Finfish-Waterbird Trophic Interactions in Tidal Freshwater Tributaries of the Chesapeake Bay

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Abstract.—With the DDT ban enacted in the early 1970s, piscivorous bird populations have grown exponentially throughout the tidal reach of the Chesapeake Bay. However, avian population growth is not uniform throughout the Chesapeake Bay watershed; several species including Bald Eagles (*Haliaeetus leucocephalus*) and Ospreys (*Pandion haliaetus*) experienced significantly greater population growth rates in riverine tidal freshwater and oligohaline regions than in higher salinity portions of the bay. Shifting fish prey resources may provide an explanation for the observed influence of salinity on distribution of piscivorous bird populations. Changes in the fish resources available to avian predators over the past 40 years include changing temporal and spatial distribution of fish prey, as well as shifts in taxonomic and trophic structure of resident and migratory fish assemblages. Historical ecological changes, including long- and short-term changes in the abundance of anadromous clupeid fishes, Atlantic Menhaden (*Brevoortia tyrannus*), and the relatively recent introduction and establishment of non-indigenous fishes, within tidal freshwater rivers may be influencing piscivorous bird distributions and abundance, particularly for Bald Eagles and Ospreys, in the Chesapeake Bay. Predator-prey interactions among piscivorous birds and fish prey have received little attention from wildlife managers. Collaborative efforts between fishery scientists and avian ecologists will ultimately lead to better ecosystem management of the Bay's living resources.

Key words.—Chesapeake Bay, fresh tidal river, Bald Eagle, Osprey, fish-bird interactions, American Shad, Atlantic Menhaden, catfishes.

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After DDT was banned in the early 1970s, many piscivorous bird populations have grown exponentially throughout the tidal reach of the Chesapeake Bay (Watts and Byrd 1998; Watts *et al.* 2004; Watts and Byrd in press; Watts *et al.* in press). Several species experienced dramatic population declines prior to 1970 and have now recovered to near-historic levels. For example, after reaching a low of less than 60 breeding pairs in the early 1970s, the Bald Eagle (*Haliaeetus leucocephalus*) breeding population now likely exceeds 900 breeding pairs (Watts *et al.* in press). An estimated additional 1,500 to 2,000 eagles migrate north to spend the summer months within the Bay from breeding populations throughout the southeast, and during the late fall and early winter the Chesapeake Bay supports migrant Bald Eagles from the northeastern United States and Canada (Watts *et al.* 2007). Other species show similar population recoveries. In less than 30 years, Ospreys (*Pandion haliaetus*) increased from 1,400 pairs to 3,500 pairs (Watts *et al.* 2004), Great Blue Herons (*Ardea herodias*) increased from approximately 1,000 to more than 18,000 pairs, and Great Egrets (*Ardea alba*) increased from 1,400 to 3,600 pairs (Watts and Byrd 1998; Watts 2004; Watts and Byrd in press; D. Brinker, Maryland Department of Natural Resources, unpubl. data). However, avian population growth is not uniform throughout the Chesapeake Bay watershed; several species including Bald Eagles and Ospreys experienced significantly greater population growth rates in riverine tidal freshwater and oligohaline regions than in higher salinity portions of the bay (Watts *et al.* 2004; Watts *et al.* 2006).

Shifting fish prey resources may provide an explanation for the observed influence of salinity on distribution of piscivorous bird populations (Watts *et al.* 2004; Watts *et al.* 2006). As piscivorous bird populations rebounded in the Chesapeake Bay, *ca.* 1970-2006, coastal and riparian habitats were being transformed by activities such as shoreline development, over-harvesting of estuarine and riverine fisheries, and industrial and agricultural pollution. In addition, the relatively recent introduction and establishment of several non-native fishes within Chesapeake Bay tributaries may have significantly altered prey resources for avian predators (Edmonds 2003). The resultant changes in the fish resources available to avian predators over the past 40 years include changing temporal and spatial distribution of fish prey (Viverette 2004), as well as shifts in taxonomic and trophic structure of resident and migratory fish assemblages (CBV, unpubl. data). In this paper we will discuss how historical ecological changes, including long- and short-term changes in the abundance of anadromous clupeid fishes (Foerster and Reagan 1976), Atlantic Menhaden (Uphoff 2003a, b), and the relatively recent introduction and establishment of non-indigenous fishes within tidal freshwater rivers (McAvoy *et al.* 2000; Edmonds 2003) may be influencing piscivorous bird distributions and abundance, particularly for the Bald Eagle and Osprey, in the Chesapeake Bay.

FISHERY RESOURCES

Tidal Freshwater Fish Assemblage

Located between non-tidal freshwater and estuarine ecosystems, tidal freshwater habitats support a unique and diverse assemblage of estuarine, marine, and freshwater fish species (Wagner and Austin 1999). The resulting fish community is not only taxonomically diverse compared to adjacent non-tidal and estuarine habitats (Fig. 1), but also more temporally dynamic than adjacent aquatic systems (Viverette 2004) because many of the fish species are transitory and only inhabit tidal freshwaters during specific seasons or life-stages (Setzler-Hamilton 1987; Peterson and Ross 1991; Garman and Macko 1998; Yozzo and Smith 1998). Among the seasonal inhabitants of tidal freshwaters are the anadromous (migratory) clupeids—marine planktivores that migrate into freshwaters every spring to spawn. Anadromous clupeids

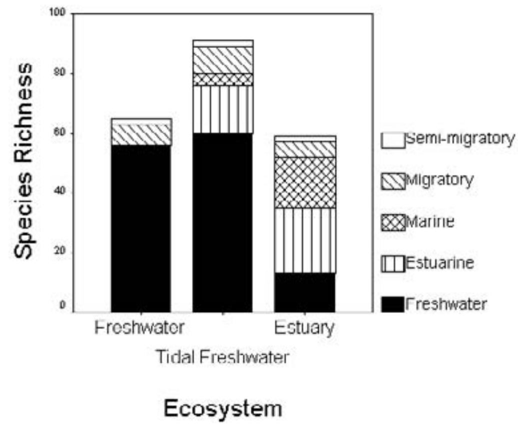

Figure 1. A comparison of fish species richness among non-tidal freshwater, tidal freshwater, and the saline estuary in the Virginia portion of the Chesapeake Bay. Species richness based on Garman and Smock 1999; Wagner 1999; Wagner and Austin 1999; Viverette 2004; and unpubl. data (from GCG).

native to the Chesapeake Bay include the American Shad (*Alosa sapidissima*), Hickory Shad (*A. mediocris*), Alewife (*A. pseudoharengus*), and Blueback Herring (*A. aestivalis*) (Jenkins and Burkhead 1994). The upstream migration of anadromous clupeids provides a substantial subsidy in the form of marine-derived carbon to the nutrient and energy budgets of coastal freshwater habitats each spring (Garman 1992; Garman and Macko 1998; MacAvoy *et al.* 2000). Reproductive fish are particularly nutritious prey due to lipid-rich eggs and sperm (Poole 1989) and represent a potentially important and predictable seasonal nutritional subsidy for piscivorous birds nesting within the Chesapeake Bay.

Shift from Spatially Widespread to Spatially Concentrated Fish Resources

Historically, migratory shads and herrings were abundant and geographically widespread in the Chesapeake Bay and its tributaries. Early European colonists describing the annual spawning runs along Chesapeake Bay tributaries consistently noted the immense quantity of herring and shad moving upstream each spring (Loesch and Atran 1994). This abundant fishery soon became an economically important industry (Foerster and Reagan 1977), with catches increasing dramatically during the 1800s as fishing techniques improved. The fishery peaked in the early 1900s with catches of American Shad in the Chesapeake Bay reaching eight million pounds annually. Archeological evidence indicates American Shad migrated upstream in Virginia tributaries as far as West Virginia (Garman and Nielsen 1992), and records from Thomas Jefferson's estate at Monticello, indicate a herring fishery as far upstream as Charlottesville, Virginia (J. Kauffman, Virginia Department of Game and Inland Fisheries, pers. comm.).

By the early 20th century however, overfishing, combined with dams blocking migration began to impact populations of anadromous fish along the Atlantic coast (Loesch and Atran 1994). Anadromous fish stocks declined steadily throughout the 20th century and in the 1970s, just as Bald Eagle and Osprey populations were beginning to recover, populations of anadromous fish in the Chesapeake Bay basin declined precipitously, experiencing up to a 90% reduction in abundance (Fig. 2; Garman and Nielsen 1992). The causes for the most recent declines include commercial overfishing, barriers to upstream migration, habitat loss, and the introduction of non-native fishes (Foerster and Reagan 1977; Garman and Macko 1998). The construction of dams in particular has restricted the range of anadromous fish by limiting access to inland spawning and nursery grounds (Loesch and Atran 1994). Until recently (e.g., construction of fishway at Boshers Dam, James River ca. 1999, Weaver *et al.* 2003), dams at the upstream limit of tidal influence have largely confined anadromous clupeid spawning activity to the tidal freshwater regions of large Chesapeake Bay tributaries (Jenkins and Burkhead 1994; McIninch and Garman 1999).

Figure 2. Commercial American Shad catch for the Chesapeake Bay from 1880-1972 (from Foerster and Reagan 1976).

Shift from Temporally Concentrated to Temporally Widespread (Resident Year-Round) Fish Resources

As migratory clupeids declined, there was a concomitant shift in the tidal freshwater fish community from migratory to non-migratory species, i.e., from a seasonally abundant resource to one that is available year-round. On an annual basis, non-migratory (i.e., resident) *Dorosoma* species, both Gizzard Shad (*Dorosoma cepedianum*) and the non-native Threadfin Shad (*D. petenense*), dominate clupeid assemblages in tidal freshwater habitats within the Chesapeake Bay. In a study of the relative abundance of clupeids in the James and Rappahannock rivers (Garman and Mitchell 1989), Gizzard Shad were the numerically dominant species. By the late 1990s migratory clupeids made up less than 1% of individuals and relative biomass of shads and herrings sampled annually in the tidal freshwater James River (Fig. 3; CBV, unpubl. data). Threadfin shad, introduced into the Chesapeake Bay system in the 1950s, and rare in the James through the 1960s (Jensen 1974), are now well established in western tributaries of the Chesapeake Bay (Jenkins and Burkhead 1994). However, recent data from Virginia and Maryland indicate that Gizzard Shad abundance may be declining from a peak in the late 1990s (J. Uphoff, Maryland Department of Natural Resources, unpubl. data; Maryland Department of Natural Resources 2007). The decline may be attributable to increasing populations of novel apex predators introduced into tidal freshwaters in the last 40 years (R. Greenlee, Virginia Department of Game and Inland Fisheries, pers. comm).

Shift from Migratory Planktivores to Apex Predators

In addition to a shift from migratory to resident species, tidal freshwater fish com-

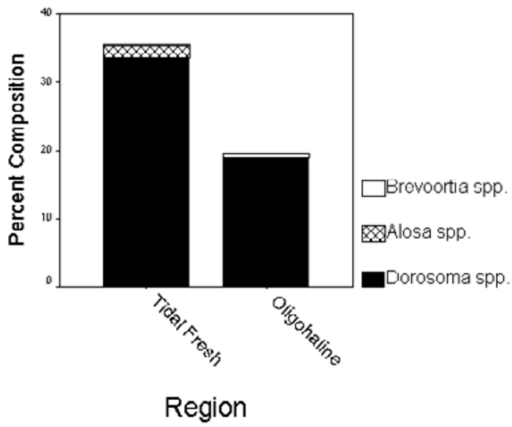

Figure 3. Contribution of anadromous shads (*Alosa* spp.) and resident shads (*Dorosoma* spp.) to fish assemblages of the James River, Virginia (R. Greenlee, Virginia Department of Game and Inland Fisheries unpubl. data; W. Bolin, Dominion Power, unpubl. data).

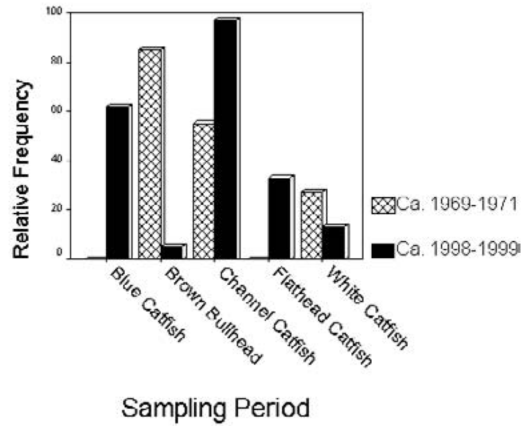

Figure 4. Frequency of occurrence of native and introduced catfish fish species in the James River, Virginia (Jensen 1974; R. Greenlee, Virginia Department of Game and Inland Species, unpubl. data; W. Bolin, Dominion Power, unpubl. data).

munities experienced a shift in trophic structure. Tidal freshwater fish assemblages along the Atlantic slope have few native piscivorous species, explaining perhaps the evolution of anadromous life-history strategies among migratory clupeids (McAvoy *et al.* 2000). However, concurrent (ca. 1975) with the severe declines in anadromous clupeid populations, the nonindigenous Blue Catfish (*Ictalurus furcatus)* and Flathead Catfish (*Pylodictus olivarus*) were introduced to the Atlantic slope of Virginia. Both catfish species are large and long-lived (up to 50 kg and 30 years) predators; adults prey extensively on fish and are able to ingest most native fishes found in tidal freshwater reaches (Chandler 1998; Graham 1999). Blue Catfish introductions occurred in the James, Rappahannock, Mattaponi, and Potomac drainages between 1974 and 1989, and Flathead Catfish introductions took place in the tidal James and Potomac River drainages (Occoquan Reservoir) between 1965 and the mid-1970s (Jenkins and Burkhead 1994; Edmonds 2003). Both Blue and Flathead catfishes are now well established in Virginia's coastal rivers, particularly tidal freshwater reaches (Fig. 4; Jenkins and Burkhead 1994; Edmonds 2003). More recently, Blue Catfish populations have been expanding in the fresh tidal portion of the Potomac River (SPM, pers. obs.). Flathead Catfish have recently been documented in the Potomac River (summer 2005, D. Hopler, Virginia Commonwealth University, pers. comm.) and the upper Bay in Maryland (J. Uphoff, Maryland Department Natural Resources, pers. comm.).

The Channel Catfish (*Ictalurus punctatus*) populations also increased substantially in the Chesapeake Bay since the 1970s. Like Blue Catfish and Flathead Catfish, Channel Catfish are not native to the Chesapeake Bay, but were introduced to the mid-Atlantic over 100 years ago (Sauls *et al.* 1998). Channel Catfish were the most common catfish in the James River of Virginia in the late 1990s (Fig. 4) and in 1996, comprised 93% of the commercially harvested catfish in Maryland's portion of the Chesapeake Bay (J. Uphoff, Maryland Department of Natural Resources, unpubl. data). However, since that time, Channel Catfish populations in some Chesapeake Bay tributaries may be declining as Blue Catfish and Flathead Catfish populations continue to expand (Jim Uphoff, Maryland Department of Natural Resources, unpubl. data; Maryland Department of Natural Resources 2007). Over the same 40-year period that the three non-indigenous catfish populations were expanding in the Bay, the smaller, native catfish species including the Brown Bullhead (*Ameiurus nebulosus*) and

White Catfish (*A. catus*), became rare in the mainstem of many of the Chesapeake Bay's tidal tributaries (Fig. 4).

Another introduced piscivore, the Largemouth Bass (*Micropterus salmoides*), also became more common in the last 40 years, along with the native Striped Bass (*Morone saxatilis*), an anadromous piscivore that spawns in tidal rivers (Fig. 5). Striped Bass experienced a population decline in the 1960s and 1970s but by the late 1990s the population was recovered fully (Uphoff 2003a).

Although the effect of introducing apex predators such as the Blue Catfish and Flathead Catfish to these relatively predator-poor coastal rivers is not well documented, introductions of apex predators elsewhere have been linked to declines in native fish populations (Moyle and Light 1996). Flathead Catfish and Blue Catfish may prey heavily on anadromous clupeids during the spring spawning run (Chandler 1998; Garman and Macko 1998) and the impact of these novel predators on on-going recovery efforts for American Shad and other migratory species, as well as their impact on native and naturalized catfish species, is not well understood.

Shift from Smaller to Larger Size Classes of Fish Resources

The increase in abundance and diversity of top predators within tidal freshwater fish communities resulted in a shift in size distribution of available fish prey toward larger size classes. A comparison of total lengths of the ten most abundant fish species in collections in the tidal freshwater James River in 1969 (Fig. 5; Jensen 1974) and 1999 (Fig. 6; Greenlee, VDGIF, unpubl. data; W. Bolin, Dominion Power, unpubl. data) suggests a substantial increase in available prey size during that 20-year period. Increased availability of larger prey may improve foraging efficiency by avian predators.

ECOLOGICAL IMPACTS OF SHIFTING FISHERY RESOURCES ON AVIAN PREDATORS

Bald Eagles

Access to relatively predictable, annual concentrations of prey, as represented by spawning migrations of anadromous fish, may have profound effects on the distribution and abundance of predators such as Bald Eagles (Willson and Halupka 1995; Restani *et al.* 2000). The annual spring spawning run of anadromous clupeids within the Chesapeake Bay coincides with the nesting season of Bald Eagles, which begin nesting in January and are feeding young during the peak of the runs in April and May (Markham 2004; Watts *et al.* 2006; ACM and

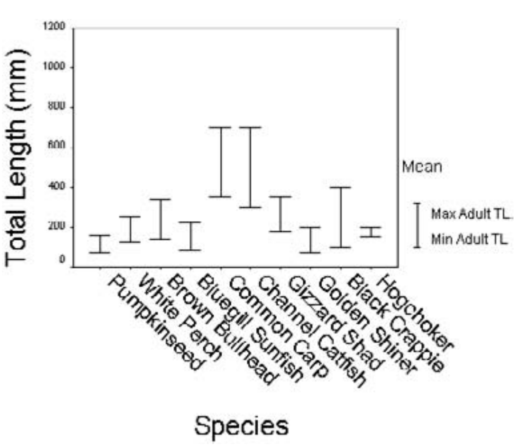

Figure 5. Total length of ten most frequently occurring fish species in the tidal freshwater reach of the James River, Virginia *ca.* 1968-1971 (Jensen 1974). Species ranked by frequency of occurrence (left to right).

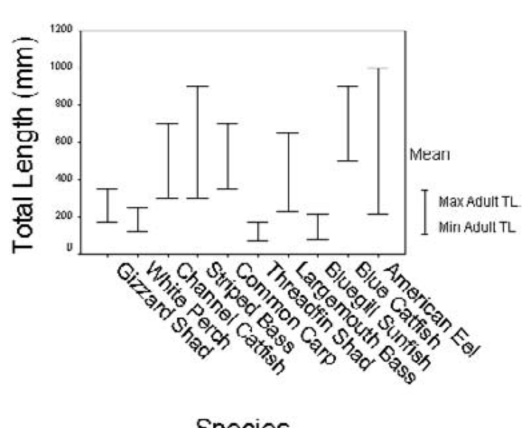

Figure 6. Total length of ten most frequently occurring fish species in the tidal freshwater reach of the James River, Virginia *ca.* 1998-1999 (Greenlee, Virginia Department of Game and Inland Fisheries, unpubl. data; W. Bolin, Dominion Power, unpubl. data). Species ranked by frequency of occurrence (left to right).

BDW, unpubl. data). In addition, Bald Eagles feed on carrion as well as live fish (Skagen *et al.* 1991). Carrion, in the form of post-reproductive carcasses of *Alosa* species, may be relatively plentiful during spring months because a significant percentage (50% to 70% in the mid-Atlantic region) of the adult clupeids dies after spawning (Leggett and Carscadden 1978; Browder 1995).

Anadromous fishes are important components of Bald Eagle diets in other regions of North America, where they congregate near spawning streams in both breeding and non-breeding seasons (Willson and Halupka 1995; Bennetts and McClelland 1997; Restani 2000; Restani *et al.* 2000). In Alaska, anadromous salmonids are an important prey item, and up to 90% of the salmon consumed by Bald Eagles is carrion (Imler and Kalmbach 1955). Studies in Manitoba and Saskatchewan documented a direct, positive relationship between Bald Eagle nest success and proximity to salmon spawning streams (Gerrard *et al.* 1975). In Maine, managers determined that recovery goal success for the Bald Eagle population was linked to restoration of anadromous clupeid populations, specifically Alewife (B. Owens, University of Maine, pers. comm.).

The importance of anadromous clupeids to the diets of Bald Eagles nesting along the mid-Atlantic coast has not been well documented. Only two published studies (Table 1) document Bald Eagle feeding activities in the Chesapeake Bay region during the breeding season. Tyrell (1936) did not report anadromous herrings or shads (*Alosa* spp.) in prey remains, a substantial proportion (55%) of which were unidentified. However, analyses of prey remains are biased in favor of prey with indigestible parts that decompose slowly (Todd *et al.* 1982; Simmons *et al.* 1991). Assessing the percentage of anadromous clupeids in bird diets using traditional methods is difficult because clupeids are relatively soft-bodied and leave scant skeletal remains that are unlikely to persist in the environment. To avoid these potential errors, Markham (2004) used nest cameras to identify fish prey delivered to Bald Eagle nests during the breeding season. Monitored nests were located along the Rappahannock, York, and James Rivers in Virginia. Fish accounted for 90% of prey items delivered and clupeids represented 45% of the identified fish (N = 625). Of the clupeid remains photographed and handled in nests, only anadromous species were observed (BDW, pers. obs.).

Gizzard Shad and Threadfin Shad are year-round (i.e., nonmigratory) residents of tidal freshwater rivers (Jenkins and Burkhead 1994), are increasing in abundance in many Chesapeake Bay habitats, and may, therefore, represent important prey resources for both resident and migrant Bald Eagles. Non-migratory shad are consumed by Bald Eagles in other regions (Southern 1973; Fischer 1982; Thompson *et al.* 2005) during breeding and non-breeding seasons. Gizzard Shad were a numerically important component (13% of prey) of breeding and migrant Bald Eagles diets in the tidal freshwater Hudson River during the mid-summer months (Thompson

Table 1. Bald Eagle (*Haliaeetus leucocephalus*) feeding studies from the Chesapeake Bay region (considering fish only). Note that Markham (2004) and Mersmann (1989) were observational studies and Haines (1998) and Tyrell (1936) were from prey remains.

Fish species	Markham (2004) (%, N = 695) Breeding	Mersmann (1989) (%, N = 253) Non-breeding	Haines (1988) (%, N = 45) Non-breeding	Tyrell (1936) (%, N = 44) Breeding
Shads and Herrings	40.86[a]	15.01[b]	0.00	0.00
Catfish	33.67	3.56	95.45[c]	44.44
Other (unknowns)	25.47	81.42 (68.38)	4.55	55.55

[a]Migratory and resident.
[b]Gizzard shad.
[c]Native brown bullhead (75%).

et al. 2005). In a study of wintering Bald Eagles in Illinois, Gizzard Shad was the primary prey item (Southern 1973; Fischer 1982) and Mersmann (1989) reported that Bald Eagles on northern Chesapeake Bay foraged heavily on winter-killed Gizzard Shad. In addition, introduced Threadfin Shad experience high mortality at water temperatures below 7°C (Jenkins and Burkhead 1994) and may provide an important food resource for resident and migrant Bald Eagles occupying the Chesapeake Bay during severe winters.

Catfish species (Ictaluridae) comprise a substantial proportion of Bald Eagle diets in North America, particularly inland populations (Haywood and Ohmart 1986; Grubb 1995; Mabie *et al.* 1995). Catfish prey remains persist in the environment due to the large pectoral girdle and spines, and may result in overestimation of catfish dietary importance. However, in a study of prey preference conducted on the upper Chesapeake Bay, Bald Eagles chose catfish species over other fish species (e.g., Gizzard Shad), and other prey types (e.g., mammals and waterfowl, DeLong 1990). Catfish were a numerically dominant item in the diets of breeding and migrant eagles on the tidal freshwater Hudson River between April and September (Thompson *et al.* 2005). Bullhead catfish comprised 35% of prey remains in Bald Eagle nests in Minnesota (Dunstan and Harper 1975) and 25% of prey identified in a diet study of both wintering and nesting Bald Eagles at inland sites in Maine (Todd *et al.* 1982).

In Markham's (2004) diet study of Chesapeake Bay Bald Eagles, catfish species comprised 34% of fish delivered to the nest and 31% of all nest deliveries. The catfish in Markham's study were not identified to species; however, based on anecdotal evidence (BDW, pers. obs.), non-native Blue Catfish, Channel Catfish, and in the James River, Flathead Catfish, likely provided the bulk of the catfish consumed. Native catfishes were a food resource for Chesapeake Bay Bald Eagles prior to the widespread introduction of non-indigenous Blue and Flathead Catfishes, *ca.* 1975. Tyrell's (1936) breeding season study of nest remains included catfish in nearly 45% of collections. Similarly, in a study of prey remains at a summer roosting site on the Potomac River, approximately 95% of prey remains consisted of catfish species, primarily native Brown Bullhead (Haines 1988). Catfish remains were observed during 232 (37%) of 630 visits to nests distributed throughout the Chesapeake Bay during the breeding seasons between 1978 and 1986 (K. Cline, Virginia Dept. of Game & Inland Fisheries, unpubl. data). Of 106 nest visits where catfish species were identified, White Catfish were present on 55 (52%), Channel Catfish were present on 52 (49%), bullhead species were present on 18 (17%), and Blue Catfish were present on only one (<1%).

Osprey

Since the early 1970s, the Osprey population in the Chesapeake Bay has more than doubled. Since recovery from pesticide-related declines, Osprey populations were initially concentrated in the Chesapeake Bay mainstem and the mouths of the major tributaries (Watts *et al.* 2004). Ospreys occurred rarely in tidal fresh and brackish portions the Chesapeake Bay tributaries in the 1970s, and were extirpated from some areas such as the tidal freshwater James River (Kennedy 1972; Watts and Paxton 2007). However, by the mid-1980s, Osprey populations in higher salinity regions had reached pre-pesticide levels, appeared to be approaching carrying capacity (Watts *et al.* 2004), and localized populations were beginning to exhibit signs of food stress such as brood reduction and sibling aggression (McLean and Byrd 1991a). In contrast, since the 1980s, Osprey populations within tidal freshwaters have experienced the highest colonization and growth rates in the Chesapeake Bay, and exhibit no signs of approaching carrying capacity (Watts *et al.* 2004).

The only published diet study of Ospreys within the Chesapeake Bay, conducted in the higher salinity reaches of the lower bay during the mid-1980s, showed that Atlantic Menhaden (*Brevoortia tyrannus*) comprised 75% of nest deliveries to Osprey nests (McClean and Byrd 1991b). Atlantic Menhaden are a major component of the diet of coastal

Osprey populations in New England (Poole 1989), coastal New Jersey (Steidl *et al.* 1991a) and the Delaware Bay (Steidl *et al.* 1991b). Unlike the anadromous clupeids, Atlantic Menhaden, a marine clupeid, spawn over the continental shelf. Larval Atlantic Menhaden move into the Chesapeake Bay as far upstream as tidal freshwater, but the larger, forage-size juveniles are most common in the middle to lower tributaries and mainstem areas of the Chesapeake Bay, where they remain throughout the spring, summer and fall (Murdy *et al.* 1997). Atlantic Menhaden are important forage for a variety of fish predators in the Chesapeake Bay such as Striped Bass, Weakfish (*Cynosion regalis*), and Bluefish (*Pomatomus saltatrix*), as well as supporting one of the most important commercial fisheries in the United States (Murdy *et al.* 1997; Uphoff 2003a).

During the early to mid-1980s Atlantic Menhaden stocks began to decline in the Chesapeake Bay (Fig. 7; Uphoff 2003b), coinciding with the first evidence of brood reduction and sibling rivalry recorded in lower Chesapeake Bay Osprey populations (McClean and Byrd 1991). Similar evidence of food stress was not apparent a decade earlier (Stinson 1977) when Atlantic Menhaden stocks were comparatively larger (Uphoff 2003b). By the early 1990s symptoms related to food stress were also being reported for fish piscivores, including Striped Bass and Weakfish, that are dependent on Atlantic Menhaden (Uphoff 2003b, 2006). Declining abundance of Atlantic Menhaden in higher salinity regions (e.g., Bay mainstem) may be negatively affecting Osprey population stability in high salinity areas of the Chesapeake Bay at the same time that comparatively abundant fish prey resources in oligohaline and tidal freshwater river habitats may be supporting expansion and local population growth in Ospreys.

The Osprey's arrival on Atlantic slope breeding grounds coincides with the beginning of the spring anadromous clupeid spawning run and later, during the height of the spawning season, Ospreys are laying and incubating eggs (Poole 1989; M. Byrd, College of William and Mary, unpubl. data). Anadromous clupeids are an important dietary component for Osprey nesting along the Atlantic coast, particularly riverine populations (Jamieson *et al.* 1982). Along the southern coast of New England, newly arrived adult Ospreys fed on anadromous her-

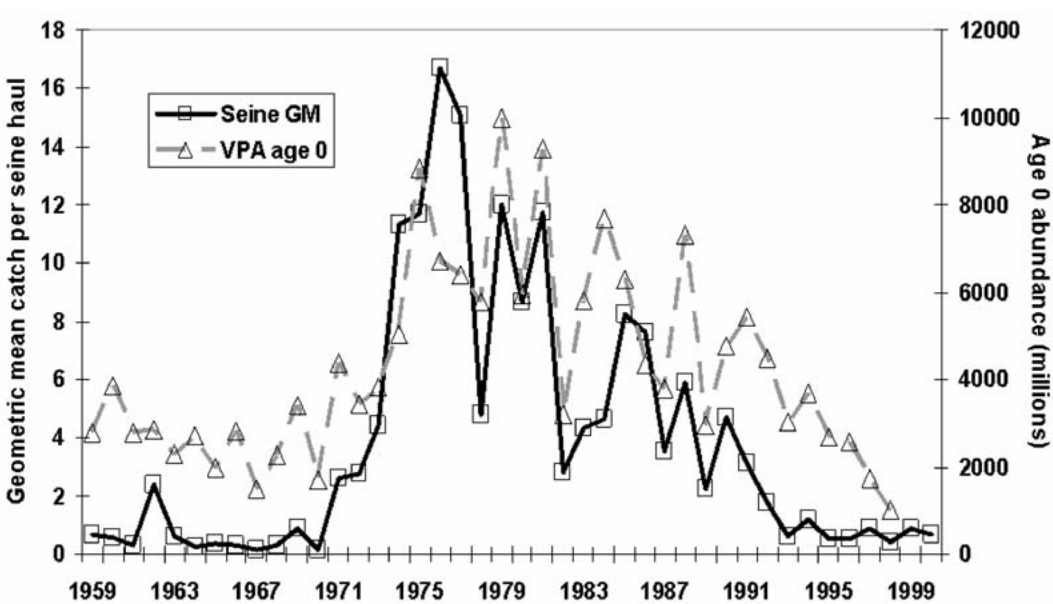

Figure 7. Geometric mean catches per standard seine haul for Atlantic Menhaden collected from Maryland's portion of Chesapeake Bay, and VPA (virtual population analysis) estimates of age zero Atlantic Menhaden abundance, 1959-2000 (from Uphoff 2003).

ring (*Alosa* spp.) almost exclusively, but switched to other locally abundant species after herring availability declined (Poole 1985, 1989). Ospreys from mid-Atlantic and New England regions may continue feeding nestlings into late July and August (Poole 1989), well after the anadromous clupeids have left spawning grounds, requiring a switch to alternative prey. For instance, in Nova Scotia, inland Osprey populations nesting on rivers and lakes fed heavily on Alewife and Blueback Herring early in the breeding season when spawning fish were abundant, but switched to foraging for alternative prey in the estuary, an average distance of 23 km from nest sites, later in the breeding cycle (Jamieson *et al.* 1982).

Unlike Ospreys in Nova Scotia that travel to the lower estuary to feed when the spawning migration ends (Jamieson *et al.* 1982), Ospreys inhabiting tidal freshwaters in the Chesapeake Bay may exploit a variety of locally abundant fish prey during the latter portion of the breeding season. For instance, catfish prey contributed to Osprey diets in Delaware Bay (Steidl *et al.* 1991b) and made up a small but significant proportion of prey deliveries in the lower tributaries and mainstem regions of the Chesapeake Bay (McClean and Byrd 1991). Catfish made up the bulk of prey taken by Ospreys nesting in Idaho, but consumption varied significantly with the availability of spawning salmonids, the second most numerous prey item observed (Van Daele and Van Daele 1982). In addition to anadromous clupeids and catfish, inland populations of Ospreys in other regions of North America feed on Gizzard Shad and Threadfin Shad (Swenson 1979; Edwards 1988), centrarchids (Dunstan 1974; Swenson 1979; Edwards 1988), and a variety of benthic species (Swenson 1979; Van Daele and Van Daele 1982; Grover 1984), all of which are abundant in tidal freshwaters of the Chesapeake Bay.

Other Waterbird Species

Great Blue Heron distribution within the Chesapeake Bay is also heavily skewed toward oligohaline and tidal freshwater habitats (BDW, unpubl. data). The first breeding record for Double-crested Cormorants (*Phalacrocorax auritus*) occurred in 1978 within the tidal freshwater James River (Blem *et al.* 1980). By 1995, the cormorant breeding population in the tidal freshwater James River grew to over 200 pairs (Watts and Bradshaw 1996). Heron and cormorant feeding studies are lacking for the Chesapeake Bay, and most such studies conducted elsewhere are in response to perceived depredation of commercial fisheries or aquaculture facilities. These studies indicate that both waterbird species feed on a variety of fish species in tidal freshwaters (Hoy 1994; Trapp 1998; Simmonds *et al.* 2000; Glahn *et al.* 2002; Steinmetz 2003; Fenech *et al.* 2004) including migratory and non-migratory shads and herrings, yellow perch, catfishes, and centrarchid species.

MANAGEMENT IMPLICATIONS

The predator-prey interactions among piscivorous birds and fish prey has received little attention from wildlife managers (Steinmetz *et al.* 2003). The potential role of fish population dynamics and commercial harvest in affecting avian distribution, including those that are of national conservation concern such as the Bald Eagle, is largely undescribed for the Chesapeake Bay. In fact, most Chesapeake Bay ecosystem and management models (e.g., Baird and Ulanowicz 1989) ignore avian predators and competitors, and fishery stock assessments for the region generally fail to incorporate these potentially important ecological interactions (Chesapeake Fisheries Ecosystem Plan, Technical Advisory Panel 2004). Fisheries management decisions may, however, directly impact piscivorous bird populations in the Chesapeake Bay. For example, considerable resources have been invested in American Shad recovery efforts in the Chesapeake Bay watershed (Weaver *et al.* 2003) and successful restoration of anadromous fishes into historical habitats could have an impact on distribution of avian predators. The effect on American Shad recovery efforts of recently introduced piscivorous fishes, which feed on anadromous clupeids (McAvoy *et al.* 2000), is unclear. Thus, catfish management could influence both prey (shads) and predator (piscivorous birds) distribution in the region. Additionally, Maryland has conducted a Chesapeake Bay-specific stock assessment of Atlantic Menhaden (Uphoff 2003a) and the NOAA Chesapeake Bay Program is currently supporting a similar assessment (J. Uphoff, pers. comm.), the results of which could influence future management decisions within the Bay.

Further conservation implications may result from documented shifts in historic trophic relationships among the fish and piscivorous bird communities within tidal freshwaters. Bald Eagles, Osprey, and other piscivorous birds are feeding at a higher trophic level in Chesapeake Bay tributaries where large, long-lived, and nonidigenous fish predators are now established. Such shifts may lead to greater risks from bioaccumulation of toxic compounds. Garman *et al.* (1998) documented critical levels of PCB's in James River Blue Catfish populations in an area also inhabited by the east coast's largest population of both breeding and non-breeding Bald Eagles (Watts and Whalen 1997). High methyl mercury levels have led to fish consumption advisories within tidal freshwater tributaries of the York and Piankatank Rivers in Virginia, as well as impoundments along the James and Chickahominy (Virginia Department of Environmental Quality 2007). Because the Chesapeake Bay Bald Eagle populations represents a nexus of three distinct breeding populations (Buehler *et al.* 1991; Watts *et al.* 2007), the conservation implications may, in fact, reach well beyond the borders of the Chesapeake Bay basin.

Finally, in order to understand and address research and conservation issues surrounding fish-bird interactions in the Chesapeake Bay, better communication and collaboration among fisheries and avian researchers should be encouraged. Development of, and access to, accurate and relevant data regarding the status, distribution, and abundance trends for fish communities in estuarine and tidal freshwater habitats of the Chesapeake Bay is integral to understanding patterns of distribution and abundance of waterbird populations. However, the challenges in obtaining, analyzing, and interpreting existing fisheries data are considerable, and include: 1) few published studies; 2) published studies that do exist are primarily driven by concerns about perceived depredation of fish stocks by avian predators (e.g., see Cowx 2003); 3) limited access to fisheries data; and 4) fish stock assessment techniques are unfamiliar to avian ecologists, include inherent biases, and may compromise accurate data interpretation. Collaborative efforts between fishery scientists and avian ecologists, along with the use of new technologies, including nest video cameras (Watts *et al.* 2004), stable isotope analyses (MacAvoy *et al.* 1998; Knoff *et al.* 2001), and hydroacoustics (Speckman 2005) may overcome these challenges, eliminate data gaps, and ultimately lead to better ecosystem management of the Bay's living resources.

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