppt or greater, the energetic costs are substantial to coastal populations (Glover et al. 2012). On the other hand, populations close to or even within brackish water, can exhibit faster growth rates (Peer et al. 2006) which is partially at odds with some research (Love 2011), but may be consistent with the identification of resident oligohaline populations that remain in saline waters (Farmer et al. 2013; Lowe et al. 2009). Largemouth bass have fairly high oxygen demands, showing reduction of growth at 4 mg l^{-1} with an optimum dissolved oxygen concentration of 8 mg l^{-1} (Brown et al. 2009).

Given that critical habitat parameters are vegetation abundance, salinity, and oxygen levels, environmental stressors that affect these ecosystem attributes will have negative impact on largemouth bass growth and survival (Glover et al. 2012). Aquatic vegetation is light dependent, so eutrophication (which causes algal blooms) and runoff will cause die-offs of these plants (Orth et al. 2010). Additionally, removal via dredging or other means will also remove critical habitat for largemouth bass (Erftmeijer and Lewis III 2006). Salinity is a naturally fluctuating variable in estuarine systems, but not lotic ones. Therefore, any cause of rapid or prolonged salinity increases in typical largemouth bass habitat will reduce the overall habitat available. Major changes in precipitation due to climate change can have this effect (Walther et al. 2002). In addition, warming temperatures coupled with eutrophication, causes major reductions in dissolved oxygen. In deeper waters, this effect is more pronounced and can cause a major habitat squeeze for fish that utilize these areas (Breitburg 2002; Campbell and Rice 2014; Eby et al. 2005).

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Macoma (*Macoma balthica*)

Although multiple species of *Macoma* inhabit the Chesapeake Bay and its tributaries (Holland et al. 1977; Lippson et al. 1981), the greatest amount of literature and research has been conducted on the Baltic macoma, *Macoma bathica.*

Macoma is the dominant soft-bottom species in unvegetated habitats of Chesapeake Bay (Holland et al. 1977) and is often the dominant species in temperate to arboreal coastal environments (Powers et al. 2002). Its preferred habitat is relatively shallow (tidal or intertidal) soft, unvegetated bottom comprised of either sand or silt/muds (Hiddink 2003b; Lippson et al. 1981; Powers et al. 2002). Macoma clams are highly tolerant of hypoxic conditions (Babarro and De Zwaan 2008), although it exhibits behavioral changes when exposed to hypoxic conditions (Long et al. 2008) that may impact overall fitness of a population. Because of this, dissolved oxygen levels are best when > 3.0 mg l^{-1}, particularly for juveniles (Jansson et al. 2015). In addition, Jansson et al (2015) found that moderate hypoxia was even less of a stressor when pH was higher indicating a potential stress from decreasing pH associated with ocean acidification (Birchenough et al. 2015). Macoma is found over a wide range of salinities (Dauer et al. 1987; Lippson et al. 1981) ranging from 5-28 ppt. Although Macoma prefers mesohaline salinities, there is evidence thatabundances increase in lower salinity areas, likely as a physiological tradeoff to increase access to food resources (Seitz 2011).

The primary driver of ecosystem disruption in coastal systems is eutrophication (Nixon et al. 2001) and the various consequences (e.g. hypoxia). Although Macoma clams are tolerant of hypoxic conditions as shown by DeZwaan et al (2001), they do show a behavioral change of reducing their burrowing depth when exposed to low DO conditions (Long et al. 2008). This behavior modification, in turn, results in greater susceptibility of predation on the exposed siphons, thereby impacting the overall fitness of the population. Eutrophication can also promote disruptions in food web dynamics, leading to unbalanced populations between prey and predators (Philippart et al. 2007) In addition to eutrophication, climate change can have a significant impact on coastal systems, either from temperature increases and concomitant changes in phenology (Birchenough et al. 2015; Philippart et al. 2003), or through ocean acidification (Birchenough et al. 2015; Jansson et al. 2015). The primary effect of ocean acidification on $CaCO_2$-secreting bivalves is dissolution, or thinning of shells, which increases overall mortality. In addition to indirect stressors resulting from eutrophication and climate change, Macoma clams are also threatened by direct impacts to their habitats, either through coastal development (Seitz et al. 2006) or dredging (Hiddink 2003a).

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Potomac Sculpin (*Cottus girardi*)

As a surrogate species for:

Blue Ridge Sculpin (*Cottus caeruleomentum)*

The Blue Ridge sculpin is a recently described species of sculpin (Kinziger et al., 2000) found in several major basins within the Chesapeake Bay watershed. Due to the relatively recent description of the species, there is a paucity of published research on the Blue Ridge sculpin's ecological attributes, habitat requirements, and potential stressors. To develop meaningful management actions based on sound science, a literature review on its congener, the Potomac sculpin (*C. girardi*), was conducted as the ecology of the two species is thought to be similar.

In addition to tolerating both warm- and cold-water streams (Jenkins & Burkhead, 1994), the Potomac sculpin is tolerant of a wide variety of habitat types. It is found in the Appalachian and Piedmont streams draining into the Potomac River (WV, VA, MD) and from an isolated population in the James River (VA) drainage (Page & Burr, 1991). Preferred habitat appears to be rocky runs and pools of creeks and small to medium sized rivers (Page & Burr, 1991) though apparently low in abundance in the mainstem Potomac River (Jenkins & Burkhead, 1994). Potomac sculpin tend to favor stream reaches with a swift current rather than pools (Jenkins & Burkhead, 1994), so alterations to sediment loads or the intentional damming of streams can reduce the preferred habitat. Similarly, dredging of in-stream gravel deposits (typically found in swift current riffles) has a large impact on those species that inhabit this environment (Freedman & Stauffer, 2013). In addition, Potomac sculpin are known to frequent submerged vegetation habitat (Matheson & Brooks, 1983), so stressors that reduce vegetation abundance will likely have negative impacts on Potomac sculpin population resilience. Potomac sculpin are typically sedentary benthic fish that show low dispersal (Jenkins & Burkhead, 1994) distances. However, recent research (Hudy & Shiflet, 2009) demonstrated that a small contingent of the population will disperse greater distances (> 250m), likely a strategy to exploit potential resources and colonize (or escape) new areas (Secor, 1999). In this context, stressors to long-term population resilience would be increased habitat fragmentation and loss of habitat connectivity (Stoll et al., 2013).

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Selected Anodontine Species: Dwarf Wedgemussel (*Alasmidonta heterodon* [Lea]), Green Floater (*Lasmigona subviridis*), and Brook Floater (*Alasmidonta varicosa* [Lamarck 1819])

(Bivalvia: Unionidae)

These three species of anodontine mussels (Dwarf Wedgemussel (*Alasmidonta heterodon*), Green Floater (*Lasmigona subviridis*), and Brook Floater (*Alasmidonta varicosa*) were selected for a single review because, collectively, they: a) share many life history traits; b) are far more sensitive to a variety of environmental stressors than either *Elliptio complanata* or *Pyganodon cataracta*; and c) represent a broad geographic distribution throughout much of the nontidal Chesapeake Bay drainage, with at least two of them occurring in each state. All are generally found in Atlantic coastal streams and rivers from South Carolina to Nova Scotia and New Brunswick (Bogan and Proch 1997; Burch 1973; Clarke 1981; Swartz and Nedeau 2007), with *Lasmigona subviridis* (Green floater) somewhat more geographically restricted from North Carolina to New York (Strayer and Jirka 1997). Among the three species, only one live individual was found as part of a statewide stream survey in Maryland (Ashton 2009). These species are found only in lotic systems, never in lakes or reservoirs, and are rare in streams that are prone to scour. They can be found in rivers, but tend to prefer smaller freshwater streams that are mostly low gradient with consistent flows (Strayer 1993; Strayer and Ralley 1993; Swartz and Nedeau 2007). Several studies have shown that these anodontine mussels are more common in soft waters (low calcium) (Campbell 2014; Strayer 1993; Strayer and Ralley 1993; Swartz and Nedeau 2007). Specific habitat preferences for the Brook floater and the Dwarf wedgemussel include stream bottoms with a higher percentage of medium sand (grain size, 0.25-1.0mm) or smaller, water depth (mean 27.7cm, range 0.4-104 cm) and moderate/uniform current speed (mean 11.8 cm/s, range 0.0-65.0 cm/s) (Michaelson and Neves 1995; Strayer and Ralley 1993); both of these studies suggest that any hydrologic factors that cause sediment mobilization, such as increased flashiness, will be detrimental. Additionally, hosts for these species are poorly known, but any factors restricting glochidial access to fish host species threaten survival and reproduction (Watters 1996). An interesting, but apparently unexplained finding, was that there may be a negative association of mussels with macrophytes (Strayer and Ralley 1993). Although Strayer and Ralley (1993) found that particle size, depth, and current speed were important for mussel distribution but minimally useful for predicting their actual occurrence, they, along with Michaelson and Neves (1995) noted a strong affinity in the Dwarf wedgemussel and Brook floater for medium sand or smaller. Logically then, streams undergoing accelerated rates of bottom erosion will have diminished occurrence of sand, either in patches (as observed by Strayer and Ralley [1993] in the Neversink River, New York) or continuous, thus limiting the habitat value for these mussels. Strayer et al. (1996) suggest that potentially the most important threat to the survival and reproduction of mussel populations was their abundance and density, pointing out the findings of Downing et al. (1993) that fertilization may not occur unless males and females are in close proximity. The Strayer study was focused on *A. heterodon*, but the concerns are relevant for most mussels, primarily that techniques for field observation, data analysis and interpretation are so coarse that gaining an understanding of mussel microhabitats is often problematic.

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Smallmouth bass (*Micropterus dolomieu*)

Smallmouth bass inhabit tidal and non-tidal streams and rivers of the western shore of the Chesapeake Bay watershed (they are absent from the eastern shore tributaries) but are uncommon in tidal waters with salinities > 5 ppt (Murdy et al. 1997; Schmidt and Stillman 1998). In addition, they inhabit many of the reservoirs found throughout the watershed (Murdy et al. 1997).

Smallmouth bass tend to prefer large lakes and rivers and high order streams (Brown et al. 2009) that have cool water, cover, and relatively slow currents (Sechnick et al. 1986). These conditions are similar to those found in lakes and reservoirs (Turner and MacCrimmon 1970). Sechnick et al (1986) found that both juveniles and adult preferred currents < 10 cm/s, low light, and sufficient cover in the form of boulders, although logjams and rootwads were also found to be preferred habitat (Todd and Rabeni 1989). Depth appears to be a critical component for bass, particularly when it comes to nest-building (Orth and Newcomb 2002), although the authors stipulate that depth may be a surrogate as cover (boulders, rootwads) will co-vary with depth. Smallmouth bass tolerate a wide range of temperatures, with an optimum for adults being 21-27^{o} C, and 12.8-21^{o} C during spawning (Brown et al. 2009). Optimum values for smallmouth bass vary by life stage (Brown et al. 2009); normal activities (adult) require > 6 mg l^{-1}, spawning adults require > 7 mg l^{-1}, and larvae development requires > 6.5 mg l^{-1} (Davis 1975).

Threats to physical habitats preferred by small mouth bass (e.g. boulders, interstitial space, rootwads) are varied, but removal of structure (Helmus and Sass 2008) or destruction of nesting sites (Chu et al. 2006) will impair smallmouth bass populations. Chu et al (2006) modeled population dynamics of smallmouth bass in a lake system where nesting habitat and cover were manipulated and demonstrated that management activities should focus on protection of this habitat. Lotic systems do not tend to be as susceptible to hypoxic conditions, but larger riverine and reservoir systems can be negatively affected by low oxygen conditions associated with eutrophication (Jones and Hoyer 1982). In addition, although smallmouth bass tolerate a wide range of temperatures, climate change will likely have a negative impact on their populations as temperature would rise during the spawning period when smallmouth are more successful with lower temperatures (Brown et al. 2009; Walther et al. 2002). However, since small mouth are also tolerant of warmer temperatures (non-spawning activity), there is an indication that growth rates will increase with increasing water temperatures and may only be limited by prey availability (Pease and Paukert 2014).

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Spot (*Leiostomus xanthurus*)

Spot exploit various habitats depending on the point in their life history, whether post-larval, juvenile, or adult. These habitat associations have been elucidated through various studies conducted in the Chesapeake Bay in addition to similar coastal US habitats along the southeastern seaboard and the Gulf of Mexico.

Larval spot occur in pelagic continental shelf waters and enter estuaries as juveniles (Able and Fahay 1998). Juvenile spot are strongly associated with higher salinity and temperatures (Jones 2013). In addition, they are historically the dominant species associated with seagrass habitat (Orth and Heck Jr. 1980), although there seems to have been a shift in this dominance more recently (Sobocinski et al. 2013) indicating the decline in seagrass habitat may have had a density-dependent negative effect on spot populations in Chesapeake Bay. In addition to seagrass habitat use, young-of-year (YOY) juvenile spot are a dominant feature of subtidal salt marsh habitats (Able et al. 2007; Szedlmayer and Able 1996). Moreover, it should be noted that within geographic distribution of spot, the Chesapeake Bay appears to provide the optimum growth conditions for YOY (McCambridge Jr. and Alden III 1984). Adult spot utilize multiple habitats as they must remain motile to find food in mostly demersal habitats and adjacent to structure (Buchheister et al. 2013). The habitat value of oyster reefs to transient fishes such as adult spot is much more complicated than essential vs non-essential as evaluations of oyster reefs must consider reefs in the context of locally available habitat types. Spot will utilize reefs opportunistically as feeding habitat when surrounding habitat is unsuitable (Harding and Mann 2001).

Threats (stressors) to critical habitats (seagrasses, marshes, deep water) are many and varied, with respect to spatial and temporal variability. Seagrasses (eelgrass, in particular for spot), as photosynthetic organisms, are susceptible to decreased light availability associated with poor water quality stemming from increased algal blooms (stimulated by eutrophication) and suspended solids (runoff from upland erosion) (Batiuk et al. 2000; Dennison 1987). In addition to the effects of eutrophication on seagrass habitat, increased nutrient loads stimulate algal blooms, ultimately resulting in areas with decreased dissolved oxygen, which can be lethal to spot (Brady and Targett 2013; Pihl et al. 1991). These areas have been shown to “squeeze” water column habitat typically occupied by spot and forcing them into less productive habitats (Buchheister et al. 2013; Campbell and Rice 2014; Eby et al. 2005; Froeschke and Stunz 2012). Marsh habitat is threatened by a variety of factors, generally through destruction or erosion (Seitz et al. 2006; Zapfe and Rakocinski 2008). At the population level, climate change may have an even greater impact on spot habitat utilization. As coastal waters warm, increased temperatures are likely to reduce dissolved oxygen, leading to decreases in foraging, growth, and fecundity (Roessig et al. 2004). Climate change may also disrupt precipitation patterns, potentially increasing runoff and creating larger fluctuations of salinity in estuaries which spot cannot tolerate (Diaz and Onuf 1985).

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Summer flounder (*Paralichthys dentatus*)

Summer flounder (*Paralichthys dentatus*) begin their lives spawned offshore in the Atlantic Ocean, before moving into nearshore waters and estuaries as larvae and juveniles (Able and Fahay 1998). Summer flounder undergo a dramatic morphological transformation, accompanied by a complex habitat shift from the plankton to the benthos while leaving the ocean and entering the estuarine environment (Able et al. 1990). Summer flounder generally utilize demersal habitats in the higher salinity portions of Chesapeake Bay during all aspects of their life phases, though they may occasionally be found in oligohaline regions (Smith and Daiber 1977). Post-larval settlement occurs in estuarine waters (Able and Fahay 1998; Able et al. 1990), typically in either seagrass beds (Packer and Hoff 1999) or on sandy substrate and tidal flats (Burke et al. 1991). The key feature appears to be substrate particle size as summer flounder preferred sandy substrates to mud in laboratory experiments (Keefe and Able 1994). Juvenile summer flounder are known to occupy seagrass habitats in Chesapeake Bay where they can feed and take refuge from predation (Jones 2013). In addition, juvenile are also found in high densities in salt-marsh dominate tidal creeks (Able et al. 2007; Rountree and Able 2007), although it has not been well-studied in Chesapeake Bay. Adults prefer deeper water habitats (>6m) (Sackett et al. 2008) with DO levels >4.0mg l^{-1} (Buchheister et al. 2013), and temperatures <20 ^{o}C (Sackett et al. 2008). There is also evidence that adult summer flounder utilize regions adjacent to reefs, presumably for increased foraging opportunities (Henderson et al. 2014).

Abiotic factors, which comprise one component of a habitat, can affect growth rates of juvenile fishes (Necaise et al. 2005). Threats (stressors) to critical habitats (seagrasses, marshes, deep water) are many and varied, with respect to spatial and temporal variability. Additionally, abiotic factors such as DO, temperature and salinity that contribute to habitat suitability can interact with each other, making it difficult to parse out specific thresholds (Necaise et al. 2005). Seagrasses, as photosynthetic organisms, are susceptible to decreased light availability associated with poor water quality stemming from increased algal blooms (stimulated by eutrophication) and suspended solids (runoff from upland erosion) (Batiuk et al. 2000; Dennison 1987). In addition to the effects of eutrophication on seagrass habitat, increased nutrient loads stimulate algal blooms, ultimately resulting in areas with decreased dissolved oxygen. These areas have been shown to “squeeze” water column habitat typically occupied by summer flounder and forcing them into less productive habitats (Campbell and Rice 2014; Eby et al. 2005; Froeschke and Stunz 2012). Marsh habitat is threatened by a variety of factors, generally through destruction or erosion. At the population level, climate change may have an even greater impact on summer flounder habitat utilization. As coastal waters warm, increased temperatures are likely to reduce dissolved oxygen, leading to decreases in foraging, growth, and fecundity (Brady and Targett 2010; Roessig et al. 2004). Climate change may also disrupt precipitation patterns, potentially increasing runoff and creating larger fluctuations of salinity in estuaries which summer flounder avoid (Able et al. 1990).

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White Perch (*Morone americana*)

White perch are an abundant, year-round resident of Chesapeake Bay ranging from freshwater to full marine salinities along the entire gradient of the bay and its tributaries (Murdy et al. 1997). Considered a true estuarine species, white perch spend their entire lives within the estuary and are thus sensitive to impacts on Bay waters (Setzler-Hamilton 1991; Stanley and Danie 1983).

Larval white perch exhibit fastest growth at temperatures 15 – 20^{o}C in Chesapeake Bay (Marguiles 1988) with a minimum dissolved oxygen requirement > 5.0 mg l^{-1} (Setzler-Hamilton 1991). Juveniles require DO levels > 40% saturation (Hanks and Secor 2011) and prefer salinities 0-3 ppt (though can tolerate up to 30 ppt) (Setzler-Hamilton 1991; Stanley and Danie 1983). Juvenile white perch exhibit a complex pattern of habitat use (Kerr and Secor 2012; Kraus and Secor 2004; Kraus and Secor 2005b) whereby contingents of a river population may remain in the freshwater portion of the system, whereas another contingent may exhibit partial migration in higher salinity waters (Kraus and Secor 2004). For this reason, both the freshwater and brackish regions must be considered when managing habitat for juvenile white perch (Kraus and Secor 2005a). Adult white perch are commonly found in shallow waters in portions of the Bay with salinities less than 18 ppt (Murdy et al. 1997) during the summer where they utilize this area to feed (Stanley and Danie 1983), including submerged aquatic vegetation beds (Kraus and Jones 2012). It appears that white perch are selective in terms of using salt marsh areas for feeding as they have displayed site fidelity in these areas (Jones et al. 2014) and therefore these habitats can be considered critical. White perch also utilize oyster reefs for feeding opportunities, but are considered transient (as opposed to facultative or resident) users of reefs (Coen et al. 1999).

White perch are ubiquitous in estuarine waters, and although they are found in high abundances within SAV beds and oyster reefs, they do not appear to be obligate users of these habitats (Stanley and Danie 1983). Therefore, the most critical habitat requirement for all stages of white perch life history is adequate water quality, particularly temperature and oxygen levels. The major threats to these parameters are eutrophication (Breitburg 2002) and climate change (Roessig et al. 2004). Eutrophication results in hypoxic conditions in coastal waters, thereby reducing nursery habitat (Hanks and Secor 2011) and the amount of open water habitat available to adult white perch (Campbell and Rice 2014). The primary effect of climate change on Chesapeake Bay will be increasing water temperatures which will add metabolic strain on adults, and potentially create poor spawning conditions or mis-matched hatching times with food resources (Roessig et al. 2004). Climate change will also affect the hydrologic cycle, changing the frequency and intensity of precipitation events, thereby causing wider variation in salinity. Although white perch are adapted to a wide array of salinities, they do not acclimate to rapid salinity changes (Stanley and Danie 1983).

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