

ABSTRACT

Title of Document: HABITAT USE OF SURF SCOTERS WINTERING
IN THE MESOHALINE REGION OF THE
MAINSTEM CHESAPEAKE BAY, MARYLAND

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Surf scoters (*Melanitta perspicillata*) are common in coastal waters during winter, but remain one of the least studied waterfowl in North America despite indications of a declining population. Surf scoter foraging was investigated in the mesohaline region of the Chesapeake Bay during 2004-2005. The study compared biological and physical characteristics of benthic habitats among scoter feeding and non-feeding sites. Benthic samples were taken seasonally via a Peterson grab. Surf scoters foraged over oyster bar, packed clay and sand habitats, but avoided mud habitats. Clam densities varied seasonally, increasing over winter regardless of scoter foraging activity. Foraging activity appeared to influence hooked mussel densities on hard bottoms. Lack of foraging over mud habitats, where visibility can be limited, suggests that surf scoters may be visual predators and may have been impacted by habitat sedimentation. The correlation of hooked mussel with oyster bars suggests that oyster restoration efforts may positively impact surf scoters.

HABITAT USE OF SURF SCOTERS WINTERING IN THE MESOHALINE
REGION OF THE MAINSTEM CHESAPEAKE BAY, MARYLAND

By

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INTRODUCTION

Surf Scoters

A member of a broad group of waterfowl known as seaducks (Anseriformes, tribe Mergini), surf scoters (*Melanitta perspicillata*) are a common winter waterfowl species indigenous to North America (Savard et al. 1998). Seaducks are generally characterized by inhabiting coastal waters in winter, the ability to dive, and a tolerance to salt water and cold temperatures (Weller 2001). Concern over seaducks has been increasing as a number of populations appear to be declining (Kehoe 1994, Elliot 1997, and Caithamer et al. 2000), yet surf scoters are one of the least studied North American waterfowl (Savard et al. 1998). The Sea Duck Joint Venture (SDJV), under the auspices of the North American Waterfowl Management Plan, has outlined research priorities to improve management of seaducks in an effort to understand and mitigate these apparent population declines. In response to informational needs identified by the SDJV, this study sought to quantify the foraging ecology of surf scoters within the wintering area of Chesapeake Bay through the identification of key biological and physical characteristics of benthic habitats in which they forage.

Distribution and Population Trends

Surf scoters are a common winter waterfowl species found on both the Atlantic and Pacific coasts of North America, and unlike white-winged (*Melanitta fusca*) and black scoters (*Melanitta nigra*), they are endemic to North America (Savard et al. 1998). Surf scoters are thought to comprise Atlantic and Pacific populations, with their annual distribution typically categorized based on season and life-history stage into wintering, staging, breeding, or molting grounds. Wintering and staging surf scoters are typically

found in coastal systems, as are molting males, whereas large lakes and ponds in the boreal forest are used as breeding habitats (Perry et al. 2006).

During the winter along the Atlantic coast, surf scoters range from Newfoundland to Florida (Bellrose 1980), although the mid-Atlantic region typically supports the highest densities (Stott and Olson 1973). An examination of migrational patterns conducted by Perry et al. (2006) from 2001 to 2005, where 20 surf scoters were tracked via satellite telemetry for approximately one year, provided the first analysis of surf scoter distribution along the Atlantic coast. Scoters used for the study by Perry et al. (2006) were collected prior to spring migration on Chesapeake Bay and consisted predominantly of females. Scoters that were able to be tracked the following winter exhibited strong site fidelity, returning to Chesapeake Bay for at least a portion of the winter. In addition, one scoter traveled south to Pamlico Sound, North Carolina before returning to Chesapeake Bay and a number of scoters spent time around Cape Cod, Massachusetts and along the Atlantic coast while en route to the Bay.

In the Chesapeake Bay, Perry et al. (2006) found surf scoters to be widely distributed through the mainstem and major tributaries in both Maryland and Virginia. The majority of satellite telemetry observations occurred in the middle portion of the Bay, between Kent Island and the Rappahannock River, as well as within the Potomac and Choptank Rivers. The highest concentration of observations was located within Maryland, although it is possible this was an artifact of where scoters were instrumented (Choptank River and mainstem locations north).

An aerial seaduck survey conducted by Koneff et al. (2005) during February of 2005 found similar results, although they found the greatest number of observations

occurred in the Virginia portion of the Bay. However, most of the Virginia observations represented rafts of scoters in small numbers (between 5 and 10) whereas most Maryland observations represented large rafts of scoters, the largest of which contained more than 10,000 scoters, located in Herring Bay. It is worth noting that this survey was not designed to approximate the number of surf scoters in the Bay, but rather test the efficacy of two survey techniques. Therefore, the large numbers of observations in the Virginia portion of the Bay may reflect sampling bias due to more intensive surveys in this area.

Estimating population size and trends for surf scoters has proven difficult along the Atlantic coast, including the Chesapeake Bay. Along the Atlantic coast, Caithamer et al. (2000) analyzed four survey techniques (mid-winter waterfowl surveys, Christmas bird counts, harvest indices, and breeding bird surveys) to estimate long-term trends in abundance and concluded that surf scoter populations had declined from 1972 to 1993. However, similar analyses have not been conducted in Chesapeake Bay. Within the Bay, mid-winter waterfowl surveys (MWS) are currently the only indication of long-term population trends for scoters. Analysis of MWS data (five-year running average; data obtained from J. Serie, U.S. Fish and Wildlife Service, 2006) since 1955 suggests a long-term cyclical pattern in scoter numbers and also reflects the downward trend indicated by Caithamer et al. (2000) during the 1972-1993 period (Figure 1). Since 2000, however, MWS data suggests a large increase in surf scoters over the 1990s baseline, the cause of which is not known. Throughout the period of 1955-2006, the population of scoters within Chesapeake Bay, as estimated by MWS, has averaged 9,200 scoters annually. Although the MWS is a useful metric with which to assess long-term trends, this average population value is not considered an accurate representation of population size. MWS

are conducted in near-shore waters and likely underestimate numbers of surf scoters, which typically occur in offshore waters. Additionally, high temporal and spatial variation of large rafts frequently lead to high inter-annual variability in population estimates (Koneff et al. 2005). Further limiting the accuracy of the MWS is the combining of all three species of scoters (surf, white-winged, and black) into a single variable, although surf scoters are considered to constitute the majority of the populations (D. Forsell, U.S. Fish and Wildlife Service, pers. comm.). To date, the only population size estimate of surf scoters in Chesapeake Bay was conducted during the winter of 1992-1993 through aerial surveys. At that time the population was estimated to be 120,000 scoters, approximately one-third of the Atlantic population (D. Forsell pers. comm.).

Food Habits and Habitats

Similar to other species of seaducks, surf scoters feed on a variety of marine/estuarine invertebrates, especially bivalves (Cottam 1939, Stott and Olson 1973, Vermeer 1981, Nyström et al. 1991, Bustnes et al. 2000, Lovvorn et al. 2003). Throughout much of their range, surf scoters commonly feed sub- and inter-tidally on epifaunal invertebrates, particularly blue mussels (*Mytilus edulis*; Cottam 1939, McGilvrey 1967, Vermeer 1981). However, surf scoters also feed on a variety of infaunal bivalves. In British Columbia, Japanese littleneck (*Venerupis philippinarum*) and purple mahogany-clam (*Nuttallia obscurata*) are frequently consumed at inter-tidal and shallow sub-tidal aquaculture sites (Lewis et al. 2005). Along the New Hampshire and Massachusetts coastline, Stott and Olson (1973) found that the arctic wedgeclam (*Mesodesma arctatum*) and the Atlantic razor (*Siliqua costata*) constituted the majority of surf scoter diet. Additionally, McGilvrey (1967) found that short yoldias (*Yoldia*

sapotilla) as well as blue mussels were major constituents of surf scoter diet along a similar portion of coastline. Analyses by M. C. Perry et al. (USGS Patuxent Wildlife Research Center, unpub. data) in Massachusetts and the Canadian Maritimes also found a variety of bivalves consumed, including Atlantic jackknife clam (*Ensis directus*), Atlantic surfclams (*Spisula solidissima*), and blue mussel.

Although bivalves constitute a significant portion of surf scoter diet, other invertebrate species are also consumed, including gastropods, crustaceans, and amphipods (Cottam 1939, Stott and Olson 1973, Vermeer 1981). Surf scoters will also opportunistically switch from traditional food sources to ephemerally superabundant resources. Along the Pacific coast, surf scoters routinely consume energetically rich herring spawn over a 2-week period in late-winter (Vermeer 1981, Bishop and Green 2001). Additionally, Lacroix et al. (2005) observed surf scoters feeding on a large aberrant bloom of polychaetes (*Ophryotrocha* sp.) in British Columbia.

Further suggesting a varied diet, Perry et al. (2007) found a variety of epifaunal and infaunal invertebrates consumed by surf scoters (n = 246) in Chesapeake Bay. In their analyses, Perry et al. (2007) found that dwarf surfclam (*Mulinia lateralis*) and hooked mussel (*Ischadium recurvum*) represented the most frequently consumed prey (35% and 22% of their diet, respectively; Figure 2) Bay wide. A more detailed analysis of food habits by Perry et al. (2007) indicated spatial differences in consumption, especially in association with the north – south salinity gradient. The hooked mussel was the dominant food item in scoters collected from the Choptank River and Herring Bay, whereas dwarf surfclam was the dominant prey near Tilghman Island. Farther south at Smith Island (bordering Maryland and Virginia), species associated with high-mesohaline

and polyhaline waters, such as false angel wing (*Petricola pholadiformis*), Atlantic jackknife clam (*Ensis directis*), and stout razor clam dominated surf scoter consumption. Analyses of surf scoters near the mouth of the Bay would have likely yielded additional bivalve species characteristic of high salinity waters.

Although their wintering habitats have been poorly examined, surf scoters appear to use a variety of habitat types. Most studies conducted to date have been limited to coastal waters of British Columbia and Vancouver Island along the Pacific coast. Consistent with a diet containing large amounts of blue mussel, scoters have been found in association with rocky substrates (Vermeer 1981, Lacroix 2001). Further, surf scoters are commonly found in fjords in British Columbia, where they prey on blue mussels attached directly to the fjord wall (Vermeer 1981). A detailed analysis identified scoters as major intertidal predators within a fjord/sound system, where the majority of scoters were associated with rocky substrates on or near reefs (Lacroix 2001). These scoters principally fed intertidally, and were primarily located in waters of 10 m or less. Conversely, scoters at nearby Vancouver Island were positively associated with sandy substrates, where they fed on infaunal clams (Žydelis et al. 2006). Although this study found different substrate use, similar to LaCroix (2001), scoter feeding was strongly associated with the intertidal zone.

Along the Atlantic coast, analyses of winter habitat use by surf scoters have been limited (Savard et al. 1998). Stott and Olson (1973) found a preferential use of sandy beaches by surf scoters along the New Hampshire coastline, with decreasing abundance as the percentage of rocky habitat increased. However, 19% of surf scoters analyzed by Stott and Olson (1973) for food habits contained blue mussel, suggesting that some

feeding over rocky/hard substrates was occurring. No known habitat analyses have been made within Chesapeake Bay.

Chesapeake Bay

Since the onset of colonization of the region, the Chesapeake Bay has experienced declines in environmental quality via processes including increased turbidity and eutrophication, which have had significant impacts on benthic communities (Holland et al. 1977, Officer et al. 1984, Dauer et al. 2000). While there are many pathways through which changes in benthic communities have occurred, declines in oyster abundance, increased sedimentation, and recurring deep-water hypoxia are some of the most pervasive (Newell 1988, Kemp et al. 2005). Because surf scoters rely on benthic communities for food resources, these perturbations have the potential to impact surf scoters through negative influences on foraging ecology.

Loss of Oyster Habitats

The Chesapeake Bay was historically one of the most productive systems for oysters in the United States. However, loss of oyster beds as a result of a number of anthropogenic and environmental perturbations has led to reduced harvests and cascading effects on the ecology of the Bay (Brooks 1891, Kennedy and Breisch 1981, Rothschild et al. 1994). Able to tolerate a wide range of salinities, the eastern oyster (*Crassostrea virginica*) is a common estuarine species along the Atlantic coast (Kennedy and Breisch 1981). In the Chesapeake Bay, oysters typically occur from the mesohaline central Bay to the polyhaline southern Bay in waters 2.5 to 7.6 m in depth (Lippson and Lippson 1997).

Oysters, and their associated reefs, are a vital ecological component of the estuarine ecosystem (Wells 1961, Newell 1988). Research by Meyer and Townsend (2000) found 33 macrofaunal species utilizing oyster reefs in the southeastern United States. In the Chesapeake Bay, restored oyster bars are colonized by large densities of sea squirts (*Mogula* sp.), hooked mussels, barnacles (*Balanus* sp.), small fish, and many other species (Rodney and Paynter 2006). The multitude of fauna inhabiting oyster habitats likely provides a significant prey source for surf scoters, especially since hooked mussel (22% of diet) is prevalent on oyster bars.

Concerns over oyster numbers in Chesapeake Bay were first noted in the late 19th century as harvests in the Bay reached their peak. By 1874, harvests had reached 14 million bushels (Kennedy and Breisch 1981) and five years later that number peaked to nearly 17 million bushels (Warner 1976). A fisheries report in 1880 found that nearly 120 million pounds of oysters were harvested that year alone (see Ernst 2003). The unsustainable level of harvests in Chesapeake Bay was recognized by Brooks (1891), who advocated increased use of oyster aquaculture in Maryland to supplant decreasing harvests and to meet increasing demand. In the 1950s harvests of 35 million pounds were being recorded. The subsequent 50 years resulted in further declines in the Bay oyster population, with recent harvests at 1% of their historic levels.

Although the initial drop in the oyster population within the Bay was primarily a result of overfishing, infections from two diseases have decimated stocks since the 1950s and have severely hampered restoration efforts. Dermo disease, caused by the parasite *Perkinsus marinus* (Mackin et al. 1950, Andrews and Hewitt 1957, Rothschild et al. 1994), was first detected in the Bay in 1949. Easily transmitted from oyster to oyster in

high salinity waters (>15; Paynter 1996), Dermo typically infects oysters older than one year of age, with initial retardation in growth eventually leading to death (Paynter and Burreson 1991, Paynter 1996). Because Dermo typically causes oyster mortality before they reach market size, high infection rates can inhibit reef development and commercial harvest. Since 1959, a second disease, MSX (*Haplosporidium nelsoni*), which also causes significant mortality before oysters reach market size, has infected oyster populations throughout the Bay (Andrews 1966, Andrews and Wood 1967). The combination of these diseases has all but eliminated the oyster fishery in Virginia and has severely hampered efforts to replenish oyster populations in Maryland.

Sedimentation

Elevated sedimentation associated with land-use changes has been associated with burial of oyster reefs (Seliger and Boggs 1988, Rothschild et al. 1994, Smith et al. 2003) and declines in submerged aquatic vegetation (Davis 1985). Concomitant with colonization of the region and clearing of forests for agricultural use, sedimentation rates in the Bay increased significantly in the 18th century (Brush 1989, Cooper 1995). By 1800, approximately 50% of the watershed had been converted from forests for agriculture or other land uses, subsequently resulting in impacts to benthic communities (Davis 1985). By the late-19th century nearly 80% of forests had been cleared for agriculture (Brush 1989) and increased mechanization of farming practices exacerbated soil runoff, turbidity, and sedimentation during this time (Cooper 1995). Documented changes in benthic ostracod communities in the 1800s were largely attributed to these changes in land-use and resultant increases in turbidity (Cronin and Vann 2003). This trend reversed slightly in the early 20th century, as farm abandonment (in the 1930s) and

the construction of dams on major tributaries resulted in a slight decrease in sedimentation rates (Brush 1989, Cooper 1995). Sedimentation rates have continued to decline slightly through the 20th century (Cooper 1995), although this has been coupled with increased urbanization and fertilizer use and thus increased runoff of nutrients and toxins (Davis 1985).

Although increased sedimentation rates affect a number of Bay habitat types (e.g., submerged aquatic vegetation beds), the contribution of sedimentation to declines in oyster beds has the greatest potential to directly impact surf scoter foraging. The primary mechanism by which sedimentation impacts oyster reefs is through siltation, or covering, of previously viable oyster beds with sediment (Seliger and Boggs 1988, Rothschild et al. 1994). Lenihan (1999) found that oyster bars with reduced vertical profiles, such as those subject to harvest pressure, were subject to altered hydrodynamic action relative to reefs with greater heights. These shorter reefs experienced reduced flow rates and increased sedimentation that resulted in burial of 91% of the reef in 16 months (Lenihan 1999). The burial of oyster reefs not only leads to high individual mortality rates, but also inhibits rejuvenation of reef height through reduced recruitment of settling oyster spat (Seliger and Boggs 1988, Rothschild et al. 1994, Lenihan 1999). While oysters can be effective in removing suspended sediments from the water column through the biodeposition of pseudofeces (DeAlteris 1988), excess sediments can reduce metabolic efficiency and gill function (Rothschild et al. 1994). Finally, sedimentation can increase oyster susceptibility to infection from disease. Lenihan (1999) attributed increased *Perkinsus marinus* infection in oysters at the base of reefs and on short reefs to reduced flow rates and the resultant increase in sedimentation and decreased quality of food.

Hypoxia

In addition to increased sedimentation rates, anthropogenically induced land-use changes have also resulted in increased nutrient run-off to the Bay. Progressive eutrophication of the Bay has resulted in seasonally recurring hypoxia, which has served as an additional stressor to both oyster beds (Lenihan and Peterson 1998) and soft bottom communities. Low oxygen conditions (hypoxia) were first reported in the Bay in 1938 (Newcombe and Horne 1938) and subsequent analyses of water quality data have shown these events to be increasing in both frequency of occurrence and volume of water affected (Officer et al. 1984). On a spatial scale, low oxygen events are typically found in the deep water habitats (>10 m) in the Bay mainstem and major tributaries (Dauer and Ranasinghe 1992). Causes of this increase of occurrence in low dissolved oxygen are typically attributed to anthropogenic nutrient inputs from both non-point and point sources (Cooper and Brush 1991). Increased nutrient inputs produce large blooms of phytoplankton, which eventually die, sink, and decompose, inducing a high biological oxygen demand in bottom waters. The persistence of hypoxia generally occurs when high benthic oxygen consumption combines with a stratified water column that prevents mixture with oxygen-rich surface waters (Seliger et al. 1985, Diaz and Rosenberg 1995).

Although most analyses of Bay benthos occurred after hypoxia became an annual event (Kemp et al. 2005), the effects of hypoxia on benthic communities in Chesapeake Bay have been well documented. Throughout the mainstem and major tributaries of the mesohaline Chesapeake Bay, degraded communities of deep-water benthos are strongly related to the recurrence of hypoxia (Dauer et al. 2000). These degraded communities typically are characterized by reduced benthic biomass, species diversity, and altered

community structure (Holland et al. 1977, Dauer and Ranasinghe 1992, Dauer et al. 2000), modifying energy flow across trophic levels (Breitburg et al. 1997). Holland et al. (1977) found benthic fauna in nine meter mud habitat to be in a cyclical pattern of mortality and recolonization. These restructured benthic communities due to hypoxia (Diaz and Rosenberg 1995) tend to be composed of large numbers of opportunistic species (such as dwarf surfclams) and a decreased number of equilibrium (typically larger, longer lived) species (Dauer and Ranasinghe 1992).

Although hypoxic events typically occur below the feeding depth of scoters (up to 10 m based on observational and preliminary satellite telemetry data), hypoxic events have sometimes been observed in shallower waters where surf scoters feed. Seliger et al. (1985) documented a prolonged and severe low oxygen event in 1984 in which the pycnocline moved upward to approximately five meters. The cause of this hypoxic movement was attributed to a persistent southwest wind that increased the amount of high salinity water into the Bay. A subsequent analysis of oyster bars conducted that fall in the Choptank River by Seliger et al. (1985) found high mortality of all shellfish and fouling organisms (typically hooked mussel and barnacles, *Balanus* spp.) below six meters.

This Study

With an estimated one-third of the Atlantic surf scoter population spending a portion of the winter within Chesapeake Bay (D. Forsell, pers. comm.), the Bay is clearly a seasonally important habitat for surf scoters. Further, with an average depth of 6.4 m, the Chesapeake Bay provides a large area of accessible foraging habitat for surf scoters. Although no assessment of surf scoter habitat has been conducted within the Bay, the

distributions of other diving duck species in the Bay have been shown to be directly related to food resources and benthic habitats (Perry and Uhler 1988). Canvasback (*Aythya valisineria*) and redhead (*Aythya Americana*) traditionally feed on submerged aquatic vegetation (SAV) and winter in large numbers within the Bay (Perry et al. 1981, Perry and Uhler 1988). However, following severe degradation and loss of SAV beds (in addition to market hunting in the early 20th century), redhead populations plummeted, and remain low today, whereas a switch to a primarily molluscivorous diet has allowed canvasback populations to recover, albeit at reduced levels (Perry and Uhler 1988, Perry and Deller 1994). These population trends for canvasbacks and redheads in relation to resource availability demonstrate the impacts that changes in benthic habitats can have on diving duck species.

With the ongoing degradation of many benthic habitats in Chesapeake Bay and the lack of information on surf scoter foraging within the Bay, there is a clear need to assess the quality of surf scoter habitats. This study employed a novel comparative approach to elucidate the habitat use of foraging surf scoters within the mesohaline portion of the mainstem Chesapeake Bay. To determine the habitat of foraging surf scoters, benthic faunal composition and habitat composition were comparatively analyzed at sites with and without feeding scoters (hereby called Feeding and Non Feeding sites, respectively). This study sought to test the following null hypotheses:

H₁: Composition and characteristics of benthic macrofauna at Feeding sites do not differ from Non Feeding sites.

H₂: Biomass and density of benthic macrofauna at Feeding sites do not differ from Non Feeding sites.

H₃: Sediment composition at Feeding sites does not differ from Non Feeding sites.

Specific objectives of this study were to:

1. Identify locations of feeding surf scoters in the mesohaline portion of Chesapeake Bay and identify sites in which feeding scoters do not occur.
2. Determine benthic macroinvertebrate community composition and seasonal trends at Feeding and Non Feeding sites.
3. Quantify the sediment grain size, organic content, and habitat type at all sites.
4. Determine the relationships among prey communities, sediment types, and distribution of foraging surf scoters in the mesohaline portion of the Chesapeake Bay.

STUDY AREA

The Chesapeake Bay is the largest estuary in the United States and, with a deep-water channel extending throughout its length, is typical of a drowned-river valley. Salinity in the Bay is marked by a strong north-south gradient, ranging from freshwater at the head of the Bay to over 30 at the mouth. Additionally, salinities along the eastern portion of the mainstem tend to be slightly higher than those on the western side due to strong freshwater inflows from western shore tributaries combined with the Coriolis effect. The mainstem of the Bay has a mean depth of 9.14 m, with bathymetry gradually sloping to a deep mid-Bay channel. Tidal fluctuations range from 0.9 m at the mouth to 0.3 m near Annapolis, MD before increasing to 0.6 m at the head of the Bay (Lippson and Lippson 1997).

The specific location of the study area extended from the mouth of the West River south to Holland Point on the western shore, and Poplar Island south to the southern portion of Tilghman Island on the eastern shore (Figure 3). The exact location of the study area was based on prior telemetry work by Perry et al. (2006; Figure 4). This study routinely found large numbers of surf scoters from 2001-2005 in and around the area of the study area, where 19 surf scoters implanted for satellite telemetry were captured. Additionally, of the scoters that were able to be tracked the following winter, all returned to the Bay, with 3 returning to this region. Western shore bathymetry gradually slopes to the deep-water channel, whereas the eastern shore is marked by a broad shelf extending to a comparatively abrupt slope to the channel. The deepest portion of the Bay (53.0 m) is located in the channel just outside the study area (Lipson and Lipson 1997). Salinity in the study area is typical

of a mesohaline regime, with a mean salinity of 11.5 (MD DNR, <http://mddnr.chesapeakebay.net/eyesonthebay>). A prominent feature within the study area is Poplar Island, which had receded from over 404 hectares in the 19th century to less than 4 hectares in 1990 due to erosion, sea level rise and land subsidence. Since the late-1990's, this island has been the subject of a massive restoration project using dredge spoil and now consists of approximately 230 hectares of upland and 230 hectares of wetland.

METHODS

Distribution of Surf Scoters

Boat-based Surveys

Many diving duck habitat studies have utilized land-based observations (Nilsson 1972, Hamilton 2000, Lewis et al. 2005) or aerial surveys (Mitchell 1992, Kaiser et al. 2006) to determine waterfowl locations. However, large amounts of private land along the shores of the study area, the distance of scoter flocks from shore (typically >1 km), and large spatial variability of flock locations precluded the use of land-based observations. Additionally, conducting a special aerial survey of the study area was cost prohibitive.

Therefore, boat-based surveys were used to determine the numbers and locations of seaducks within the study area. Surveys were conducted monthly in December, February, March, and April during winter 2004 - 2005. November and January surveys were not conducted due to weather and significant icing, respectively. Efforts were made to conduct boat surveys during periods when waves were 30 cm or less (Gould and Forsell 1989) during which the entire study area was

traversed. Surveys were randomly initiated between 0900 and 1400 hours, as weather and sea conditions permitted, to minimize the effects of light on raft (aggregation or flock of scoters) detection. Although significant nocturnal feeding by other waterfowl species has been documented (Nilsson 1969, Custer et al. 1996), Lewis et al. (2005) found surf scoters were primarily diurnal feeders (98% of observations); indicating daylight surveys were optimal in determining the location of foraging sites. It is worth noting that the purpose of the boat surveys was not to obtain specific population level estimates for surf scoters, but to determine the location of large numbers of feeding scoters. Therefore, concerns often associated with boat-based bird surveys (e.g., detection rates; Gould and Forsell 1989) did not apply to this study.

To minimize disturbance to ducks and reduce the risk of duplicate counting, boat surveys were conducted by moving once through the study area at a constant speed of approximately 40 km h^{-1} in a sinuous pattern parallel to the shore (Figure 5). The route of the sinusoidal pattern was based on the bathymetry of the area and included water ranging from 1.0 m to 13.7 m deep. Preliminary surveys of this area did not detect scoters outside of this range. A predetermined route was not followed for the survey, with the route being determined on-site by the location of scoters. When rafts were observed off route, boat trajectory was altered to determine the exact spatial location of scoter rafts using Global Positioning System (GPS). Two observers stood at the bow of the boat, each covering a 90° range, from the bow to the port or starboard beam (Mack et al. 2002). Numbers of scoter observed were recorded into a voice recorder and exact locations recorded into a handheld GPS device (corrected for travel speed). All observations were made with the unaided eye

and scoters flushed while moving through the survey area were counted, all other flying ducks were counted separately. A contract boatman experienced in the identification of waterfowl, located in the center of the boat behind the observers, acted as an independent observer and remained in constant communication with the primary observers regarding the location of scoter flocks.

Because static observations for raft locations could not be made, diving behavior observed during surveys was used as an indicator of active feeding. Typically, as the boat approached a raft at speed, scoters would begin to fly at a distance of approximately 300 – 400 m. Although scoters had dispersed before the boat passed over their location, disturbance to the water surface allowed for easy GPS identification of raft location. Occasionally, as the boat passed over the surface location of a raft (subsequent to the dispersal of ducks), a number of scoters could be observed surfacing. This surfacing behavior indicated that the ducks were diving (and thus presumably feeding) as the boat approached, even though surfacing scoters would quickly take to flight. Although diving has been noted in scoters for predator evasion, these observations have only been observed on molting grounds (when ducks are flightless) and flight seems the preferred evasion method throughout the remainder of the year. Additionally, as the boat approached rafts in which scoters could be observed surfacing, the distance from the boat at which the raft dispersed reduced (approximately <200 m), with scoters often flying directly around the boat. Guillemette (1994) found lower gut content mass in flying common eider (*Somateria mollissima*) than in raft feeding birds, suggesting a desire to limit flying immediately after feeding to reduce the energetic costs of flight. This suggests that scoters which

were reluctant to fly during this study were possibly either satiated or hesitant to leave a feeding site. Therefore, surfacing behavior and reluctance to disperse during surveys were noted as an indication of active feeding. For all other observations, scoters were considered to be engaging in non-feeding behaviors, such as loafing or sleeping.

Survey data from the GPS unit were downloaded to a computer, matched with the appropriate transcribed voice data, and imported into GIS (ArcGIS 9.1, ESRI Redlands, CA). Water depth and distance from shore for all downloaded points (individual rafts) surveyed were determined using electronic nautical charts. Occasionally, multiple points represented a single, large raft (typically numbering >2000 scoters). Because these large rafts occupy a broad spatial area, and possibly varying water depth, all points were considered separate for analysis. To determine water depth, digital coastal maps (NOAA 2001) of Chesapeake Bay were imported into GIS and displayed as raster images. To increase accuracy, only maps of 1:20,000 and 1:40,000 scales were used. Points were manually assigned depths based on proximity to displayed map values. For those points between values, the mean of nearby points was used. Distance from shore was determined through the shortest linear distance to shore, including marsh and island, using GIS.

Site Selection

Using GIS, data layers from all surveys were pooled and filtered for points with ≥ 100 surf scoters. Points were further filtered to identify flocks that were observed feeding. Based on these criteria, as well as repeated use of an area by scoters, three study sites with feeding scoters (Feeding sites) were chosen for habitat

analyses. In addition, three study sites were chosen at locations which did not contain either feeding scoters or appreciable numbers of loafing scoters (Non Feeding sites). These Non Feeding reference sites were located near, and at similar depths to, Feeding sites to minimize the effects of dissolved oxygen, depth, and salinity on faunal composition (Figure 6; Table 1). For analyses, the presence or absence of feeding scoters was considered the experimental treatment.

Habitat Composition

Sample Collection

For each of the selected study sites ($n = 6$), a 1-km² plot was centered using GIS. Within each plot, 10 random sampling locations were determined using a random point generator within Hawth's Analysis Tools for ArcGIS (Beyer 2004). Coordinates for each randomly determined point were loaded into a hand-held GPS for field location of sampling points.

At each sampling location, benthos were sampled ($n = 60$) using three grabs from a Petersen dredge (area = 593.5 cm²). To gain a broader perspective of benthic habitat across study sites, the boat was allowed to drift a maximum distance of 100 m from each GPS location during sampling. Contents of the three grabs were elutriated on-site through a 1.0-mm sieve and placed into bags and frozen until analysis. In addition, to gain a seasonal perspective of macrofaunal composition, habitat sampling was conducted in summer and fall 2005, and spring 2006. Fall sampling was conducted in late October and early November, in conjunction with the arrival of scoters on the Bay. Typically, scoters begin arriving on the Bay in late October, with most scoters having arrived by late November or early December. Although a few

scoters can be found on the Bay during the summer (likely crippled ducks from the previous hunting season), the vast majority of scoters depart the Bay by mid-April for their breeding grounds. Thus, to determine the level of depletion of macrofauna over the course of the winter, spring sampling was conducted in May, after the ducks had left the Bay. Winter sampling was not conducted due to logistical and weather constraints. Summer sampling was conducted when macrofauna were predicted to be at their most depleted due to possible summer hypoxia.

Sample Analysis

Frozen samples were chosen haphazardly, thawed, and analyzed using a 1-4X power dissecting microscope. Although scoters feed primarily on bivalves in Chesapeake Bay (Perry et al. 2007), all sampled organisms (including demersal fish) were included in this analysis since surf scoters have been shown to feed on non-bivalve organisms at other wintering locations, and thus other fauna represented potential prey. In addition, enumerating all sampled organisms allowed for comparisons of habitat characteristics (e.g., diversity indices). All macrofauna collected were sorted, identified to the lowest possible taxonomic level and counted. Bivalves were measured using digital calipers to the nearest 0.01 mm; hooked mussels along the posterior – anterior axis and all other bivalves laterally. Amphipod (Amphipoda) and polychaetes (Polychaeta) species were occasionally broken into multiple segments, either from field sampling or during analysis, requiring a standardized method for counting individuals. Therefore, density estimates were based on the presence of head as an indicator of an individual, although all body segments found were included in biomass measurements.

To determine faunal biomass, all individuals from a species were pooled for each sample and dried at 55° C to a constant mass, typically 24 – 36 hrs. Dry mass of each species was determined using an ACCULAB digital balance and measured to the nearest 0.001 g, burned in a muffle furnace for a minimum of six hrs at 550° C, and then measured to the nearest milligram. For each species within a sample, the difference between the burned mass and dry mass for a sample was used to determine ash-free dry mass (AFDM). Because several studies have suggested common eider feeding on blue mussel select specific size classes to minimize shell intake (Bustnes and Erikstad 1990, Bustnes 1998, Hamilton et al. 1999), the AFDM and ash content of the shell and tissue component were analyzed separately for all bivalves. Although these studies focused on the shell:tissue ratio of several size classes of blue mussel, the shell and tissue were examined separately to discern possible differences between bivalve species. For all bivalves, all removed tissue was from the shell before drying, including the adductor muscle, mantle, and siphon. In addition, encrusting bryozoans (primarily on hooked mussel) and sediment (infaunal bivalves) were removed from the shell by both manual removal and through flushing with fresh water before drying.

High densities of gemclam (all sampling periods) and bay barnacle (*Balanus improvisus*; spring sampling) warranted the use of sub-sampling procedures to determine density and biomass. Samples containing high densities of gemclam (>200) were homogenized (Figure 7) and the first 100 gemclam encountered were removed. To estimate the total number of gemclams in the sample, the volumetric proportion of 100 count gemclam to the total sorted sample volume was multiplied

against the total volume of the sample (unsorted + sorted). Sub-sampled gemclam individuals (100 counts) were analyzed identically to other bivalves and the total biomass of the shell and tissue portion was separately determined by the equation:

$$\hat{T}_s = N\bar{y}_s$$

where \hat{T}_s is the total estimated AFDM biomass or ash mass for species s , N is the total number of estimated individuals, and \bar{y}_s is the mean within season value (density or biomass) across sub-samples for species s (Thompson 2002). Because bay barnacles were not homogeneously distributed throughout individual samples, all barnacles were counted, and tissue and shell AFDM and ash mass were estimated using the above equation.

Sediment Analysis

Sediment elutriated from invertebrate collection during fall sampling was retained and pooled from each of the three grabs at a sampling location. Sediment was then homogenized, collected, and frozen until analysis. Samples containing little or no sediment (e.g., oyster beds) were noted and accounted for three of the 60 sampling locations.

Frozen samples were thawed, placed in aluminum weigh boats and dried at 55° C to a constant mass (typically 36 hrs) for grain size analysis. Approximately 5 – 6 g of the dried sample (calcareous fragments removed) was analyzed using procedures adapted from Poppe et al. (2000). Samples were mixed with 30 ml of a 5% solution of the anti-flocculent sodium hexametaphosphate. Samples were vortexed for a minimum of five minutes to aid in the mixture of the solution and the disaggregation of sediment particles. Vortexed samples were immediately wet sieved

over a 62-micron sieve (U.S. Standard 230) to separate the silt/clay and sand portion. The remaining portion was sieved over stacked 0.25 and 0.125 mm sieves (U.S. Standard 60 and 120, respectively). The contents of all sieves were placed into specimen containers and dried to a constant mass (typically 36 hrs). The silt/clay portion was dried to constant mass (typically five days) and mass corrected for the presence of sodium hexametaphosphate content. To correct for sodium hexametaphosphate, 30 ml of a 5% solution were placed into three specimen containers (mass predetermined) and dried to a constant mass, approximately five days. This resulted in a correction factor of 1.670 g (\pm 0.158) for the silt/clay fraction, with the silt fraction being calculated as:

$$\text{silt/clay (g)} = \text{sample (g)} - \text{container (g)} - 1.670\text{g}$$

To determine sediment organic content, 2 to 3 g of dried sediment sample had all calcareous material removed and placed into a tarred weigh boat. Sample mass was pre-determined then burned for a minimum of six hrs at 550° C. Organic content was then determined from the difference in mass from the ashed and dried sample mass.

Grab Efficiency Analysis

While use of a grab sampling device in marine benthic studies is widespread (e.g., Holland 1985), concerns exist over the reliability of this method over hard substrates (K. Paynter, University of Maryland, pers. comm.). Possible problems identified with grab sampling on hard substrates include loss of sample from inadequate closure of the grab (i.e., rocks or shells inhibiting full closure) and inadequate penetration of the substrate resulting in only partial sampling. Because a

large number of sampling locations in this study were found to consist of hard substrates, the results of grab sampling on hard substrates were compared with samples obtained by divers.

Approximately one week before the start of the spring sampling period, divers were used to sample benthos at two Feeding sites (Poplar Island and Herring Bay). The third Feeding site (Shady Side) did not contain hard substrate, and thus was not used. Within the two identified study sites, a diver sampled benthos at each sampling location used for grab sampling. An anchored float was placed at the location and divers, using SCUBA, placed a 0.33 m² quadrat at the base of the anchor. All material within the quadrat was collected and placed into a mesh bag lined with 0.25 mm screening. This was repeated two additional times at each location (triplicate sampling) by moving the quadrat an arms-length distance from each sample and placing the material into the same mesh bag. Samples were brought to the surface and frozen with the mesh lining until analysis. Frozen samples were thawed, removed from the mesh lining, and thoroughly elutriated over the same 1.0 mm sieve used for grab sampling. The elutriated sample was then either analyzed immediately using the same procedures outlined for grab sampling, or frozen for later analysis.

Data and Statistical Analyses

Scoter Distribution

Scoter locations were analyzed using all points recorded on surveys for water depth and distance from shore. While there is high certainty that rafts recorded as feeding were indeed feeding, it was not possible to state with absolute certainty that

rafts recorded as not feeding, were indeed not feeding. Therefore, all points were analyzed (including both feeding and loafing rafts) since the determination of feeding behavior was not conducted using static and sustained individual scoter or raft observations.

Habitat

Sediment composition (grain size and organic content) was compared between treatments using a paired t-test (PROC TTEST; SAS Institute 2003) and pairwise comparisons between sites, both across and within treatments, were conducted using a one-way mixed model ANOVA (PROC MIXED). Analyses of sediment, as well as all statistical tests used in this study were considered significant with $\alpha < 0.05$. Five grams of sediment could not be obtained at all sampling locations; therefore dried values were standardized by calculating the proportion of the original mass represented by the dried mass for each grain size. This procedure was repeated for organic content as well. All sediment variables were square root transformed (\sqrt{x}) to meet the assumptions of normality and homogeneity of variances. Additionally, the two largest grain sizes, >0.25 and 0.125 mm (fine sand and larger), and the two smallest grain sizes, <0.125 and 0.062 mm (very fine sand and the silt/clay fraction), were grouped into two variables. This grouping was done to allow for the examination of broad sediment classifications and because of high within site variability of grain size proportions. These broad sediment variables were compared across treatments and sites using the same procedures outlined previously and were also used in the determination of habitat types for each location.

Benthic invertebrates are often closely associated with substrate composition of a given region (Holland and Dean 1977) and substrate is the typical metric through which habitat classifications are made in marine/estuarine systems. Therefore, to examine the effects of benthic habitat on invertebrate and scoter distribution, sampling locations were categorized into either hard, sand, or mud/silt habitats. Sampling locations were considered hard bottom if they consisted primarily of oyster (live and fossil shell), rock, or packed clay. All other samples were considered soft-bottom. For the purposes of this study, soft-bottom habitats were categorized based on the results of the grouped grain-size variables described previously. Locations consisting of >50% fine sand or larger grain sizes were considered sand habitats and locations consisting of >50% very fine sand or smaller grain sizes were considered mud habitats.

In addition to broad habitat classification, the small-scale variability (patchiness) of these habitats was determined in each study site. To estimate patchiness, the number of habitats represented in each pooled triplicate sample was quantified *post hoc* based on analysis of field notes and samples. The number of habitats was averaged across seasons for each location. No location contained more than two habitats, therefore, mean values of two indicate highly patchy habitats while a mean value of one indicates homogenous habitat. Differences between treatments and sites for both habitat and patchiness values were examined using chi-squared test (PROC FREQ).

Faunal Composition

Faunal composition between and within treatments was compared using a suite of broad, categorical, and species specific variables. Shannon-Weiner diversity and species richness indexes are broad ecological variables commonly employed in marine/estuarine systems as measures of community structure (Boesch 1972, Mountford et al. 1977, Drake and Arias 1997). Shannon-Weiner diversity indices were calculated for all locations for each sampling season using the equation:

$$\hat{H}' = -\sum_{i=1}^s \hat{p}_i \ln \hat{p}_i$$

where \hat{H}' is the sample diversity index and \hat{p}_i is the proportion of the i^{th} species in the sample. In addition, species richness (S) of each location was quantified within a season as the total number of faunal species present. An organism not able to be identified to the species level (e.g., *Polychaete* spp.) was included in calculations if it could be reasonably surmised that it was not already represented in the sample. Both diversity indices (\hat{H}' and S) were analyzed using a paired t-test (PROC TTEST) to detect differences between treatments and a one way mixed model ANOVA (PROC MIXED) to determine the effects of season, site, and habitat type. Additionally, a mixed model ANOVA was used to examine differences in diversity indices between categorized habitat types.

All density and biomass variables were converted to m^{-2} by correcting for grab sampling area and dividing by three to account for triplicate sampling. Preliminary analyses of density and biomass data for several faunal species using standard parametric statistical tests indicated violations of normality and equality of variances.

Standard data transformation techniques reduced the severity of deviation from these assumptions and, in some cases, corrected for them. However, this correction was not uniform across all faunal species and assemblages. Therefore, a nonparametric two-sample randomization test (resampling without replacement) was used to conduct pair-wise comparisons of faunal variables between treatments, sites, habitats, and seasons (Butler 2001). The primary advantage of using a randomization, or permutation, test is that meeting the assumptions of normality and homogeneity of variances, used for ANOVA and t-test, is not required (Manly 1997). Manly (1997) does discuss limitations in using two-sample randomization tests with data containing unequal variances, suggesting that the test may detect differences in variance rather than differences between sample means. However, this analysis uses a procedure that does not pool variances among treatments, and thus is appropriate for comparing means when equal variance cannot be assumed (Butler 2001). Randomization tests were conducted using the macro “twosampleran” developed for MINITAB (1999) by Centre for Ecology and Hydrology (CEH; Monks Wood, United Kingdom) to test for differences between sample means. Each pair-wise test used 999 randomizations (resampling) of the data to minimize Monte Carlo variation when calculating the test statistic.

Analysis of faunal assemblage variables included total (sum of all fauna present), infauna, and epifauna. Epifauna was defined as those species which spend the majority of their life history on the surface of the substrate and included hooked mussel, bay barnacle, anemones (except the burrowing anemone, *Edwardsia elegans*), the tube-building amphipod (*Corophium lacustre*) decapods, isopods,

gastropods, and demersal fish (Rodney and Paynter 2006). Infaunal species were defined as those which spend the majority of their life history below the sediment surface and included clams, polychaetes, and amphipods (except the tube-building amphipod). Species specific comparisons were examined on all bivalves and those species which constituted a relatively significant portion of sampled fauna.

To minimize the number of pair-wise comparisons with the randomization method, the Pearson correlation coefficient for density and biomass was calculated for all faunal species and assemblages. This test was based on the assumption that a linear relationship exists between density and biomass (i.e., as density increases, biomass increases). Correlation coefficients were calculated using a randomization macro for Minitab (999 randomizations) obtained from CEH and the results of this test indicated significant correlation between density and biomass for all faunal variables (Table 2). Based on these results, pairwise comparisons between treatments were only completed on density variables for each species.

Bivalve shell lengths were placed into 2 mm bins for histogram development and compared across season and treatment for each species. Gemclam was excluded from shell length analysis due to their small size (typically < 4.0 mm).

Grab - Diver Comparisons

To compare the grab sampling technique with samples acquired by divers, similar analyses as stated above for faunal composition were repeated for diver obtained samples. However, strong variability in data and high patchiness between samples from the same locations yielded statistical results of little meaningful value.

Therefore, only qualitative comparisons of frequency of occurrence and density were made between the two sampling techniques.

Faunal Energetic Content

Energetic content of sampled bivalves was comparatively examined across species using two variables. To determine whether scoters were selecting food items based on the shell minimization theory (Bustnes and Erikstad 1990, Bustnes 1998, Hamilton et al. 1999), the ratio of total shell mass (AFDM + ash mass) to tissue AFDM (hereby shell: tissue) was calculated for all bivalve species, except softshell clam, in addition to the ratio of total ash (shell + tissue) content to total AFDM. Softshell clam was excluded from this analysis because of a lack in individuals sampled in summer and fall. Shell to tissue AFDM mass and total ash to AFDM was compared within species to determine effects of season and location and across species to determine species differences using one-way mixed model ANOVA (PROC MIXED). Data were log transformed [$\log(x + 1)$] to meet assumptions of normality and heteroscedasticity of variances, when needed. Results for all other fauna are presented in Appendix 1. Mean values for all variables analyzed in this study, including faunal energetic content, are presented as the mean \pm the standard error of the mean where appropriate.

Institutional Animal Care and Use Committees

This project was conducted under the auspices and approval of the University of Maryland Institutional Animal Care and Use Committee permit number R-05-11 as well as under United States Geological Survey Patuxent Wildlife Research Center Animal Care and Use Committee approval.

RESULTS

Distribution of Surf Scoters

Seven species of waterfowl were observed in the study area during the winter of 2004 – 2005, including surf scoter, black scoter, white-winged scoter, long-tailed duck (*Clangula hyemalis*), bufflehead (*Bucephala albeoa*), canvasback, and scaup (*Aythya* spp.). Canvasback and scaup observations were limited to extreme near-shore waters, whereas all scoters and long-tailed duck were located primarily offshore. Buffleheads were typically found farther offshore than canvasback and scaup, but did not extend to deeper waters occupied by scoters. Notably absent from all surveys was common goldeneye (*Bucephala clangula*), which is typically a common species in this portion of Chesapeake Bay (D. Kidwell, pers. observation).

Surf scoters and long-tailed ducks were the most frequently encountered waterfowl observed during surveys. Surf scoters were the most numerous with an average of 5,682 (SE = 2,580) individuals over the 4 surveys (Table 3). However, the total number of surf scoters observed was largely dependent on the presence or absence of large rafts (>1000 scoters). The high standard error of surf scoter numbers was largely due to the absence of these large scoter rafts during the December and April surveys (Table 3). Surf scoter raft size ranged from 12 – 10,000 (some individual scoters were observed), however, the majority of surf scoters occurred in a small number of rafts. A single raft in February (out of 25) accounted for 92% of observations, whereas four rafts (out of 38) in March accounted for 64% of observations. This contrasted with long-tailed duck numbers, which had a comparatively low standard error ($\bar{x} = 3,287 \pm 128$) across surveys and tended to be

more spatially dispersed and in smaller rafts (range 1 – 1000 birds per raft) than surf scoters ($\bar{x} = 188 \pm 96$ birds per raft).

Rafts of surf scoters and long-tailed ducks occasionally occurred in close proximity, especially at Poplar Island, but there was typically little mixing between species. White-winged scoters (Table 3) occurred in small numbers, but also did not typically mix with surf scoters. Black scoters were found in very small numbers, and were found to mix with surf scoter rafts. Although black scoters were difficult to separate from surf scoters, they comprised only a negligible number of the overall scoter population, an observation also noted by M. C. Perry (pers. comm.).

Surf scoter rafts were widely distributed throughout the study area, although feeding scoters typically concentrated in specific areas. The study site centered off the southeastern portion of Herring Bay contained the largest numbers of surf scoters, with feeding rafts in excess of 10,000 and 3,000 during February and March, respectively. The northwestern waters off Poplar Island routinely contained large numbers of surf scoters, as well as long-tailed ducks and white-winged scoters. Additional feeding surf scoters were found in the waters off Shady Side, including the Shady Side Feeding site, especially during the March and April surveys. The western shore of Tilghman Island also contained rafts of feeding scoters, although overall numbers were lower than the previously described sites. Additional locations of feeding scoters were observed, however, the numbers of rafts and individuals were comparatively low and specific locations inconsistent across surveys. The remaining scoter observations were likely scoters engaging in other behaviors (e.g., loafing), although it is possible these scoters were feeding. One complicating factor during the

December survey was the presence of guided seaduck hunting boats, which have increased in number on the Bay (Perry and Deller 1994). Approximately five boats were scattered along the west-northwest portion of Poplar Island (including the Poplar Island Feeding site), and likely played a role in the lower numbers of scoters counted during that survey (Table 3). No hunting boats were located at Herring Bay during the December survey. However, boats were observed at this location on other occasions and may have contributed to reduced scoter numbers during December.

No seasonal pattern in water depth of scoter rafts was observed across the survey period (Figure 8). Mean depth of observations across surveys was 4.0 (SE = 0.18) m with over 96% of scoters counted occurring at depths between 1.5 and 5.5 m. The greatest depth observed occurred while crossing the Bay during the March survey, when 96 scoters were observed in waters 11.6 – 15.2 m in depth. Conversely, the shallowest observation was recorded over waters 0.6 m in depth.

As with water depth, no seasonal pattern in distance from shore was found for scoter rafts throughout the survey period (Figure 9). Mean distance from shore of rafts was 1.4 (SE = 0.10) km, with over 97% of scoters counted being found between 0.5 and 3.0 km from shore. As would be expected, the maximum observed distance from shore was from the same 96 scoters observed in deeper water (mentioned above), which were found between 6.5 and 8.6 km from shore. Minimum observed distance from shore was 0.2 km, although it is likely that a few scoters closer to shore were not observed due to the inability of the survey boat to move into shallow water.

Habitat Composition

Sediment Grain-size and Organic Content

Analysis of grain-size distribution indicated large treatment effects, with the largest grain-sizes associated with Feeding sites and the smallest grain-sizes associated with Non Feeding sites (Figure 10). The largest grain-size analyzed (0.250 mm) comprised more than 68% of sediment size spectra at Feeding sites, which was more than 3-times higher ($t = 8.3$, $P < 0.001$) than the proportion found at Non Feeding sites. Conversely, the smallest grain-size (silt/clay; < 0.062 mm) fraction at Non Feeding sites (33% of sediment size spectra) was more than twice the proportion found at Feeding sites ($t = -4.9$, $P < 0.001$). Of the middle grain-sizes, the 0.125 mm – 0.062 mm fraction was significantly higher ($t = -8.6$, $P < 0.001$) at Non Feeding sites (comprising 66% of the sediment size spectra) relative to Feeding sites (18% of sediment size spectra; Table 4). The 0.250 mm – 0.125 mm grain-size portion was the only grain size where no difference between treatments was found ($t = 1.6$, $P = 0.122$). Differences in sediment grain-size content between treatments were even more apparent when pooled. Pooled proportions of the two largest grain-sizes (contents of 0.250 and 0.125 mm sieves) within the Feeding treatment were more than twice that of Non Feeding sites ($t = 9.5$, $P = 0.002$), whereas the proportion of the two smallest grain-sizes (contents of 0.062 mm and silt/clay) in the Non Feeding treatment were over 6-times that found at Feeding sites ($t = -9.5$, $P = 0.002$; Figure 10).

Overall sediment organic content was significantly higher at Non Feeding sites than Feeding sites ($t = -4.2$, $P < 0.001$). Although Non Feeding sites contained

over 54% more organic content, the mean proportion of organic content was relatively low for both treatments. Site effects were found for both Feeding ($F_{2, 23} = 4.2$, $P = 0.028$) and Non Feeding sites ($F_{2, 27} = 7.6$, $P < 0.001$). At Feeding sites, Shady Side sediment contained the lowest proportion of organic content, which was significantly lower than Herring Bay ($t = 2.5$, $P = 0.050$) but not Poplar Island ($t = 2.4$, $P = 0.060$). Conversely, sediments at the Poplar Island Non Feeding site contained the lowest organic content, although this difference was only significantly lower than Herring Bay ($t = 2.8$, $P = 0.012$). Lower organic values at the Poplar Island Non Feeding and the Shady Side Feeding sites were primarily due to the high proportion of sand habitats (Shady Side entirely so) found at those sites. Sand habitats (pooled across treatments and sites) contained lower organic content ($\bar{x} = 0.009 \pm 0.001$) than both hard ($t = 5.4$, $P < 0.001$) and mud habitats ($t = -5.4$, $P < 0.001$). No difference in organic content was detected between hard and mud habitats ($t = -0.4$, $P = 0.943$).

Habitat Categorization

Habitat at Feeding sites varied between hard and sand substrates, with only one sampling location containing a small amount of mud/silt (Figure 11). Conversely, Non Feeding sites contained a large proportion of mud/silt habitat as well as a mixture of sand and hard substrates. Overall, habitat at Feeding sites was significantly different than that found at Non Feeding sites ($\chi^2 = 85.9$, $P < 0.001$; Figure 11). Within Feeding sites, Herring Bay consisted mostly of hard bottom with a small amount of sand habitat. Hard bottom at Herring Bay was characterized by hard packed clay with either scattered large rocks and debris (70%) or scattered fossil

oyster shell (10%). Poplar Island was similar to Herring Bay in that it consisted of a mixture of hard bottom and sand. However, hard bottom at Poplar Island was characterized entirely by degraded oyster bar consisting of fossil oyster shell and live oyster. A portion of the Poplar Island study site was located within a Maryland Department of Natural Resources oyster reserve. In contrast to Herring Bay and Poplar Island, Shady Side contained no hard bottom and consisted nearly entirely of sand habitats. The small amount of mud/silt habitat found within the Feeding Shady Side site comprised only a portion of one sampling location located at the edge of the site.

Non Feeding sites contained a mixture of mud/silt, sand, and hard habitats. Herring Bay habitats were comprised of mostly mud, but three locations were comprised of hard bottom. Hard bottom composition at this site was similar to the Herring Bay Feeding site, and was comprised of packed clay. However, no sites at Herring Bay Non Feeding contained fossil oyster shell. The Poplar Island Non Feeding site contained the greatest variety of habitats of all sites sampled; containing a mixture of hard, sand, and mud/silt habitats (Figure 11). Hard substrate at the Poplar Island Non Feeding site was composed entirely of degraded oyster bar, also with a mix of fossil oyster shell and live oysters. However, unlike the Poplar Island Feeding site, the Poplar Island Non Feeding site was not located within an oyster sanctuary or reserve. The Shady Side Non Feeding site did not contain hard bottom and consisted entirely of mud/silt bottom.

No difference was found between treatments for degree of habitat patchiness ($\chi^2 = 0.3$, $P = 0.587$). At Feeding sites, 10% of Herring Bay samples contained more

than one habitat type compared to 43% and 7% at Poplar Island and Shady Side, respectively. The Poplar Island Non Feeding site contained the highest degree of patchiness, with 53% of samples containing two habitat types, whereas the Shady Side Non Feeding site was entirely homogenous. Heterogeneity of habitats was closely associated with samples containing oyster shell at both Poplar Island sites. However, oyster shell habitat at Feeding sites was interspersed with sand substrates, while oyster shell at Non Feeding sites was interspersed with mud.

Faunal Composition

A total of 137,785 organisms was sampled over the course of three sampling seasons (summer, fall, and spring), representing 40 species (identified to genre or species) in 12 faunal groups (Table 5). However, samples were dominated by gemclam (130,623 individuals) from eight samples representing three locations. Exclusion of these values yielded a gemclam-adjusted total of 6,162 organisms. Of the 40 identified species, seven were found once and another eight organisms were found in <10% of samples. While occurring in differing proportions, the majority of species sampled were found in both treatments; ten unique to Feeding sites and three unique to Non Feeding sites (Table 5). Mitchell macoma was the most frequently occurring species within both treatments.

Across seasons and sites, species richness (S) was highest at Non Feeding sites ($t = -3.8$, $P < 0.001$), averaging >1 more species per sample than Feeding sites. Significant site effects were found within the Feeding treatment ($F_{2, 87} = 7.8$, $P < 0.001$), but not between Non Feeding sites ($F_{2, 87} = 2.9$, $P = 0.062$). Within the Feeding treatment, Poplar Island contained nearly 65% more species per sample than

Herring Bay ($t = -3.2$, $P = 0.002$) and Shady Side ($t = -3.7$, $P < 0.001$). Overall, Poplar Island in both treatments averaged more species per sample ($\bar{S} = 7.0 \pm 0.6$ and $\bar{S} = 7.9 \pm 0.57$ for Feeding and Non Feeding sites, respectively) than Herring Bay and Shady Side (Table 6). High species richness values at Poplar Island were likely related to high heterogeneity of habitat, allowing for species representative of both soft and hard substrates. Richness values were related to substrate present ($F_{2, 177} = 3.6$, $P = 0.031$), with mud/silt containing the highest values ($\bar{S} = 6.9 \pm 0.40$), likely a result of numerous species of polychaetes present, and sand containing the lowest ($\bar{S} = 5.6 \pm 0.37$). This difference was statistically significant ($t = 2.6$, $P = 0.027$), while differences in richness between hard ($\bar{S} = 5.9 \pm 0.41$) and mud ($t = -1.8$, $P = 0.166$) and hard and sand habitats ($t = -0.7$, $P = 0.493$) were not. As would be expected following late-fall and/or early-spring invertebrate recruitment periods, richness values were highest during the spring for both treatments and all sites.

Similarly to species richness, Shannon-Weiner diversity indices (H') for the Non Feeding treatment were significantly higher than the Feeding treatment ($t = -3.8$, $P < 0.001$) across seasons. However, much of the difference between treatments can be attributed to differences in H' between habitat types. Mud/silt averaged higher diversity values ($\bar{H}' = 1.2 \pm 0.07$) compared to hard ($\bar{H}' = 0.9 \pm 0.07$) and sand ($\bar{H}' = 0.9 \pm 0.06$) substrates, although mud was only statistically higher than sand ($t = -3.84$, $P = 0.010$). Additionally, the lowest H' value, found at the Poplar Island Feeding site, can also be attributed to disproportionately high densities of gemclam. No within treatment differences between sites or seasons were found for the diversity index (Table 6).

Because gemclam represented a disproportionately large portion of fauna sampled and was highly local in distribution (approximately 95% of all fauna was gemclam from only 8 samples out of 180), all relevant faunal analyses were conducted on gemclam-adjusted values. Further, it is felt that this technique was justified since the portion of surf scoter diet represented by gemclam found by Perry et al. (2007) is likely inflated. At the three locations with high densities of gemclam, the volume of deceased gemclam shell was often double that of live clams, suggesting that feeding on this abundant, but small clam would likely require the ingestion of large quantities of remnant shell. Ingestion of large quantities of remnant shell relative to live organisms would likely inflate the actual proportion of gemclam in food habits analyses.

Lowest total faunal densities (gemclam-adjusted) at Feeding sites were observed during the summer ($\bar{x} = 253.5 \pm 79.91$ ind. m^{-2}), while lowest values at Non Feeding sites were found during the fall ($\bar{x} = 196.5 \pm 22.30$ ind. m^{-2}). Although faunal density increased throughout the study period for Feeding sites (Figure 12), both treatments had significant increases ($t = -1.9$, $P = 0.048$ and $t = -7.58$, $P = 0.002$; for Feeding and Non Feeding sites, respectively) between the fall and spring sampling period. Feeding site faunal density increased from a mean of 337.2 (SE = 127.92) m^{-2} in fall to 832.0 (SE = 231.81) ind. m^{-2} in spring, while Non Feeding sites increased from a mean density of 196.5 (SE = 22.34) ind. m^{-2} in the fall to 1,381.7 (SE = 154.82) ind. m^{-2} in the spring. This increase between fall and spring was likely the result of a successful late-fall/winter larval set and subsequent growth. The only significant difference in total faunal density between treatments was observed during

the spring season, with Non Feeding sites having significantly higher faunal densities ($t = -1.9$, $P = 0.044$) than Feeding sites.

Epifauna

Total epifauna was strongly associated with hard bottom habitats, averaging 648.6 (SE = 107.6) ind. m⁻² compared to a mean of 35.3 (SE = 32.0) ind. m⁻² in soft bottom (sand plus mud/silt) habitats ($t = 4.4$, $P = 0.002$). Much of the epifauna found in habitats categorized as soft bottom consisted of bay barnacle (*Balanus improvisus*) attached to shell fragments of softshell clam or dwarf surfclam. Overall, densities of epifaunal species were significantly higher at Feeding sites than Non Feeding sites ($t = 2.1$, $P = 0.040$), with Feeding sites containing nearly three times the density of Non Feeding sites ($\bar{x} = 329 \pm 89.6$ and 116 ± 44.3 ind. m⁻², respectively).

A total of 16 epifaunal species representing 10 functional groups was sampled during the study period and included species associated with a variety of habitats (e.g., sand or hard bottom) or specific habitats (e.g., hard bottom or oyster bar).

Three isopod species (*Cyathura polita*, *Edotea trioloba*, and *Chiridotea almyra*) were found within both treatments, and were primarily associated with soft substrates.

Also sampled in association with soft substrates from both treatments were blue crab (*Callinectes sapidus*), mysid shrimp (*Americamysis bigelowi*), sand shrimp (*Crangon septimspinosa*), and the dusky pipefish (*Syngnathus floridae*). Two gastropod species (*Sayella chesapeakea* and *Boonea* sp.) were found in both sand and oyster samples.

However, none of these species occurred with adequate frequency or density to warrant detailed analyses.

The remaining epifauna sampled were typically found in association with hard bottoms, especially oyster bars. These included two species of decapod, the white-fingered mud crab (*Rhithropanopeus harrisi*) and the flat mud crab (*Eurypanopeus depressus*), and two species of demersal fish, the skilletfish (*Gobiesox strumosus*) and the naked goby (*Gobiosoma bosc*). Both species of mud crab and the naked goby were found on oyster bars within both treatments, however, a single skilletfish was unique to the Feeding treatment. As with the previous epifaunal species, these hard bottom species did not occur in sufficient quantities to warrant detailed analyses. However, hooked mussel, bay barnacle, anemones, and the slender-tube building amphipod (*Corophium lacustre*) did occur in sufficient quantities and/or frequency for detailed analyses, and are presented below.

Hooked Mussel

As with total epifauna, densities of hooked mussel were strongly related to habitat type ($t = 6.24$, $P = 0.002$), with hard bottom averaging 26.6 (SE = 4.18) ind. m⁻² compared to a mean of 0.43 (SE = 0.30) ind. m⁻² in soft bottom habitat. Within hard bottoms, hooked mussel was the dominant bivalve species in terms of ash-free dry mass (AFDM; $\bar{x} = 1,295 \pm 3.75$ mg m⁻²), but not density (gemclam averaged higher densities due to the presence of sand with oyster bars at Poplar Island).

No difference between treatments in overall mussel density was detected ($t = 1.4$, $P = 0.168$; Figure 13). The inability of the randomization test to detect a treatment effect was likely due to high standard errors of treatment means (26.0 and 19.01 for Feeding and Non Feeding sites, respectively) as a result of a patchy habitat distribution. Excluding sites at Shady Side, no within treatment difference was found

in hooked mussel density between Poplar Island and Herring Bay (Figure 14). Seasonally, hooked mussel densities at Feeding sites in summer averaged 14.5 (SE = 5.01) ind. m⁻² compared to 25.8 (SE = 7.55) ind. m⁻² in the fall ($t = -1.2$, $P = 0.310$). Between fall and spring, hooked mussel densities at Feeding sites significantly decreased ($t = 2.3$, $P = 0.028$) to a mean of 6.2 (SE = 3.59) ind. m⁻² (Figure 13). However, this seasonal pattern contrasted with hooked mussel densities at Non Feeding sites, which averaged 4.4 (SE = 2.33) ind. m⁻² in summer and 12.5 (SE = 4.80) in fall ($t = 1.5$, $P = 0.174$). But unlike Feeding sites, densities did not decrease between fall and spring ($t = 0.08$, $P = 1.000$), but remained stable between seasons ($\bar{x} = 12.5 \pm 4.83$ and $\bar{x} = 11.95 \pm 5.15$ ind. m⁻², respectively; Figure 13). Removal of the spring sampling from treatment comparisons, to account for seasonal differences between treatments, indicated higher densities of hooked mussel at Feeding sites ($\bar{x} = 20.2 \pm 4.56$) relative to Non Feeding sites ($\bar{x} = 8.6 \pm 2.74$; $t = 2.17$, $P = 0.040$).

In addition to density, differences in seasonal patterns in hooked mussel shell length distribution were found between Feeding and Non Feeding sites. Within the Feeding treatment (pooled across sites), mean shell length increased from summer ($\bar{x} = 9.8 \pm 1.22$ mm) to fall ($\bar{x} = 13.5 \pm 0.70$ mm), likely representing the growth of an individual age cohort (Figure 15). However, this age class was not detected during spring sampling, when mean shell length was 8.3 mm (SE = 1.07). Conversely, shell length at Non Feeding sites was similar between summer to fall ($\bar{x} = 10.9 \pm 1.85$ mm and $\bar{x} = 8.8 \pm 0.62$ mm, respectively) before increasing over winter to 12.71 mm (SE = 1.33). Individual age classes were not distinguishable between summer and fall,

but the increase in shell length over winter likely represents the growth of the fall cohort and possible emergence of a new age class (Figure 15).

Bay Barnacle

Barnacles were the most frequently encountered epifaunal species during the study, occurring in 38% of Feeding, and 22% of Non Feeding samples (Table 5). Consistent with results by Kennedy and DiCosimo (1983), bay barnacle was the only barnacle species identified within the study area. However, because not all barnacles were dissected and speciated, this does not preclude the presence of an aberrant barnacle species. Densities of barnacle were significantly higher ($t = 2.4$, $P = 0.012$) in hard bottom habitats ($\bar{x} = 363.0 \pm 116.20$ ind. m^{-2}) than soft bottoms ($\bar{x} = 60.9 \pm 50.51$ ind. m^{-2}). Overall densities were not found to be significantly different between treatments ($t = 1.6$, $P = 0.108$). As with hooked mussel, the inability of the randomization test to detect statistical differences was likely due to high standard errors associated with treatment means. However, in addition to high patchiness, much of the overall variability in barnacle densities was associated with strong seasonal effects (Figure 16). Seasonally at Feeding sites, mean barnacle density between summer and fall remained largely unchanged. However, a large late-fall/over-winter recruitment pulse resulted in significantly higher ($t = -2.5$, $P = 0.004$) barnacle densities in spring ($\bar{x} = 791.8 \pm 296.72$ ind. m^{-2}). A similar pattern was observed at Non Feeding sites, where very few summer and fall barnacles preceded large densities in spring (Figure 16). Within both treatments young barnacles were extremely pervasive in spring, and were found not only on hard substrates, but also on any hard objects (i.e., atop shell fragments) available on soft substrates, including live

dwarf surfclam shells. Within treatments, no difference in barnacle density was found between Herring Bay and Poplar Island (no barnacles were found at Shady Side) at Feeding sites ($t = 0.4$, $P = 1.000$), or Non Feeding sites ($t = -1.49$, $P = 0.272$; Figure 17).

Anemones

Three anemone species were identified within the study area and included the epifaunal white (*Diadumene leucolena*) and green-striped (*Haliplanella lineate*) anemones as well as the infaunal burrowing anemone (*Edwardsia elegans*). Together, both epifaunal anemone species were strongly associated with hard bottom habitats ($t = 3.4$, $P = 0.002$). However, only the white anemone was found in sufficient quantities to warrant quantitative analysis, therefore the following anemone analyses refer to white anemone.

Anemones were found in significantly higher densities on hard substrates ($\bar{x} = 26.9 \pm 7.18$ ind. m^{-2}), and as with barnacles, the presence of anemones on soft substrates reflected individuals found on shell fragments. Overall densities of anemones were higher at Feeding sites ($t = 1.9$, $P = 0.042$), which averaged close to three times more anemones m^{-2} (Figure 18). Unlike hooked mussel and bay barnacle, which were common at both Herring Bay and Poplar Island sites, anemones were found almost exclusively at Poplar Island ($t = 3.2$, $P = 0.002$; Figure 19). Seasonal differences in anemone densities varied among treatments. Similar to hooked mussel densities at Feeding sites, anemones exhibited an increase in density from summer to fall, followed by an overwinter decline (Figure 18). Conversely, anemone densities at Non Feeding sites increased throughout the study period (Figure 18).

Slender Tube-building Amphipod

Although the slender tube-building amphipod had average densities of over 156 (SE = 61.9) ind. m⁻² at Feeding sites, and 52.8 (SE = 27.8) ind. m⁻² at Non Feeding sites ($t = 0.80$, $P = 1.000$), much of the densities of the slender tube-building amphipods were driven by a few number of sites. Within both treatments, Herring Bay contained significantly more slender tube-building amphipods than Poplar Island ($t = 2.6$, $P = 0.004$ and $t = 1.9$, $P = 0.006$ for Feeding and Non Feeding sites, respectively). No slender tube-building amphipods were found at Shady Side. Within Feeding sites, over 99% of the total numbers of slender tube-building amphipods sampled were recorded at four locations at Herring Bay. Similarly, at Non Feeding sites, over 99% of the slender tube-building amphipods sampled were found at two locations at Herring Bay. These small numbers of locations which represented the majority of amphipods sampled precluded the analysis of seasonal trends in density.

Infauna

As with analyses of total fauna, analyses on infaunal invertebrates were conducted using gemclam-adjusted values. Total infauna densities were strongly associated with mud/silt habitats, containing significantly higher densities than both hard bottoms ($t = -5.8$, $P = 0.002$) and sand substrates ($t = -5.5$, $P = 0.002$). Likely due to the patchiness of sand and hard habitats at several sampling locations, no differences were found between sand and hard bottoms ($t = -0.9$, $P = 0.344$). Overall between treatments, Non Feeding sites contained over four times the density of infauna as Feeding sites ($t = -5.0$, $P = 0.002$). Both treatments exhibited similar

seasonal patterns, with no change in densities between summer and fall before increasing significantly over winter ($t = -3.4$, $P = 0.004$ and $t = -7.2$, $P = 0.002$ for Feeding and Non Feeding sites, respectively). As with several epifaunal species, this overwinter increase was largely due to a successful late-fall/winter spawning and larval settlement. Within Feeding sites, Herring Bay averaged approximately 50% less infauna m^{-2} ($\bar{x} = 89.5 \pm 19.55$) than both Poplar Island ($t = -3.44$, $P = 0.004$) and Shady Side ($t = -3.4$, $P = 0.004$), while no difference was found between Poplar Island and Shady Side ($t = 0.5$, $P = 1.000$). Conversely, at Non Feeding sites, no differences in infaunal density were detected between sites.

Twenty-four infaunal species representing 5 functional groups were sampled throughout the study period. Several species did not occur in high enough densities to warrant detailed analyses, and included the burrowing anemone, red ribbon worm (*Micura leidyi*), unknown nemertean (Nemeritan sp.), and several amphipod species. The remaining species that were analyzed specifically included dwarf surfclam, gemclam, mitchell macoma, Baltic macoma, and softshell clam, and common clamworm (*Neanthes succinea*).

Dwarf Surfclam

Dwarf surfclam was frequently sampled at both Feeding (48%) and Non Feeding (58%) sites. Dwarf surfclam had its highest densities in mud ($\bar{x} = 51.0 \pm 11.72$ ind. m^{-2}) and sand habitats ($\bar{x} = 38.5 \pm 11.14$ ind. m^{-2} ; $t = -0.8$, $P = 0.436$). Densities in soft substrates (sand and mud) were significantly higher than those in hard habitats ($t = -4.8$, $P = 0.002$). Differences between treatments were not found ($t = -0.7$, $P = 1.000$), with Feeding sites averaging 28.3 (SE = 8.53) ind. m^{-2} and Non

Feeding sites averaging 36.5 (SE = 7.89) ind. m⁻². The lack of difference between treatments was likely due to the propensity of dwarf surfclam to occur in sand and mud habitats.

Seasonally, dwarf surfclam was largely absent from summer and fall samples, with a combined averaged of 4.6 (SE = 0.74) ind. m⁻², with a maximum density of only 56 ind. m⁻². However, significant increases over winter (Figure 20) were found at both Feeding ($t = -3.1$, $P = 0.002$) and Non Feeding ($t = -4.9$, $P = 0.002$) sites, which averaged 75.9 (SE = 23.40) and 98.7 (SE = 19.15) ind. m⁻², respectively, in spring. A maximum density of 511 ind. m⁻² was found at the Shady Side Feeding site in spring. This pattern is consistent with that of other fauna, with a successful winter larval settlement and juvenile growth. However, this explosion in dwarf surfclam densities was not uniform across all sites (Figure 21), as the Herring Bay Feeding site averaged just 2.1 (SE = 0.94) ind. m⁻², which was significantly fewer than both Poplar Island ($t = 2.0$, $P = 0.002$) and Shady Side ($t = -2.76$, $P = 0.004$). No differences between Poplar Island and Shady Side were found for either Feeding ($t = 0.3$, $P = 1.000$) or Non Feeding ($t = -0.4$, $P = 1.000$) treatments.

Unlike hooked mussel, no treatment differences in seasonal shell length were found for dwarf surfclam (Figure 22). During summer, the 10 mm size class was the most prevalent for both treatments, with the distribution skewed towards smaller shell lengths. This age cohort was found to increase in size during fall sampling, skewing towards larger shell lengths, although the most common size class at Feeding sites was 12 mm compared to 10 mm at Non Feeding sites. A new age cohort, consistent

with a successful over winter larval set and growth, was observed in spring, with the 8 mm size class the most common for both treatments.

Baltic Macoma

Baltic macoma (hereafter called macoma) was common at Non Feeding sites (79% of samples), and the difference between treatments for bivalves was most apparent for this species. In terms of habitat, macoma was strongly associated with mud/silt habitats, occurring in significantly higher densities than in both sand ($t = 4.9$, $P = 0.002$) and hard bottoms ($t = 5.5$, $P = 0.002$). This habitat difference is the likely driver behind the overall difference in macoma densities between treatments ($t = -5.2$, $P = 0.002$), with an average of 24.7 (SE = 10.51) ind. m⁻² at Feeding sites and 292.1 (SE = 50.49) ind. m⁻² at Non Feeding sites.

At Feeding sites between summer and fall, macoma was found in low densities, with a combined average of only 1.1 (SE = 0.44) ind. m⁻². Comparatively, macoma densities at Non Feeding sites declined ($t = 3.10$, $P = 0.002$) from summer ($\bar{x} = 110.0 \pm 29.39$ ind. m⁻²) to fall ($\bar{x} = 18.0 \pm 4.31$ ind. m⁻²). As with dwarf surfclam, macoma exhibited strong over-winter larval recruitment and growth. However, this growth was much more significant at Non Feeding sites (Figure 23). From fall to spring, macoma densities increased over 40-fold ($t = -6.8$, $P = 0.002$), averaging 748.2 (SE = 108.09) ind. m⁻² at Non Feeding sites. Within this treatment, a maximum density of 1,628 macoma m⁻² was sampled at Shady Side. A significant increase ($t = -2.36$, $P = 0.002$) from fall to spring was also observed within the Feeding treatment, however, it only reached an average density of 72.1 (SE = 29.99 ind. m⁻²) in spring. Within treatments, no site differences were detected (Figure 24).

For Baltic macoma, size class distributions between treatments were similar as both exhibited comparatively strong over-winter recruitment. However, at Feeding sites, low numbers of macoma in summer and fall limited inferences regarding age classes. A small cohort centered around the 10-mm size class in summer was centered around the 13-mm size class in fall before a new cohort was observed in spring centered around 8 mm (Figure 25). This compared to Non Feeding sites, where several age cohorts were observed throughout the study. Three age cohorts were found in summer centered on 9, 20, and 27 mm. In fall, two cohorts were sampled, centered on the 14- and 20-mm size classes. The 27-mm age class found in summer was likely present in fall, but had reached a size enabling them to burrow below the sampling depth of the Peterson grab. Non Feeding sites in spring contained a new age cohort centered on 9 mm in addition to an age cohort centered on 24 mm. As with the larger age class in summer samples, the larger of the two age cohorts in fall likely reached a size enabling them to burrow beneath sampling depth in spring.

Mitchell Macoma

The most ubiquitous species throughout the study, mitchell macoma was commonly found in soft substrates within both treatments. However, differences in densities between habitats did exist. Densities of mitchell macoma were highest in mud/silt substrates ($\bar{x} = 64.6 \pm 6.03 \text{ ind. m}^{-2}$), which were nearly double those found in sand bottoms ($t = -4.1, P = 0.002$) and over 12-times the densities found in association with hard bottoms ($t = 9.5, P = 0.002$). As with Baltic macoma, overall treatment differences in mitchell macoma densities ($t = -5.7, P = 0.002$) were likely

driven by differences in habitats, averaging 51.4 (SE = 4.68) ind. m⁻² at Non Feeding sites compared to 18.9 (SE = 3.32) ind. m⁻² at Feeding sites.

At Non Feeding sites, densities of mitchell macoma increased throughout the season (Figure 26). Summer densities averaged 29.0 (SE = 4.69) ind. m⁻² before increasing to 51.3 (SE = 7.72) ind. m⁻² in the fall ($t = -2.5$, $P = 0.002$) and 71.9 (SE = 9.22) ind. m⁻² in the spring ($t = -1.9$, $P = 0.054$). At Feeding sites, mitchell macoma densities declined from an average density of 25.0 (SE = 6.85) ind. m⁻² in summer to 8.4 (SE = 2.66) ind. m⁻² in fall ($t = 2.3$, $P = 0.024$) before increasing to 23.3 (SE = 6.47) ind. m⁻² in the spring ($t = -2.1$, $P = 0.040$; Figure 26). Densities of mitchell macoma were relatively consistent among sites within the Non Feeding treatment, with no differences between sites detected (Figure 27). However, the Shady Side Feeding site ($\bar{x} = 44.7 \pm 7.24$ ind. m⁻²) contained significantly higher densities than both Herring Bay ($t = -4.5$, $P = 0.002$) and Poplar Island ($t = -5.56$, $P = 0.002$) throughout the study.

Size class distributions of mitchell macoma were similar to those of other infaunal bivalves. Age cohorts sampled in summer progressed in growth to the fall sampling period, followed by a new age class appearing in spring (Figure 28). As with Baltic macoma, larger mitchell macoma were likely present, but located beneath the grab sampling depth. Both treatments exhibited similar size class distributions between seasons.

Gemclam

Across both treatments, gemclam was highly associated with sand habitats, where it occurred in significantly higher densities than in both hard ($t = 2.07$, $P =$

0.004) and mud habitats ($t = -2.1$, $P = 0.002$). Densities of gemclam at Feeding sites averaged 7,065 (SE = 3,492) ind. m⁻² across sites and season, which was significantly more ($t = 1.9$, $P = 0.012$) than the 334.5 (SE = 217) ind. m⁻² found at Non Feeding sites. However, much of this difference was controlled by strong seasonal and site effects. Although seasonal densities of gemclam declined slightly at Non Feeding sites, they remained stable as compared to Feeding sites throughout the study period (Figure 29). Gemclam exhibited seasonal increases similar to other infaunal bivalves within the Feeding treatment. From summer to fall, no difference in gemclam densities was found ($t = -0.9$, $P = 0.426$). Similarly, high variance (SE = 10,111.51) resulted in no overall difference between fall and spring ($t = -1.4$, $P = 0.156$; Figure 29). However, removing the outlying two highest densities resulted in spring having significantly more gemclam m⁻² ($t = -1.2$, $P = 0.010$) than fall.

Spatially, gemclam was highly local and found almost exclusively at Poplar Island (Figure 30). Within the Feeding treatment, Poplar Island contained significantly more gemclam ($\bar{x} = 21,187 \pm 10,098$ ind. m⁻²) than both Herring Bay ($t = -2.1$, $P = 0.002$) and Shady Side ($t = -2.1$, $P = 0.002$). Additionally, high variances associated with gemclam densities were related to a highly local distribution at Poplar Island. Unlike other bivalves, gemclam broods its young and does not have a planktonic larval phase (Rankin et al. 1994). As a result, gemclam can accumulate very high local densities, often reaching into the hundreds of thousands m⁻² (Sanders et al. 1962). As previously mentioned, gemclam from just three sampling locations at the Feeding Poplar Island site represented 95% of all fauna sampled, averaging 68,878 (SE = 15,738) ind. m⁻² compared to 748.4 (SE = 248) ind. m⁻² at the remaining

sampling locations. Further, two of these locations contained $< 200,000$ gemclam m^{-2} in spring, which significantly contributed to the overall high variances in gemclam densities. At Non Feeding sites, gemclam was found at only one sampling location each at both Shady Side and Herring Bay with densities of 11 and 6 gemclams m^{-2} , respectively. Therefore, no statistical analyses were conducted comparing Poplar Island with other locations at Non Feeding sites.

Softshell Clam

Of the six bivalve species sampled, softshell clam was both the least frequently occurring and the least numerous. No explicit habitat preference was detected, although softshell clam was most numerous in sand substrates ($\bar{x} = 17.0 \pm 8.53$ ind. m^{-2}) compared to hard ($\bar{x} = 3.5 \pm 1.3$ ind. m^{-2}) and mud/silt bottoms ($\bar{x} = 4.9 \pm 1.55$ ind. m^{-2}). The inability to detect statistical differences was likely due to the overall low frequency of occurrence. Similarly, there was no overall difference in density between treatments ($t = -1.4$, $P = 0.132$). Virtually no softshell clams were sampled in summer or fall in either treatment (Figure 31). However, significant increases from fall to spring were found at both Feeding ($t = -2.4$, $P = 0.002$) and Non Feeding ($t = -2.2$, $P = 0.002$; Figure 31) sites.

Within both treatments, softshell clam was most numerous at Poplar Island. Within the Feeding treatment, Poplar Island contained significantly higher densities than Herring Bay ($t = -2.0$, $P = 0.004$), but not Shady Side ($t = -1.4$, $P = 0.196$), while within the Non Feeding treatment, Poplar Island was higher than both Herring Bay ($t = -1.7$, $P = 0.038$) and Shady Side ($t = -1.9$, $P = 0.012$; Figure 32). Analysis of shell lengths was only able to be conducted during the spring sampling, during which time

the 4-mm size class was the most numerous within both treatments, again suggesting over-winter larval settlement and growth. Similar to both macoma species (*Macoma* spp.), the lack of larger size classes does not preclude their presence, but rather larger clams may have occurred below the sample depth of the Petersen grab. Indeed in several samples, large siphons detached from large clams, and clearly identifiable as softshell clam, were present and indicate the presence of larger softshell clams below the sampling depth.

Common Clamworm

The common clamworm (*Neanthes succinea*; hereby called clamworm) was the most ubiquitous polychaete sampled and one of the most frequently occurring fauna. Densities of clamworms in mud and hard bottoms were nearly identical ($t = 0.03$, $P = 1.000$), averaging 63.4 (SE = 14.96) ind. m⁻² on hard bottoms and 62.9 (SE = 8.79) ind. m⁻² in mud substrates. However sand substrates contained significantly lower densities than both hard ($t = 1.9$, $P = 0.036$) and mud ($t = -2.6$, $P = 0.008$) substrates. Overall clamworm densities were similar between treatments ($t = -0.9$, $P = 0.392$). At Feeding sites, densities remained relatively stable throughout the study period ($t = 0.9$, $P = 0.416$; Figure 33). Conversely, clamworm densities at Non Feeding sites increased from a summer average of 31.6 (SE = 4.64) ind. m⁻² to an average of 53.3 (SE = 9.73) ind. m⁻² in fall ($t = -1.9$, $P = 0.042$), but remained stable into spring ($\bar{x} = 83.2 \pm 23.28$ ind. m⁻²; $t = -1.2$, $P = 0.260$). No site differences were found within the Non Feeding treatment (Figure 34), however, among Feeding sites, Shady Side contained significantly fewer clamworms than both Herring Bay ($t = 2.6$, $P = 0.006$) and Poplar Island ($t = -3.8$, $P = 0.002$).

Faunal Energetic Content

Differences between shell:tissue mass (ratio of shell mass to tissue AFDM) were found between all species of bivalve, except softshell clam which was not analyzed ($F_{4, 403} = 101.4$, $P < 0.001$). Pairwise species comparisons of mean shell:tissue values, pooled across seasons, yielded significant differences between all species (Table 7). Gemclam contained the highest mean shell:tissue values ($\bar{x} = 29.1 \pm 1.77$), which was >40% higher than hooked mussel ($\bar{x} = 17.9 \pm 1.73$) and more than double that of dwarf surfclam ($\bar{x} = 11.2 \pm 0.70$).

Of the five bivalve species analyzed, mitchell macoma had the lowest shell:tissue ratio ($\bar{x} = 4.7 \pm 0.61$), a value nearly half that of Baltic macoma ($\bar{x} = 9.3 \pm 1.22$). Seasonally, all bivalves, except gemclam, had the highest shell:tissue ratio in the summer. Shell:tissue values declined in fall before increasing slightly in spring (Figure 35). Conversely, seasonal differences were most pronounced with gemclams, having their lowest shell:tissue value in summer before increasing fall and remaining relatively unchanged into spring. However, these seasonal differences were not significant for gemclam ($F_{2, 49} = 1.6$, $P = 0.21$), dwarf surfclam ($F_{2, 89} = 0.4$, $P = 0.69$), Baltic macoma ($F_{2, 98} = 0.6$, $P = 0.54$), mitchell macoma ($F_{2, 124} = 1.3$, $P = 0.27$), or hooked mussel ($F_{2, 49} = 1.5$, $P = 0.24$). No differences between sites were detected.

As with shell:tissue ratio, differences between bivalve species were found for the overall ratio of total ash mass to AFDM (Ash:AFDM). However, differences were not found between all species (Table 8). Hooked mussel had the highest Ash:AFDM ratio (pooled across seasons; $\bar{x} = 2.9 \pm 0.39$), but was not significantly higher ($t = 0.9$, $P = 0.89$) than the mean ratio ($\bar{x} = 2.7 \pm 0.42$) for gemclam. Likewise, no

statistical difference was found ($t = 1.4$, $P = 0.65$) between Baltic macoma ($\bar{x} = 1.8 \pm 0.18$) and dwarf surfclam ($\bar{x} = 1.9 \pm 0.25$). As with the shell:tissue ratio, mitchell macoma had the lowest ash:AFDM ratio of all bivalves ($\bar{x} = 1.0 \pm 0.16$). Seasonally, no statistical differences were found for gemclam ($F_{2, 49} = 2.69$, $P = 0.08$), dwarf surfclam ($F_{2, 89} = 0.6$, $P = 0.54$), Baltic macoma ($F_{2, 98} = 2.1$, $P = 0.13$), mitchell macoma ($F_{2, 124} = 0.4$, $P = 0.65$), or hooked mussel ($F_{2, 49} = 2.7$, $P = 0.08$; Figure 36). No differences between sites were detected for any bivalve. However, gemclam remained relatively stable from summer to fall before declining over winter, while hooked mussel declined from summer to fall before remaining relatively stable over winter (Figure 36). No differences were found between sites.

Grab – Diver Comparisons

Twenty-one identifiable species were sampled using the diver technique during spring. This is compared to 27 identifiable species sampled via the grab sampling technique during the same time period and using the same effort (Table 9). Further, diver-obtained samples contained four unique species compared to nine obtained with grab sampling. Three of the four species unique to diver sampling were also unique across the entire study, including the isopod (*Sphaeroma quadridentatum*), the Atlantic mud crab (*Panopeus herbstii*), and the impressed odostome (*Boonea impressa*). The most frequently occurring species in diver samples was Baltic macoma, which was found in 90% of samples. In grab samples, bay barnacle was the most frequently occurring, which was found at 90% of sampling locations. Overall, no clear pattern in faunal differences between the two sampling techniques was found, although densities were higher for most species with grab

sampling (Table 9). Much of this difference was the result of adjusting uncommon species (those with one or two individuals across all sites) sampled using grab sampling to individuals m^{-2} .

For infaunal species, grab sampling contained higher densities of all identified species and higher frequency of occurrence for 10 out of 14 species. Additionally, of the nine species unique to grab samples, two-thirds (6) were infaunal. The presence of higher densities and numbers of unique species in grab samples suggests that the grab was more effective at infaunal sampling or was able to sample deeper into the sediment.

For epifaunal species, results were more variable. White anemones were found in higher densities using grab sampling, whereas the frequency of occurrence was equal. Similarly, bay barnacle was found in higher densities with grab sampling, although much of this difference is likely related to barnacles being 30% more frequent when sampled using the grab technique. Differences in frequency of occurrence (20% more common) also likely explain higher densities of tube-building amphipod in diver-obtained samples. The fourth dominant epifaunal species, hooked mussel, was found in nearly identical densities between sampling techniques, although mussels were more frequently encountered using the diver techniques. Mud crabs, common on oyster bars, were encountered at similar densities between techniques, though the diver technique yielded the Atlantic mud crab, which was not encountered in grab samples. Overall, this variation between techniques in frequency of occurrence and density is likely the result of high patchiness within the area sampled.

DISCUSSION

Determining the habitat use of diving ducks is fundamental to elucidating their ecological relationship, and niche, within a coastal ecosystem. Understanding this relationship is critical for quantifying the risk of anthropogenic stressors to a population, whether they are hunting and habitat degradation within an estuary or emerging issues such as offshore wind farms. To this end, the present study represented the first quantification of surf scoter foraging ecology within Chesapeake Bay and established surf scoters within the broader context of Bay ecology. Using a novel comparative approach to quantify habitat and faunal variables, this study determined the location of feeding surf scoters and quantified what factors may be driving surf scoter distribution. In addition to commonly used seasonal analysis of faunal variables (e.g., density), measurements of energetic characteristics were made in an attempt to further explain scoter habitat and food choices.

Surf Scoter Distribution

Surf scoter surveys within the study area indicated a tendency for scoters to feed in specific regions, although their overall distribution contained large spatial and temporal variability. As has been encountered in previous attempts to delineate surf scoter populations within Chesapeake Bay (e.g., Koneff et al. 2005), this temporal variation was due to the presence or absence of rafts in excess of several thousand scoters. The influence of large rafts on surveys was also evident during the 2005 mid-winter waterfowl survey, when a record high of 45,042 scoters (all species) were counted in the Bay. A significant proportion of this total was found in a single raft in

excess of 20,000 scoters located near North Beach, Maryland just south of the study area used in this project (J. Wortham, U.S. Fish & Wildlife Service, pers. comm.).

One possible cause of the large spatial and temporal distribution of scoter flocks in this study is increased guided seaduck hunting in Chesapeake Bay (Perry and Deller 1994), which generally runs from early October to late January or early February. Unlike traditional waterfowl hunting, which is typically conducted from a blind, the distance from shore of most seaducks requires larger boats and is often done through a commercial guide. During the December survey, approximately 5 hunting boats were anchored along the northwestern portion of Poplar Island, including the Poplar Island Feeding site. This likely contributed to the absence of large numbers of scoter during the December survey. Although not observed during formal surveys, seaduck hunting boats have also concentrated on waters within and around the Herring Bay Feeding site. The presence of guided hunting boats at these sites further supports the importance of both areas to surf scoters. In addition to displacement due to guided hunts, scoters are easily disturbed by traditional boat traffic (both recreational and commercial), as evidenced by the distance scoters would disperse during boat surveys.

Habitat Composition

Unlike surf scoters along the British Columbia coast, which are primarily located within the intertidal zone and <1 km from shore (Lacroix 2001, Lewis et al. 2005, and Žydelis et al. 2006), the majority of scoters in this study were located within a wide water depth range and >1 km from shore. Much of this difference can be attributed to the physical dynamics and bathymetry of the Chesapeake Bay, which

is relatively shallow and experiences small tidal fluctuations. This contrasts with coastal British Columbia, which is highly convoluted and contains numerous fjords and coastal bays that experience strong tidal cycles.

Sediment composition seems a major factor for determining surf scoter distribution within the mesohaline region of Chesapeake Bay. The dominance of sandy substrates with grain sizes ≥ 0.250 mm comprising $>50\%$ of the overall sediment profile (by mass) at Feeding sites is consistent with sediment descriptions by Žydelis et al. (2006) and Stott and Olson (1973), although quantitative analyses were not conducted in these studies. More striking was the lack small grain sizes at Feeding sites and their dominance at Non Feeding sites, suggesting surf scoters either avoid mud/silt substrates or have preference for sand. Two possible hypotheses for this treatment difference are 1) surf scoters prefer prey in other substrates, or 2) prefer not to feed in mud/silt. A lack of preferred food items is not the likely cause of surf scoter avoidance of mud/silt, since no statistical difference in dwarf surfclam (most frequently consumed item by surf scoters) density was found between sand and mud/silt, with mud/silt averaging 51 ind. m^{-2} compared to 38 ind. m^{-2} in sand.

Avoidance of mud/silt bottoms may be related to the foraging habits of surf scoters. One characteristic of surf scoter foraging is a tendency to feed in groups, with scoters frequently synchronizing their diving bouts when foraging (Beauchamp et al. 1992). This synchronization may increase foraging success (Lewis et al. 2005) as well as maintain raft cohesion while feeding (Beauchamp et al. 1992). It is likely that a large number of scoters feeding simultaneously in a relatively small area would also greatly disturb the sediment surface, causing sediment particles to become

suspended in the water column. The amount of suspended sediment would increase with decreasing grain size, with particle suspension greatest with mud/silt substrates, causing the water column to become cloudy and thus greatly reducing visibility and possibly, foraging success.

A number of waterfowl species are known to forage under reduced visibility, or feed nocturnally and thus rely on tactile foraging (Nilsson 1972, Perry and Uhler 1988, Tome and Wrubleski 1988). Because foraging over mud/silt habitats would greatly reduce vision, a tactile foraging strategy may be more efficient than visual foraging over mud/silt bottoms (Tome and Wrubleski 1988). It is possible that surf scoters avoid mud/silt substrates in an effort to limit the use of tactile foraging methods. Indeed, Lewis et al. (2005) suggested a preference for visual feeding as a possible explanation (in addition to others) for the strong propensity of surf scoters to avoid nocturnal feeding. The use of visual cues for the location of prey, either through direct observation of bivalve siphons protruding from the sediment surface or a successful prey capture by other scoters, may be energetically more efficient (Tome and Wrubleski 1988). This may especially be true in Chesapeake Bay, where many habitats are extremely patchy (i.e., often interspersed with other substrates), and many species have highly patchy distributions (e.g., dwarf surfclam) even in comparably homogenous bottom types.

In addition to sand, the dominant habitat types at Feeding sites were hard bottoms, which were comprised of packed clay and/or oyster bar. At Herring Bay, the three dominant epifaunal species found in this study (hooked mussel, bay barnacle, and white anemone) were often found attached to large rocks or scattered

fossil oyster shell. In addition, video and photographs captured by divers at this site also found these epifaunal species directly attached to the substrate. A portion of the Herring Bay site is located within a Yates 1911 oyster survey boundary (see Smith et al. 2003), although no evidence of an active oyster bar was found.

In contrast to Herring Bay, portions of the Poplar Island Feeding site did contain active oyster bar, although the low number of live oysters collected indicated a degraded bar. The high degree of patchiness found at this site (53% of sampling locations containing both oyster and sand) was also recorded by Halka and Ortt (2002a), who conducted side-scan sonar surveys of this area. Their sonar survey found strong movement of sand over this natural oyster bar, with strong wave energy causing the sand to be highly mobile and transient. Divers over this study site also noted a mixture of oyster shell and sand habitats.

Of particular interest was the presence of hard bottom at Non Feeding sites, especially at Poplar Island, which contained a patchwork of all three categorized habitat types. Oyster samples from this site were similar to those from the Poplar Island Feeding site, and also indicated a degraded bar. A portion of this site also overlapped with a side-scan sonar survey by Halka and Ortt (2002b) and a natural oyster bar surveyed by Yates in 1911 (see Smith et al. 2001). Side-scan sonar of this natural oyster bar, which ran along the eastern side of Poplar Island, found no oyster shell except for clusters at the northern end, which overlaps with the Non Feeding site.

In addition to containing a similar number of live oysters sampled, both oyster bars at Poplar Island also appeared to be of the same habitat quality based on faunal

diversity and numbers of live oysters present. Both bars contained a variety of fauna associated with oyster bars including mud crabs, anemones, and fish (e.g., naked goby and skillettfish; Rodney and Paynter 2006). Further, both oyster bars contained similar densities of hooked mussel (summer and fall) and bay barnacle (all seasons), both dominant fouling species (Figure 37). However, the Non Feeding oyster bar differed from the Feeding oyster bar in that it comprised an overall smaller proportion of the study site and was associated with mud/silt substrates. Both of these factors could explain why scoters were not observed foraging in this area, although it does not preclude other factors, such as high boat traffic relative to the Feeding site.

Sampling Technique

Quantitative comparisons between diver and grab sampling were difficult to make and extremely high variability yielded statistical results that were not meaningful. While differences between techniques in densities of infaunal species was likely due to sampling depth, differences in epifaunal densities were likely due to highly patchy habitat distribution. For example, although there was no overall difference in hooked mussel densities, specific locations within each site were very different between techniques. This fine-scale difference due to high habitat heterogeneity, here associated with degraded oyster bars, reflects the need for fine-scale sampling when conducting an analysis such as this to accurately discern appropriate habitat variables (Chamberlain and Fuller 1999). This fine-scale patchiness may also drive raft foraging in surf scoters in Chesapeake Bay. With large scoter rafts encompassing a large areal extent, loafing individuals would likely move or drift away from a small-scale foraging site between feeding bouts. Observing

other scoters within the raft feeding would likely reduce search time, and thus increase foraging efficiency, in a patchy environment (Guillemette et al. 1993).

Faunal Characteristics and Trends

A number of faunal species exhibited strong increases in abundance between the fall and spring sampling period (e.g., Baltic macoma, dwarf surfclam, and bay barnacle), although this increase frequently varied between treatments. Two possible explanations for this increase are 1) juveniles were present, but not detected or retained by the sieve during fall sampling, or 2) these species spawned and/or larvae settled after the fall sampling. Since fall sampling occurred through the last week in October, it is possible that both of these explanations are correct. Although there is likely inter-annual variability in recruitment, this is consistent with Holland et al. (1977) who found that Baltic macoma and dwarf surfclam had large reproductive pulses in late-fall, with juveniles not able to be detected until winter. These recruits likely entered into a seasonal pattern of having limited growth over winter before increasing their growth rate in spring (Holland et al. 1987, Gerritsen and Irvine 1994). This scenario typically results in the highest densities of clams occurring in spring (Holland et al. 1977, Gerritsen and Irvine 1994). Growth then likely slowed into summer and fall as tissue production was shifted to follicle development before spawning (Nichols and Thompson 1982, Gerritsen and Irvine 1994). Further, only one age cohort was observed for dwarf surfclam, although there was a relative spread in size classes, whereas two cohorts were found for Baltic macoma. This suggests that, in addition to a strong late-fall pulse, a low level setting of dwarf surfclam occurred for a period of time before late-fall while Baltic macoma exhibited a

bimodal setting pattern of multiple reproductive pulses (Shaw 1965). Although few softshell clams were sampled, Shaw (1965) also indicated a bimodal setting pattern, and consistent with this study, a significant fall setting period.

While barnacle densities significantly increased from fall to spring, the majority of spring barnacles were extremely small. Unlike many bivalves, the timing of barnacle setting is somewhat variable. Shaw (1967) found a bimodal setting pulse of barnacles in early and late summer, whereas Cory and Nauman (1969) found strong spring sets early in the study but very strong late-fall sets later in the study. Given the small size of most barnacles sampled in spring, and that microscope examinations would have yielded juvenile barnacles in fall, it is likely that either a very strong late-fall or early-spring set occurred during this study.

The other bivalve that exhibited strong production from fall to spring was gemclam. Overall, gemclams are very locally distributed in Chesapeake Bay and the majority of gemclams sampled in this study were found around Poplar Island, primarily at the Feeding site. Although densities of gemclams found at two locations in spring ($>200,000 \text{ m}^{-2}$) were extraordinarily high, this is consistent with other analyses involving local “hot spots” of gemclam (Green and Hobson 1970, Commito et al. 1995). Gemclams are ovoviviparous, and without a planktonic larval stage, they rely on hydrodynamic action for dispersal (Commito et al. 1995) resulting in relatively weak dispersal abilities (Holland 1985). Indeed, many gemclams dissected from summer and fall samples contained juvenile clams. The high densities of gemclam found in this study were likely the result of the release of numerous juveniles from adult clams combined with weak juvenile dispersal powers.

The remaining prominent faunal species sampled exhibited various trends in setting. Mitchell macoma densities increased throughout the study at Non Feeding sites, and differences between treatments were likely related to habitat. The steady increase in mitchell macoma abundance was likely the result of a continuous set from spring to fall (Blundon and Kennedy 1982). For both anemones and the common clamworm, densities increased throughout the study at Non Feeding sites, and declines in these species at Feeding sites may be related to predation or disturbance by surf scoters.

Depletion of Hooked Mussel

The most interesting seasonal trend involved hooked mussel. Typically regarded as a fouling organism, limited studies have examined seasonal mussel patterns. In agreement with this study where densities increased between summer and fall at a similar rate in both treatments, Shaw (1965) found hooked mussel to continuously spawn from spring into fall. Further, stable densities at Non Feeding sites between the fall and spring sampling indicate that no additional spawning or setting took place in late fall. The most obvious change in seasonal abundance occurred at Feeding sites, where densities decreased significantly from fall to spring. This striking reduction in mussel densities could be related to an over-winter mortality event at Feeding sites, although close proximity to the Non Feeding sites where densities did not decline likely precludes this cause. More likely, this seasonal decline was the result of predation pressure from a variety of crustacean and fish species, as well as from scoters.

One group of major predators of bivalves on oyster reefs, including hooked mussels in Chesapeake Bay, are xanthid crabs (Meyer 1994, Milke and Kennedy 2001), represented in this study by the estuarine and flatback mud crabs. The presence of, and correlation of mud crabs with, oyster bars (and thus hooked mussel) suggests that at least some predation by mud crabs may have taken place. Size classes of hooked mussels present at Feeding sites are within the range of sizes that mud crab feed on in Chesapeake Bay; although they represent the upper range of the largest crabs (Milke and Kennedy 2001). Whetstone and Eversole (1981) found significantly reduced predation by mud crabs at temperatures of 10°C compared to temperatures of 17°C or higher. Within this study area, bottom water temperatures typically average below 10°C from November through April (Maryland DNR), approximately the intervening time between fall and spring samples. This suggests that predation by mud crabs on hooked mussel through the winter months may have been minimal.

Other sources of benthic macroinvertebrate mortality, including hooked mussel, within Chesapeake Bay includes predation by a number of fish species (e.g., Atlantic croaker, *Micropogonias undulates*, and spot, *Leiostomus xanthurus*) and blue crab. Many species of fish either emigrate from the Bay or reduce their consumption rates during winter (Hartman and Brandt 1995, Baird and Ulanowicz 1989). Further, although blue crabs are known to feed directly on hooked mussel (Ebersole and Kennedy 1995), blue crabs are primarily dormant during the over-winter period between the fall and spring sampling (Mauro and Mangum 1992). This indicates that predation on benthic macrofauna by fish and blue crabs may also have been minimal

during the winter period between fall and spring sampling (Virnstein 1977, Baird and Ulanowicz 1989).

Reduced consumption of hooked mussel by fish and crab species during winter, combined with stable mussel densities at Non Feeding sites, strongly suggests that the observed decline in mussel densities was a result of direct predation by surf scoters. While it cannot be definitively determined that reductions in hooked mussel densities in this study were also not the result of predation by other seaducks (e.g., black scoter and white-winged scoter) that have been found to feed on hooked mussel (Perry et al. 2007), the high proportion of surf scoter numbers relative to these other species observed during surveys suggests the predation impact of other seaducks would be minimal. The use of habitat exclusion devices would likely be needed to positively attribute the decline in hooked mussel to surf scoters (or other seaducks). A number of studies have found diving ducks to be significant predators of intertidal and subtidal epifaunal and infaunal bivalves (Hamilton 2000, Larsen and Guillemette 2000, Poulton et al. 2002), including surf scoters (Lacroix 2001). Strong predation by seaducks can have significant effects on community dynamics, and Hamilton (2000) suggested that the common eider may be a local keystone predator on intertidal mussel communities. Similarly, Lacroix (2001) found significant depletion of intertidal mussels after surf scoter feeding bouts, with mussel populations taking up to two seasons to recover. These two studies dealt with blue mussel in more shallow water habitats.

Further supporting the hypothesis that surf scoters were the primary cause of the decline of hooked mussel is the size class structure of mussels at Feeding sites.

During late fall, the size class distribution of hooked mussels at Non Feeding sites indicated a slightly younger (and thus smaller) age cohort than at Feeding sites, which had a wide range of size classes. During the spring sampling, size class distribution at Non Feeding sites were widely distributed from 2 – 26 mm, with a maximum of over 34 mm. However, at Feeding sites, in conjunction with the overwinter decline in mussel densities, size classes >16 mm were largely absent in spring with only a few mussels in the 20-mm size class. Hooked mussels can grow in excess of 35 mm and the absence of these larger size classes in this study may be a reflection of poor habitat quality or predation pressure from surf scoters. The range of size classes missing at Feeding sites relative to Non Feeding sites is very similar to the most common size classes consumed by surf scoters in Chesapeake Bay. During their analysis of surf scoter food habits, Perry et al. (2007) primarily found mussels ranging from 16 – 24 mm in surf scoter gullets (accurate measurements can only be made using mussels found in the gullet) and concluded these were the ideal size classes for surf scoter foraging.

Size selection of mussels has been well documented in diving ducks, especially common eider (Draulans 1982, Bustnes 1998, Guillemette 1998, Hamilton et al. 1999). The selection of specific size classes, typically smaller than the maximum present, by common eider has been suggested to have several physiological implications. These include consuming smaller mussel to minimize salt intake (Nyström and Pehrsson 1988, Hamilton et al. 1999) and, since there is a logarithmic relationship between shell length and organic content (Guillemette 1998), minimizing the physiological costs of shell intake while maximizing organic content (Bustnes

and Erikstad 1990, Bustnes 1998, Guillemette 1998). In addition, Draulans (1982) suggests tufted duck (*Aythya fuligula*) may be seeking to reduce the mechanical difficulties in swallowing too large of prey and reduce the total handling time of prey between foraging bouts. Although size selection has not been documented in surf scoters, the near elimination of size classes >16 mm at Feeding sites coupled with food habits results suggests surf scoters may also be size selective predators.

Prey Selection: Comparisons with Food Habits

Similar to many predators, seaducks can often be categorized as selective (Guillemette et al. 1992, Larsen and Guillemette 2000), or opportunistic predators (Nilsson 1969, Richman and Lovvorn 2003) within a preferred habitat, with foraging strategies often determined by faunal densities and characteristics (Degraer et al. 1999). Within the study area, surf scoters appear to use a combination of both strategies. Gemclam represented the species averaging the highest densities and biomass at Feeding sites throughout the study, averaging 7,065 ind. m⁻² and 18,486 mg AFDM m⁻². However, gemclams are only the third most frequently consumed food item in the Bay (13% of overall surf scoter diet and preceded by dwarf surfclam and hooked mussel) and because of the small size of gemclams (typically <4 mm), it is likely that surf scoters feeding on gemclam would ingest large quantities of deceased gemclams (see Figure 7). Gemclam also averaged the highest ratio of shell mass to tissue AFDM and combined with a likelihood of deceased shell ingestion, feeding on gemclam would be energetically costly. Further, because of the high likelihood of ingesting large quantities of deceased shell, food habits samples containing gemclam, either whole or fragments, would likely be inflated relative to

their overall nutritional and energetic contribution. It is possible that surf scoters select against gemclam for these reasons.

Whereas gemclams are likely inflated in surf scoter food habit studies, clamworms may be under-represented. In the food habits analysis by Perry et al. (2007), the common clamworm represented only a trace proportion of the overall food consumption by surf scoters. However, in every instance of occurrence in food habits samples, the presence of clamworms was determined through the identification of clamworm jaws (no other portions of clamworms present). Digestion of clamworms likely occurs very rapidly, leaving only the small jaw parts for any period of time, and inclusion of clamworm soft-tissue in food habits calculations would likely result in significantly higher volumes. The common clamworm was the most frequently occurring polychaete and was common in all habitats, averaging 84 ind. m⁻² and 222 mg AFDM m⁻² at Feeding sites. Although food habits data indicate surf scoters feed very little on clamworms, declines in densities from fall to spring at Feeding sites suggest foraging on clamworms may occur.

Three other infaunal bivalves constitute a small proportion of surf scoter diet, with the ubiquitous mitchell macoma representing <1%, softshell clam <1%, and Baltic macoma 7%. Although occurring in lower densities relative to Non Feeding sites, these species did occur at Feeding sites and contained the lowest ratio of shell mass to tissue AFDM. However, it does not appear as though surf scoters actively select these species. This may be a function of the presence of mostly small juveniles in winter and/or location of these infaunal bivalves relative to the sediment surface.

Densities of dwarf surfclam, representing the largest proportion of surf scoter diet, increased over winter in both treatments. It is possible that the high proportion of dwarf surfclam in surf scoter diet is a function of depleted hooked mussel populations. Although hooked mussel has a significantly higher ratio of shell mass to tissue AFDM than dwarf surfclam, high average biomass (averaging 1,750 and 758 mg AFDM m⁻² for hooked mussel and dwarf surfclam, respectively) may make hooked mussel a preferred prey item. No seasonal examinations of surf scoter food habits have been done in this region, but doing so may indicate higher feeding on hooked mussel in early winter followed by increasing importance of dwarf surfclam by early spring. Further study is needed to determine whether surf scoters are opportunistically feeding on dwarf surfclam after preferred hooked mussel are depleted or whether food choice is based on density dependent availability.

Although the shell to tissue mass ratio was not examined for specific size classes of bivalves in this study, shell minimization (or total ash minimization) does not appear to be a significant factor for faunal selection by surf scoters across species in Chesapeake Bay. One unexpected outcome of examining the AFDM content of bivalve shell and tissue separately was the high amount of AFDM within the shell. Indeed, shell contained equal or significantly greater AFDM content as tissue in most species. Although complex and differing between species, genera, and family, the shell of most mollusks is composed of two or more layers of an organic matrix from which mineralization takes place and forms the outer calcium carbonate shell (Weiner and Traub 1984). The content of the organic matrix component is primarily proteins rich in acidic-amino acids (Weiner 1979). The high protein content of bivalve shell

would constitute a significant source of biomass were it able to be assimilated by waterfowl.

Most analyses of waterfowl bivalve energetics and/or food habits involve either tissue alone, or examinations using the whole bivalve. This is primarily due to the assumption that shell fragments inhibit tissue digestion, especially since significant amounts of defecated shell can be found at locations where seaducks frequently move onto land (i.e., breeding common eiders feeding on blue mussel in Hudson Bay, M. C. Perry pers. comm.). However, assimilation trials where several species of seaducks, including surf scoter, were starved for 24 hours before being orally gavaged a known quantity of whole mussels, did not result in scoters defecating significant quantities of shell (A. Wells-Berlin unpub. data). Similarly, Thompson and Sparks (1977) found little defecated shell fragments from lesser scaup orally gavaged Asian clam (*Corbicula manilensis*), with a simulated analysis of digestion finding a 42% reduction in mass that was attributed to tissue digestion and shell dissolution. It is possible that the lack of shell fragments in the feces of captive waterfowl may be a physiological response to starvation or stress, whereas wild populations with abundant resources quickly pass shell fragments to reduce the energetic costs of shell dissolution to access easily digestible tissue. Further studies are needed to ascertain the contribution, if any, of mollusk shells to the total biomass assimilated by waterfowl.

Relationship with Habitat Degradation in Chesapeake Bay

Of the anthropogenic stressors affecting Chesapeake Bay habitats, the combination of the loss of oyster beds and sedimentation appear to have directly

impacted surf scoter foraging habitats within the study area. The presence of foraging surf scoters over degraded oyster bars, especially at Poplar Island, strongly suggests that the loss of oyster habitats in Chesapeake Bay (currently at 1% of historic levels) has reduced the availability of prey resources. Hooked mussel densities averaged (on hard bottom locations across seasons at Feeding sites) 23.3 ind. m⁻², which is slightly higher than the approximately 15 ind. m⁻² found by Rodney and Paynter (2006) in a similar region of the Bay on non-restored, or degraded, oyster reefs. The highest hooked mussel density found at Feeding sites in this study was 117 individuals m⁻² at a Poplar Island location in fall. This compares to an average density of <3,000 ind. m⁻² found by Rodney and Paynter (2006) on restored, healthy oyster bars. This significant increase in the density of hooked mussel on restored bars relative to degraded beds represents a potentially dramatic increase in potential food resources. As oyster habitats declined over the 20th century, and thus hooked mussel densities, it is possible that surf scoters have been forced to supplement their diet with alternative food resources (e.g., dwarf surfclam). Further, a ten-fold increase in oysters over the 1994 baseline, as outlined in the *2000 Chesapeake Bay Agreement*, has the potential to produce significant habitat gains for surf scoters. More detailed analyses of the relationship between surf scoters and oyster bars, and oyster restoration in Chesapeake Bay, should be conducted to determine the impact of management actions on surf scoter foraging and populations.

In addition to its impacts on oyster beds, the avoidance of mud/silt habitats by surf scoters suggests that sedimentation of the Chesapeake Bay has also directly reduced available foraging habitat. Although the specific grain size of sediment run-

off can be locally dependent on specific land-uses within a subwatershed (e.g., the Choptank River; Brush 1989), overall sediment inputs into Chesapeake Bay are largely composed of mud/silt (Davis 1985, Cooper 1995). Suspended mud/silt sediments can travel large distances in the Bay, especially during storm events (Brush 1989), and can completely cover previously sand bottom habitats (Cooper 1995). This covering of sand habitats has likely served as an additional stressor to surf scoter foraging habitat in Chesapeake Bay. As with the restoration of oysters, management directives within the *2000 Chesapeake Bay Agreement*, in conjunction with best management practices within the watershed, aimed at reducing sediment inputs to the Bay, would likely benefit surf scoters through improvements in the quality of benthic habitats.

In contrast to oyster loss and sedimentation, current surf scoter foraging habitats (with Feeding sites averaging 3.23 m in water depth) do not appear to be influenced by seasonal hypoxia. Although direct evidence is limited, it is likely that recurring summer hypoxia in waters >10 m in depth has reduced the availability of benthic prey resources for surf scoters in the Bay. Surf scoters frequently feed intertidally along the Pacific coast, but they have been found to depths of 10 m (Lewis et al. 2005, Lacroix 2001), and seaducks as a group are known to feed at depths well in excess of 10 m (Kaiser et al. 2006, Richman and Lovvorn 2003, Beauchamp et al. 1992). It is possible that surf scoters historically fed in these deeper waters of Chesapeake Bay, which may have supplemented prey resources in shallow water during years of poor benthic productivity or significant winter icing. It has been suggested that recent increases in mid-winter waterfowl survey results for

scoters may be the result of their foraging closer to shore, thus increasing the probability of being counted (D. Forsell, U.S. FWS, pers. comm.).

CONCLUSION

This study, in seeking to delineate surf scoter foraging ecology within the mesohaline Chesapeake Bay, found that surf scoters were not evenly distributed and tended to concentrate feeding to three locations. Through utilizing a comparative approach to habitat delineation, this study was able to determine that surf scoters were feeding over sand and hard bottoms (degraded oyster bars and packed clay), and tended to avoid mud/silt substrates. Both treatments contained patches of sand and hard bottom, however these substrates comprised a larger proportion of Feeding sites while mud/silt was found nearly exclusively at Non Feeding sites. One possible theory for the seemingly avoidance of mud/silt may be the preferred use of visual feeding methods (e.g., the visual detection of bivalve siphons protruding from sediment), since dwarf surfclams were present in both mud/silt and sand substrates.

The seasonal trend of hooked mussel densities in the study area indicated that overwinter depletion of mussels was concomitant with significant increases in dwarf surfclam densities. This suggests that surf scoter feeding in the mesohaline Chesapeake Bay may be a function of both preference (e.g., hooked mussel) and availability (e.g., dwarf surfclam). However, having to feed on infaunal clams rather than epifaunal could affect dive duration and/or frequency of dive bouts resulting from increased search times that may ultimately increase energetic costs. Further, surf scoters do not appear to be selecting available food items based on a physiological attempt to minimize shell intake.

The significant decline in oyster beds over the last century has likely impacted surf scoters. The faunal composition of restored versus degraded oyster bars is dramatic, with restored bars having both significantly higher diversity of fauna and absolute numbers of individual species, including hooked mussel. This difference suggests that the restoration of oyster bars within Chesapeake Bay may be beneficial to surf scoters. In addition, sedimentation in the Bay has not only impacted oyster bars, but significant inputs of mud/silt sediments has likely served to further reduce available foraging habitat.

MANAGEMENT IMPLICATIONS

Although definitive population estimates have remained elusive, surf scoters likely represent a significant proportion of the total waterfowl in Chesapeake Bay. It is probable that surf scoters have been affected by the decline in oyster habitat over the 20th century, although it is difficult to determine whether this and/or other perturbations have impacted surf scoter numbers. For the most part, fisheries and waterfowl in Chesapeake Bay are currently managed independently, although the linkage of surf scoters to broader Bay ecosystem parameters suggests management of these resources should be linked. The potential for increased and improved foraging habitat through the restoration of oyster bars in Chesapeake Bay is significant, and the interaction of surf scoters with restoration projects should be taken into account during the restoration planning process. Further, management strategies aimed at reducing sediment inputs and improving the overall quality of benthic habitats would likely create and improve habitat conditions for surf scoters.

An additional concern for surf scoter populations in Chesapeake Bay is the impact of increased guided hunts. The effect of these hunts was clearly visible during this study. The repeated targeting of hunting boats at specific locations likely excludes surf scoters from preferred habitats and causes excess flying, both of which can lead to long-term energetic implications. In addition, numerous wounded and crippled ducks were observed during surveys. One widely suggested strategy to limit the impacts of hunting on diving ducks is the use of waterfowl reserves with limited or no hunting access. A possible mechanism for the development of waterfowl reserves could be the utilization of oyster reserves, which currently limit or exclude shellfish harvest. Extending the boundaries of oyster reserves to include surface waters with limited or no seaduck hunting would serve to not only provide a refuge for seaducks from hunting, but to do so over preferred foraging habitats. This would be especially beneficial at Poplar Island for not only surf scoters, but also long-tailed ducks and white-winged scoters.

APPENDIX 1

Faunal Assemblage	Species	AFDM ind-1 (mg)						Ash : AFDM	
		Summer		Fall		Spring		x	SE
		x	SE	x	SE	x	SE	x	SE
Cnidarians	<i>Haliplanella lineata</i>	-	-	10.16	3.24	-	-	0.13	0.08
	<i>Diadumene leucolena</i>	6.94	0.70	3.29	1.19	10.48	2.62	0.28	0.04
	<i>Edwardsia elegans</i>	-	-	6.48*	-	-	-	0.14*	-
Polychaetes	<i>Glycera sp</i>	0.09	0.0	0.67	0.27	1.85	0.54	0.12	0.08
	<i>Glycinde solitaria</i>	-	-	0.10	0.05	2.73	1.60	0.19	0.16
	<i>Neanthes succinea</i>	1.62	0.19	1.97	0.23	4.48	0.33	0.33	0.04
	<i>Pectinaria gouldi</i>	-	-	4.24	2.01	8.12	1.15	3.42	1.39
	<i>Heteromastus filiformis</i>	0.66	0.09	0.64	0.09	0.57	0.14	0.45	0.15
	<i>Nephtys sp.</i>	-	-	-	-	0.45	0.24	0.00	0.00
	<i>Eteone sp.</i>	-	-	-	-	0.19	0.09	0.00	0.00
Nemeritans	<i>Micrura leidyi</i>	-	-	-	-	7.35	1.36	0.00	0.00
	<i>Nemertan sp.</i>	1.20	0.67	3.12	1.51	0.87	0.55	1.40	1.33
Gastropods	<i>Sayella chesapeakea</i>	-	-	-	-	2.63	0.78	0.26	0.10
	<i>Boonea sp.</i>	-	-	-	-	2.99*	-	1.33*	-
Bivalves	<i>Mulinia lateralis</i>	19.76	3.48	43.74	3.78	16.86	1.51	2.57	0.69
	<i>Gemma gemma</i>	2.54	0.19	3.49	0.48	1.82	0.22	2.69	0.26
	<i>Macoma balthica</i>	17.48	3.66	30.25	3.93	5.34	0.46	1.72	0.11
	<i>Macoma mitchelli</i>	5.05	1.78	7.79	1.26	10.04	0.89	1.01	0.10
	<i>Mya arenaria</i>	58.96	24.11	-	-	50.82	17.22	0.91	0.12
	<i>Ischadium recurvum</i>	37.96	7.80	131.54	35.95	58.84	13.79	2.76	0.24

Faunal Assemblage	Species	AFDM ind-1						Ash : AFDM	
		Summer		Fall		Spring		x	SE
		x	SE	x	SE	x	SE		
Amphipods	<i>Incisocalloipe aestuarius</i>	-	-	0.08	0.01	1.01	0.20		
	<i>Leptocheirus plumulosus</i>	0.78	0.16	-	-	0.34	0.16	0.13	0.06
	<i>Corophium lacustre</i>	0.17	0.04	0.24	0.12	0.35	0.14	0.38	0.08
	<i>Monoculodes edwardsi</i>	-	-	-	-	0.53	0.13	0.12	0.07
	<i>Gammarus mucronatus</i>	-	-	-	-	0.05*	-	0	0
	<i>Haustoridae sp.</i>	0.85	0.09	0.84	0.72	1.79	0.56	0.22	0.07
Mysids	<i>Americamysis bigelowi</i>	0.55*	-	-	-	-	-	0.018*	-
Isopods	<i>Cyathura polita</i>	0.09*	-	-	-	4.66	2.30	0.08	0.07
	<i>Edotia triloba</i>	0.01*	-	-	-	1.02	0.62	0.13	0.13
	<i>Chiridotea almyra</i>	-	-	-	-	0.09*	-	0	-
Tanids	<i>Tanid sp.</i>	0.09*	-	-	-	-	-	0	-
Barnacles	<i>Balanus improvisus</i>	13.15	3.39	33.82	3.30	5.99	1.23	3.02	0.40
Decapods	<i>Crangon septemspinosa</i>	-	-	-	-	6.48	2.78	0.10	0.10
	<i>Callinectes sapidus</i>	-	-	-	-	26.09*	-	0.43*	-
	<i>Eurypanopeus depressus</i>	-	-	3.70*	-	49.67	16.33	0.35	0.18
	<i>Rhithropanopeus harrisi</i>								
Fish	<i>Gobiosoma bosc</i>	-	-	95.83	52.32	-	-	0.24	0.09
	<i>Gobiesox strumosus</i>	52.78*	-	-	-	-	-	0.33*	-
	<i>Syngnathus floridae</i>	-	-	14.81*	-	-	-	0.19*	-

* Denotes n = 1

TABLES

Table 1. Mean water depth (m) and range at surf scoter Feeding and Non Feeding sites in the mainstem Chesapeake Bay, MD 2005-2006.

Site	Mean Water Depth (m)	Range (m)
	Feeding	
Herring Bay	2.7	1.8 - 4.0
Poplar Island	4.5	2.4 - 5.2
Shady Side	2.5	1.2 - 5.8
	Non Feeding	
Herring Bay	3.6	1.0 - 4.6
Poplar Island	5	2.1 - 6.7
Shady Side	3.3	1.0 - 6.1

Table 2. Correlation coefficients (r^2) and P values between density and ash-free dry mass (AFDM) for faunal assemblages and species in Chesapeake Bay, 2005-2006.

Faunal Variable	r^2	P
Total fauna	0.97	0.002
Total infauna	0.97	0.002
Total epifauna	0.80	0.002
Bivalves	0.97	0.002
Amphipods	0.77	0.002
Polychaetes	0.83	0.002
Anemones	0.83	0.002
<i>I. recurvum</i>	0.77	0.002
<i>M. lateralis</i>	0.93	0.002
<i>M. balthica</i>	0.91	0.002
<i>M. mitchelli</i>	0.52	0.004
<i>M. arenaria</i>	0.55	0.002
<i>G. gemma</i>	0.93	0.002
<i>B. balanus</i>	0.86	0.002

Table 3. Numbers of surf scoters, long-tailed ducks, and white-winged scoters counted in the mesohaline portion of Chesapeake Bay during winter 2004 – 2005.

	December	February	March	April
Surf Scoter	1,130	10,844	9,404	1,362
Long-tailed Duck	2,989	3,488	3,160	3,512
White-winged Scoter	219	4	213	3

Table 4. Distribution of sediment grain-size proportions of four grain sizes for surf scoter Feeding and Non Feeding sites in Chesapeake Bay. Three locations in the Feeding treatment were entirely hard bottom and contained only trace quantities of sediment, and were not analyzed.

Treatment	$\geq 0.250 \text{ mm}^1$		$\geq 0.125 \text{ mm}^2$		$\geq 0.062 \text{ mm}^3$		Silt/Clay ³	
	\bar{x}	SE	\bar{x}	SE	\bar{x}	SE	\bar{x}	SE
Feeding (n = 27)	0.52	0.05	0.36	0.04	0.06	0.02	0.04	0.01
Non Feeding (n = 30)	0.01	0.03	0.16	0.04	0.66	0.05	0.13	0.02

¹ U.S. Standard Sieve No. 60.

² U.S. Standard Sieve No. 120.

³ U.S. Standard Sieve No. 230.

Table 5. Faunal species identified from surf scoter Feeding and Non Feeding sites. Frequency represents percent occurrence in 90 samples taken from both treatments (all seasons).

Faunal Group	Species	Frequency (%)		Faunal Group	Species	Frequency (%)			
		Feeding	Non Feeding			Feeding	Non Feeding		
Cnidarians	<i>Haliplanella lineata</i>	3	0	Amphipods	<i>Incisocalliope aestuarius</i>	8	2		
	<i>Diadumene leucolena</i>	24	8		<i>Leptocheirus plumulosus</i>	1	17		
	<i>Edwardsia elegans</i>	1	0		<i>Corophium lacustre</i>	18	11		
Polychaetes	<i>Glycera sp.</i>	3	22		<i>Monoculodes edwardsi</i>	16	10		
	<i>Glycinde solitaria</i>	2	8		<i>Gammarus mucronatus</i>	1	0		
	<i>Neanthes succinea</i>	53	80		<i>Haustoridae sp.</i>	21	2		
	<i>Pectinaria gouldi</i>	10	3		<i>Amphipod sp.^c</i>	6	11		
	<i>Polydora cornuta</i>	4	4		Mysids	<i>Americamysis bigelowi</i>	1	0	
	<i>Heteromastus filiformis</i>	1	52			Isopods	<i>Cyathura polita</i>	0	4
	<i>Nephtys sp.</i>	7	1				<i>Edotia triloba</i>	3	7
	<i>Eteone sp.</i>	8	10	<i>Chiridotea almyra</i>	1		0		
	Nemeritans	<i>Polychaete sp. 1^a</i>	31	42	Tanids	<i>Tanid sp.</i>	0	1	
<i>Polychaete sp. 2^{b,c}</i>		3	16	Barnacles		<i>Balanus improvisus</i>	38	22	
<i>Micrura leidyi</i>		8	4		Decapods	<i>Crangon septemspinosus</i>	1	1	
<i>Nemertan sp.</i>	3	11	<i>Callinectes sapidus</i>	1		0			
Gastropods	<i>Sayella chesapeakea</i>	8	10	<i>Eurypanopeus depressus</i>		2	0		
	<i>Boonea sp.</i>	1	0	<i>Rhithropanopeus harrisi</i>		4	3		
Bivalves	<i>Mulinia lateralis</i>	48	58	<i>Decapod sp.^c</i>		2	0		
	<i>Gemma gemma</i>	41	33	Fish	<i>Gobiosoma bosc</i>	1	2		
	<i>Macoma balthica</i>	30	79		<i>Gobiesox strumosus</i>	1	0		
	<i>Macoma mitchelli</i>	56	89		<i>Syngnathus floridae</i>	0	1		
	<i>Mya arenaria</i>	20	27						
	<i>Ischadium recurvum</i>	30	18						

^a Represents single species not identified to genus or species across samples.

^b Represents multiple species not identified to genus or species.

^c Not included in analyses of treatment species numbers.

Table 6. Mean species richness (S) and Shannon-Weiner diversity index (H') and standard errors (SE) for surf scoter Feeding and Non Feeding sites.

Treatment	Site	S	SE	H'	SE
Feeding					
	Herring Bay	4.63	0.47	0.86	0.09
	Poplar Island	7.00	0.64	0.77	0.10
	Shady Side	4.30	0.45	1.00	0.10
	Treatment Mean	5.31	0.33	0.87	0.06
Non Feeding					
	Herring Bay	6.30	0.40	1.13	0.08
	Poplar Island	7.93	0.57	1.24	0.10
	Shady Side	6.67	0.53	1.11	0.08
	Treatment Mean	6.97	0.30	1.16	0.05

Table 7. Statistical results from pairwise comparisons of shell mass to tissue AFDM (pooled across season) using a one-way ANOVA.

Species	Gemclam		Baltic macoma		Mitchell's macoma		Dwarf surfclam	
	t	P	t	P	t	P	t	P
Hooked mussel	-3.46	0.005	6.49	< 0.001	12.29	< 0.001	4.47	< 0.001
Gemclam	**	**	11.57	< 0.001	-18.29	< 0.001	8.98	< 0.001
Baltic macoma	**	**	**	**	-7.75	< 0.001	2.89	0.031
Mitchell's macoma	**	**	**	**	**	**	18.29	< 0.001

Table 8. Statistical results from pairwise comparisons of bivalve total ash to AFDM (pooled across season) using a one-way ANOVA.

Species	Gemclam		Baltic macoma		Mitchell's macoma		Dwarf surfclam	
	t	P	t	P	t	P	t	P
Hooked mussel	0.93	0.886	5.09	< 0.001	9.80	< 0.001	4.00	0.001
Gemclam	**	**	-4.51	0.001	-9.80	< 0.001	-3.05	0.008
Baltic macoma	**	**	**	**	-6.34	< 0.001	1.36	0.654
Mitchell's macoma	**	**	**	**	**	**	7.60	< 0.001

Table 9. Mean density and frequency of occurrence comparisons of two benthic sampling methods; grab and diver.

Faunal Group	Species	Density (m ⁻²)		Frequency (%)	
		Diver	Grab	Diver	Grab
Cnidarians					
	<i>Diadumene leucolena</i>	7.0	22.8	55	55
	<i>Edwardsia elegans</i>	0	0.3	0	5
Polychaetes					
	<i>Glycinde solitaria</i>	0.05	0	5	0
	<i>Neanthes succinea</i>	26	44.7	85	60
	<i>Pectinaria gouldi</i>	0.05	1.2	5	15
	<i>Polydora</i> sp.	126.5	0.7	30	10
	<i>Eteone</i> sp.	0	1.2	0	15
	<i>Polychaete</i> sp. 1 ^a	0.35	2.2	25	20
Nemeritans					
	<i>Micrura leidyi</i>	0	0.9	0	15
	<i>Nemertan</i> sp.	0.15	0.6	15	10
Gastropods					
	<i>Sayella chesapeakea</i>	0.05	1.5	5	20
	<i>Boonea impressa</i>	0.05	0	5	0
	<i>Boonea</i> sp. ^b	0	0.3	0	5
Bivalves					
	<i>Mulinia lateralis</i>	1.8	60	40	50
	<i>Gemma gemma</i>	754.5	26,270	60	75
	<i>Macoma balthica</i>	7.7	30	90	75
	<i>Macoma mitchelli</i>	0.4	4.7	15	50
	<i>Mya arenaria</i>	2.3	15.3	60	55
	<i>Ischadium recurvum</i>	7.0	6.9	55	35
Amphipods					
	<i>Incisocalloipe aestuarius</i>	1.5	13.65	35	20
	<i>Corophium lacustre</i>	339.3	66.9	60	40
	<i>Monoculodes edwardsi</i>	0	2.8	0	25
	<i>Gammarus mucronatus</i>	0	0.5	0	5
	<i>Haustoridae</i> sp.	0	0.5	0	5
	<i>Amphipod</i> sp. ^a	0.05	1.9	5	15
Isopods					
	<i>Edotia triloba</i>	0	0.3	0	5
	<i>Sphaeroma quadridentatum</i>	.05	0	5	0
Barnacles					
	<i>Balanus improvisus</i>	142.9	791.7	60	90
Decapods					
	<i>Crangon septemspinosa</i>	0	0.3	0	5
	<i>Panopeus herbstii</i>	0.05	0	5	0
	<i>Callinectes sapidus</i>	0	0.3	0	5
	<i>Eurypanopeus depressus</i>	0.05	0.3	5	5
	<i>Rhithropanopeus harrisi</i>	0.2	0.3	15	5
	<i>Decapod</i> sp. ^a	0.05	0	5	0

^a Not included in technique species counts

^b Included in grab species counts, but not diver

FIGURES

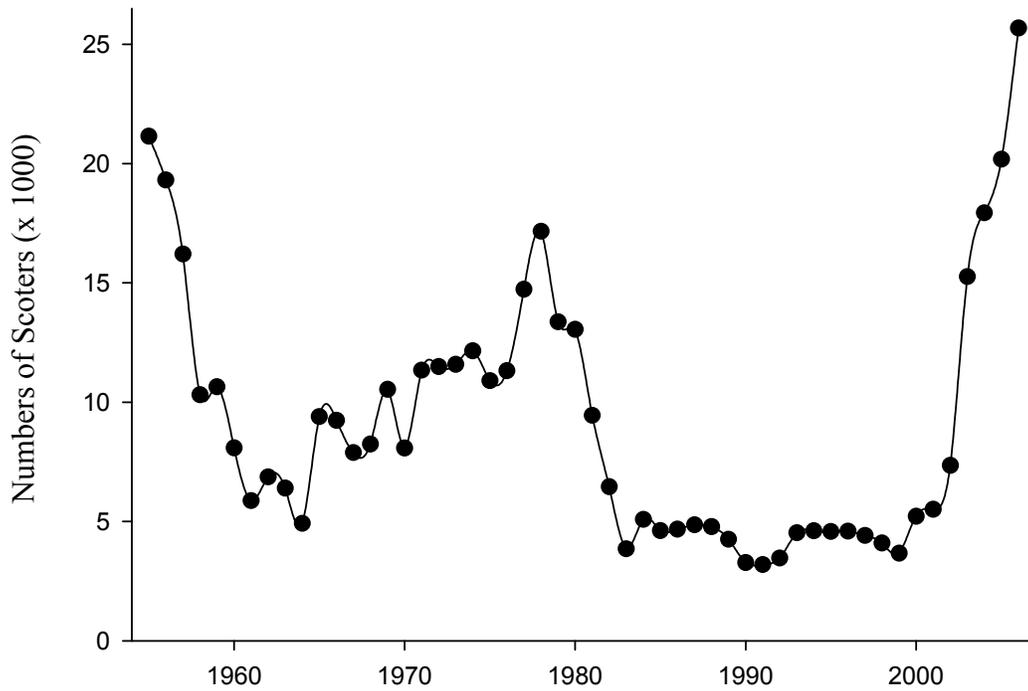


Figure 1. Five-year running average of scoter (*Melanitta* spp.) populations in Chesapeake Bay from mid-winter waterfowl surveys 1955-2006. All survey segments of the Bay and major tributaries, in both Maryland and Virginia were included, and represent surf, black, and white-winged scoters.

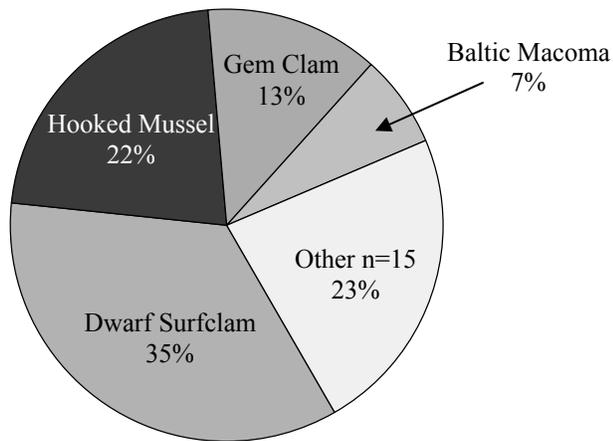


Figure 2. Food habits of surf scoters (n=246) obtained by hunters in Chesapeake Bay between 1999-2006, as determined by mean percent volume of dried contents. “Other” represents 15 species or faunal groups which, individually, represented <5% of total mean volume. Figure adapted from Perry et al. (2007).

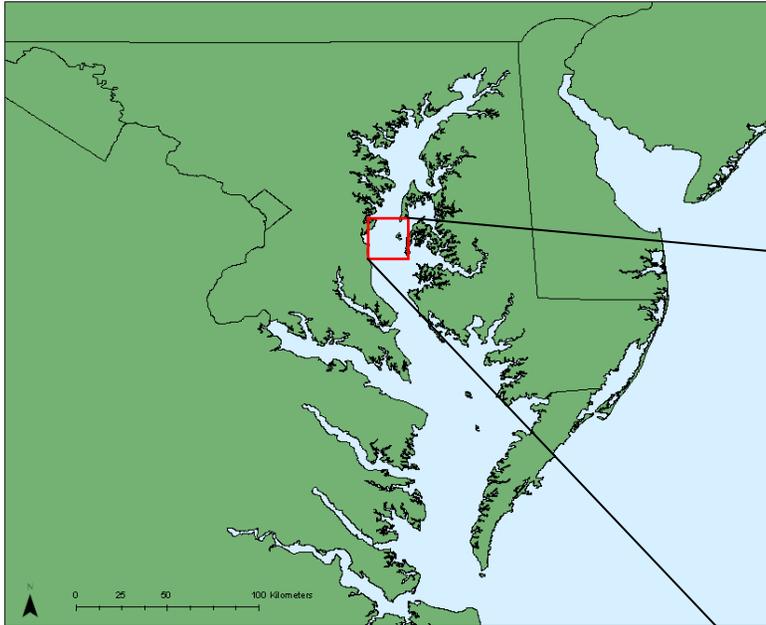


Figure 3. Regional map of Chesapeake Bay and surrounding area showing location of the study area in the Maryland portion of the Bay.

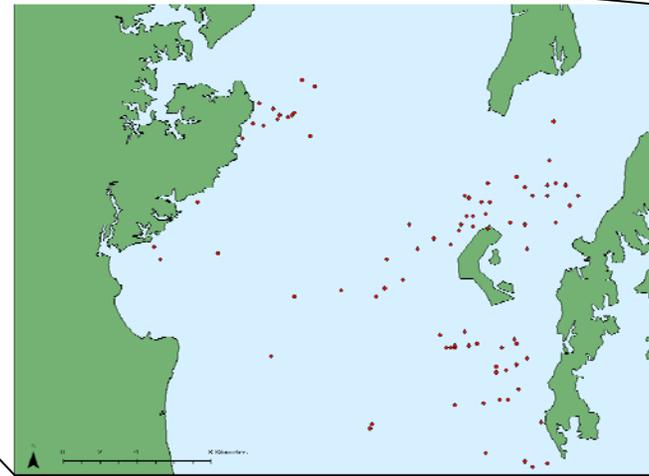


Figure 4. Map of study area showing individual surf scoter locations via satellite telemetry.

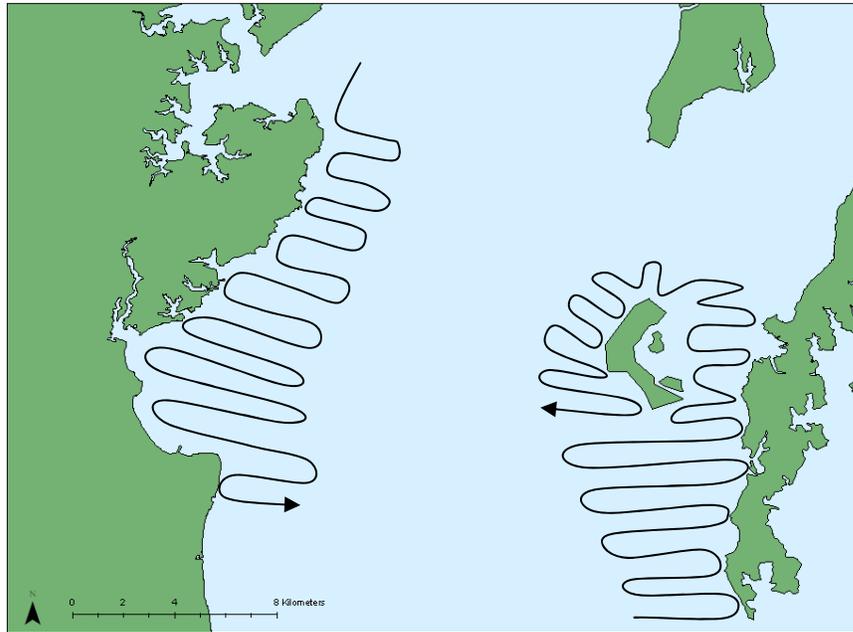


Figure 5. Stylized depiction of boat survey route to determine surf scoter use of the study area. Route was altered, when needed, in order to determine the exact location of scoter flocks with GPS.

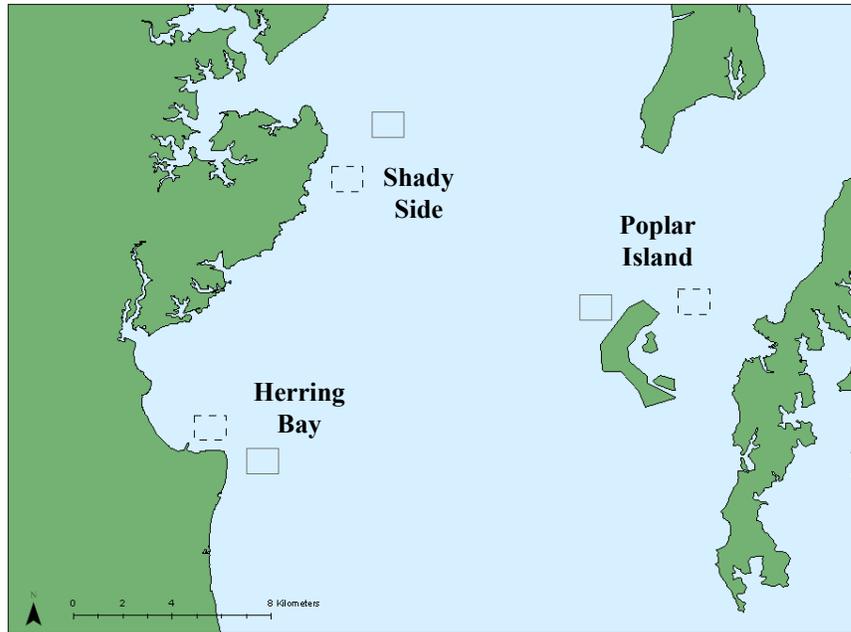


Figure 6. Location of study sites within the mesohaline portion of the mainstem Chesapeake Bay, MD. Solid outlined boxes represent surf scoter Feeding sites, whereas dashed outlined boxes represent Non Feeding sites.



Figure 7. Unsorted (pre-analysis, sediment removed) sample containing high densities of gemclam (*Gemma gemma*) allowing for sub-sampling of density and biomass. Deceased gemclam shell represents a significant portion of the sample, by volume, with live gemclam accounting for the majority of fauna present.

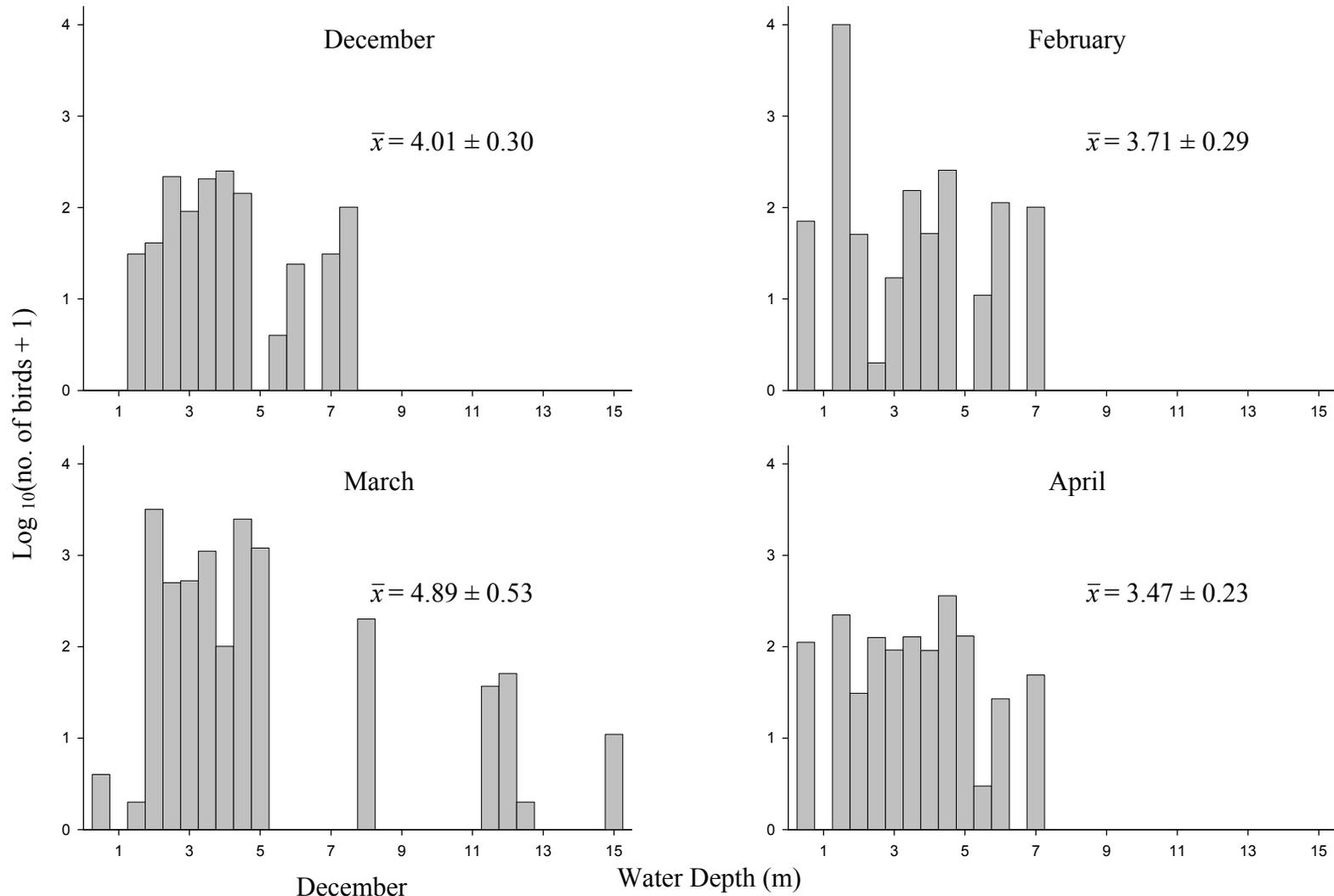


Figure 8. Distribution of surf scoter numbers [$\log(y + 1)$] by water depth in study area for four months during winter of 2004 – 2005. Numbers represent the sum of all surf scoters counted at a water depth and includes all scoters observed.

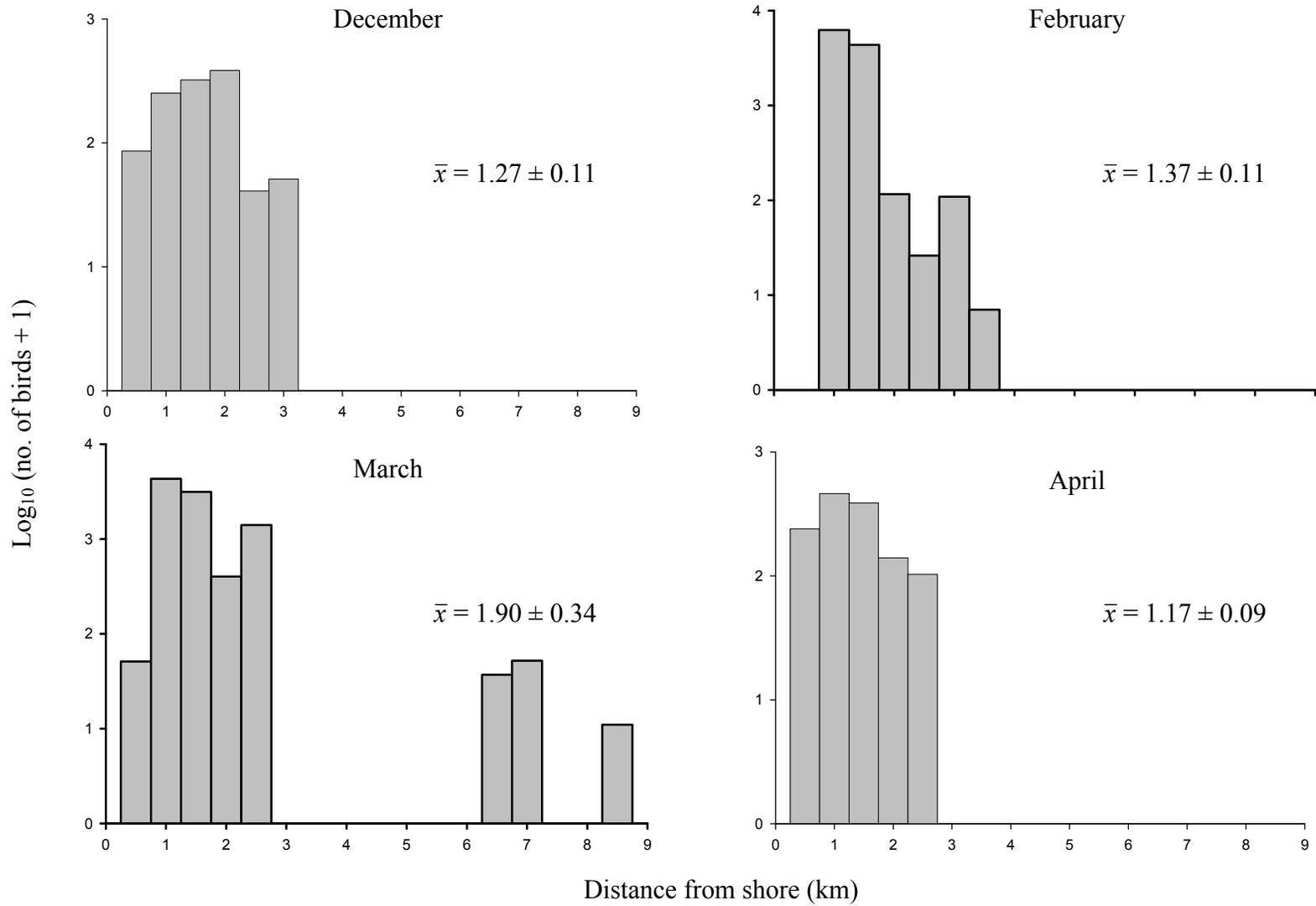


Figure 9. Distribution of scoter numbers [$\log(y + 1)$] by distance from shore in mesohaline portion of the mainstem Chesapeake Bay for 4 survey periods during winter 2004-2005. Distribution includes all scoters observed.

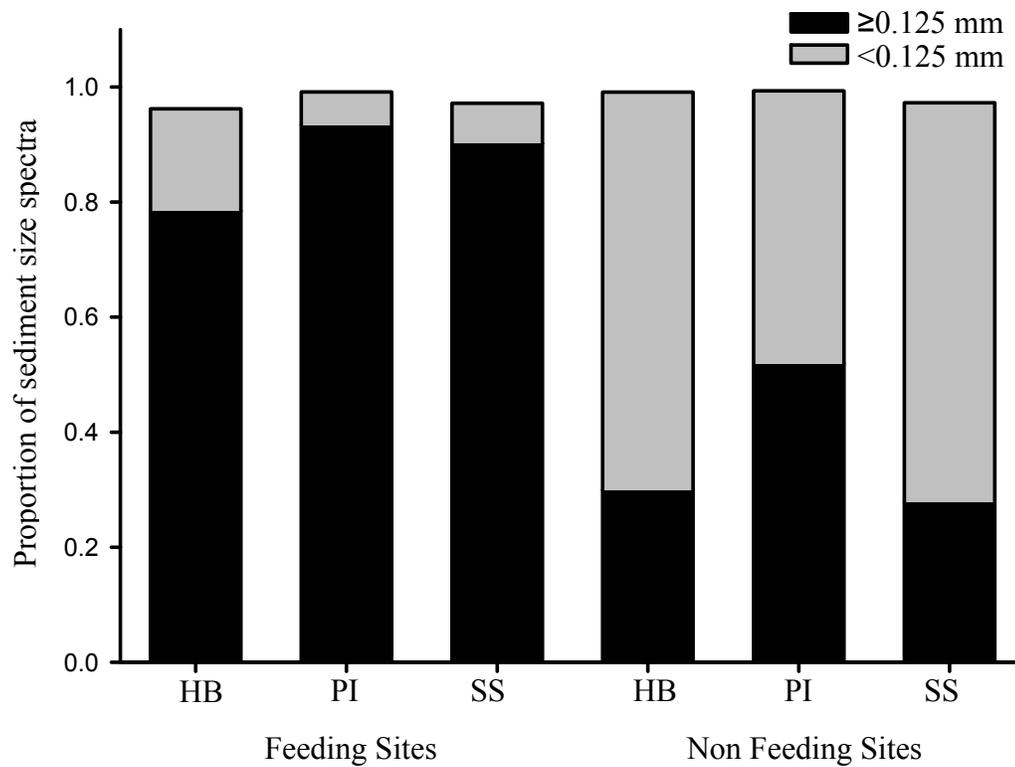


Figure 10. Mean proportion of pooled sediment grain-sizes for each study site and grouped by treatment. ≥ 0.125 mm grouping contains contents of 0.125 mm and 0.250 mm sieve contents. < 0.125 mm grouping contains combined contents of 0.062 mm sieve and silt/clay portion. HB = Herring Bay, PI = Poplar Island, and SS = Shady Side.

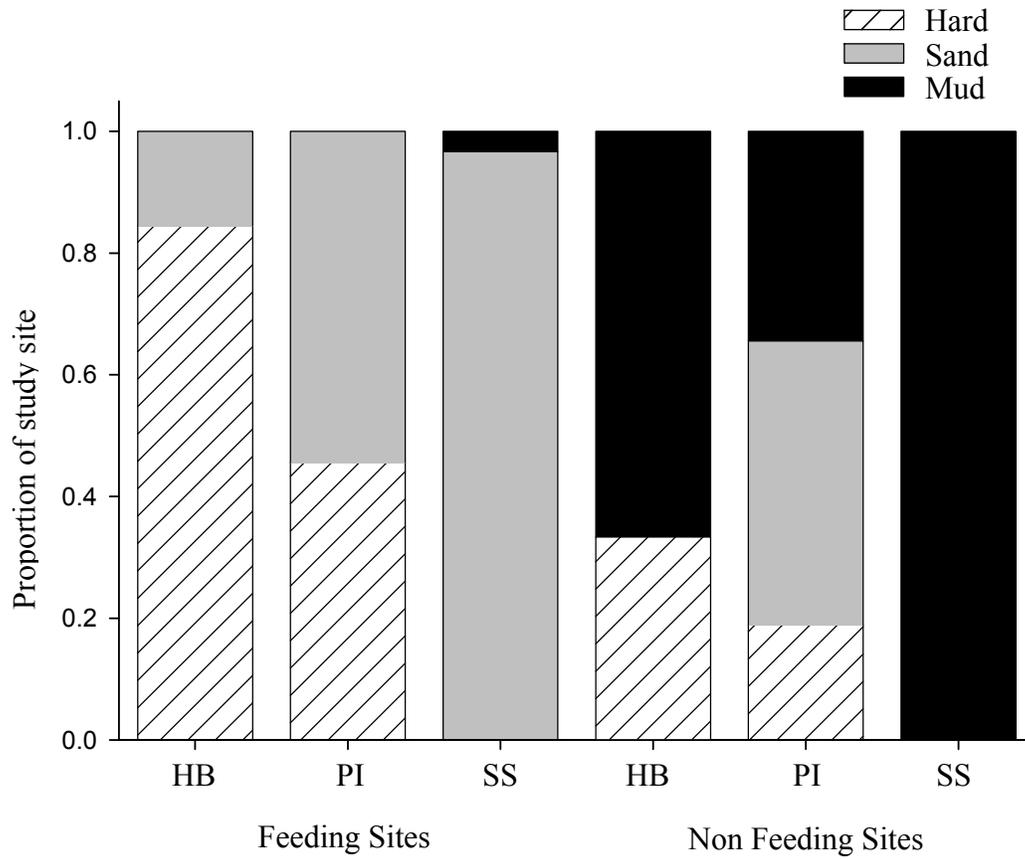


Figure 11. Proportion of categorized habitats at surf scoter Feeding and Non Feeding sites. Categorizations were based on diver observations, grab contents, and results of sediment grain-size analysis. HB = Herring Bay, PI = Poplar Island, and SS = Shady Side.

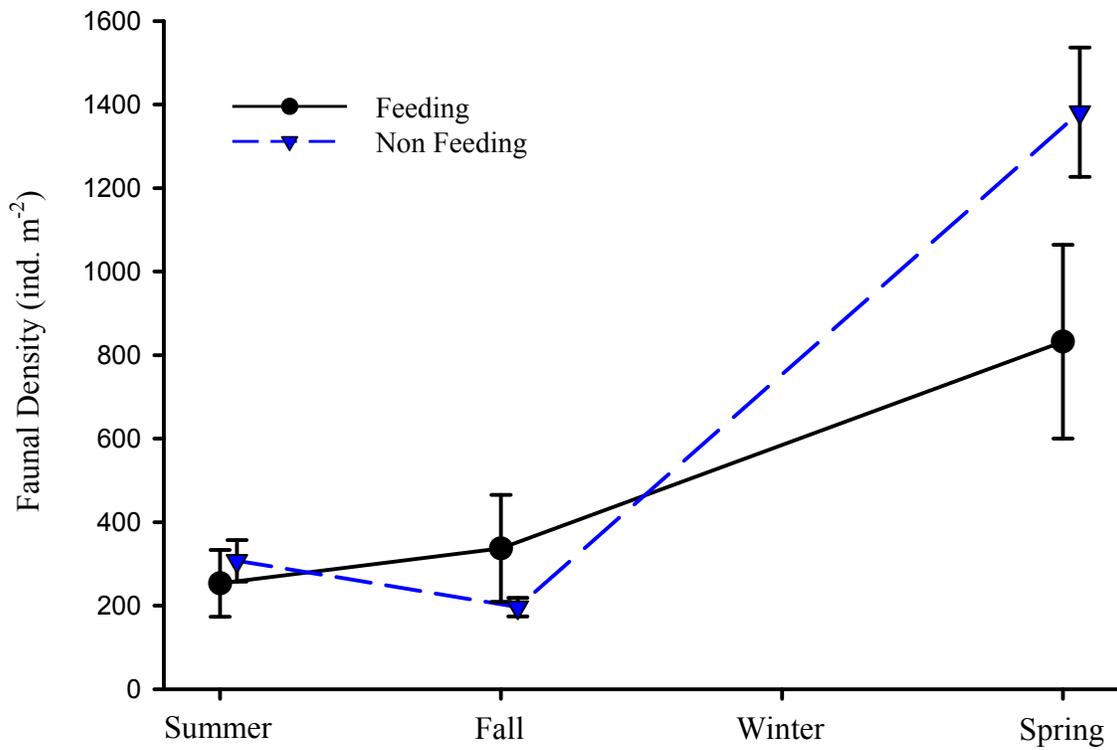


Figure 12. Mean seasonal density (ind. m⁻² ± SE) of total fauna sampled (gemclam adjusted) at surf scoter Feeding and Non Feeding sites. Treatment differences were found only during the spring ($t = -1.97$, $P = 0.044$), while seasonal differences were found between fall and spring within both treatments.

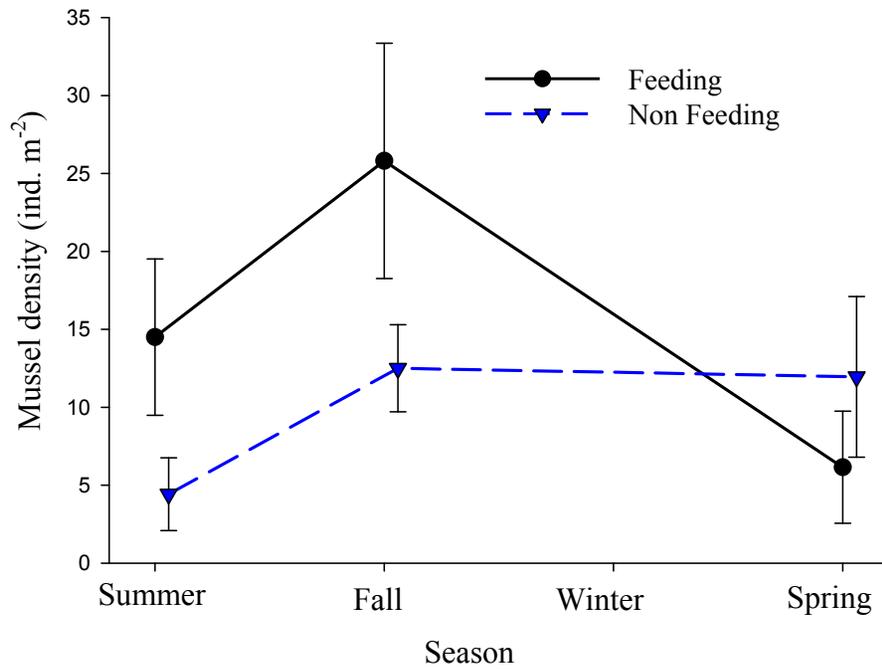


Figure 13. Seasonal densities (ind. m⁻² ± SE) of hooked mussel (*Ischadium recurvum*) at surf scoter Feeding and Non Feeding sites.

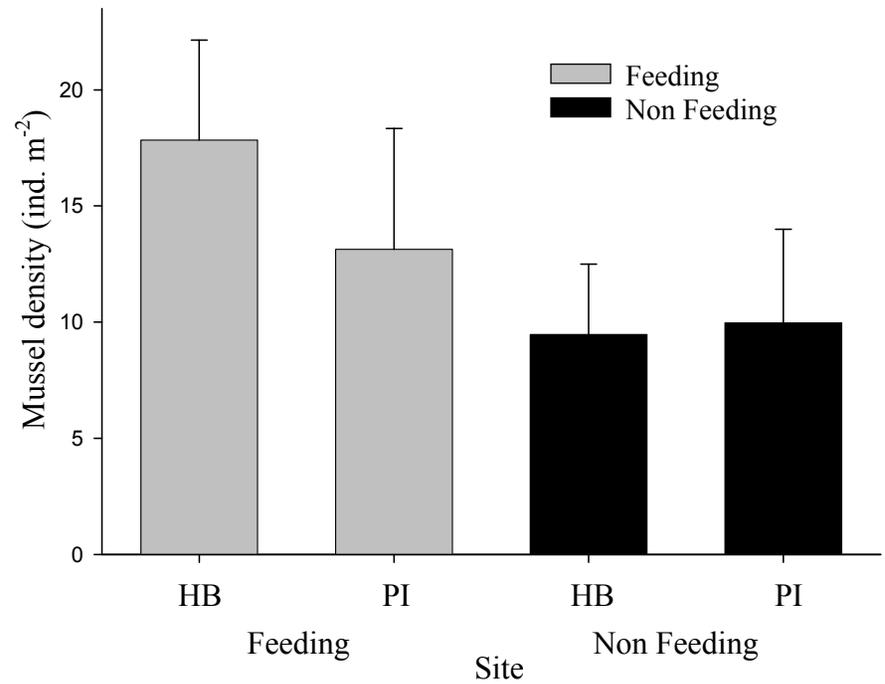


Figure 14. Location of hooked mussel (*Ischadium recurvum*) densities (ind. m⁻² ± SE) at surf scoter Feeding and Non Feeding sites. A third location, Shady Side, did not contain mussels at either treatment and is not included. HB = Herring Bay and PI = Poplar Island.

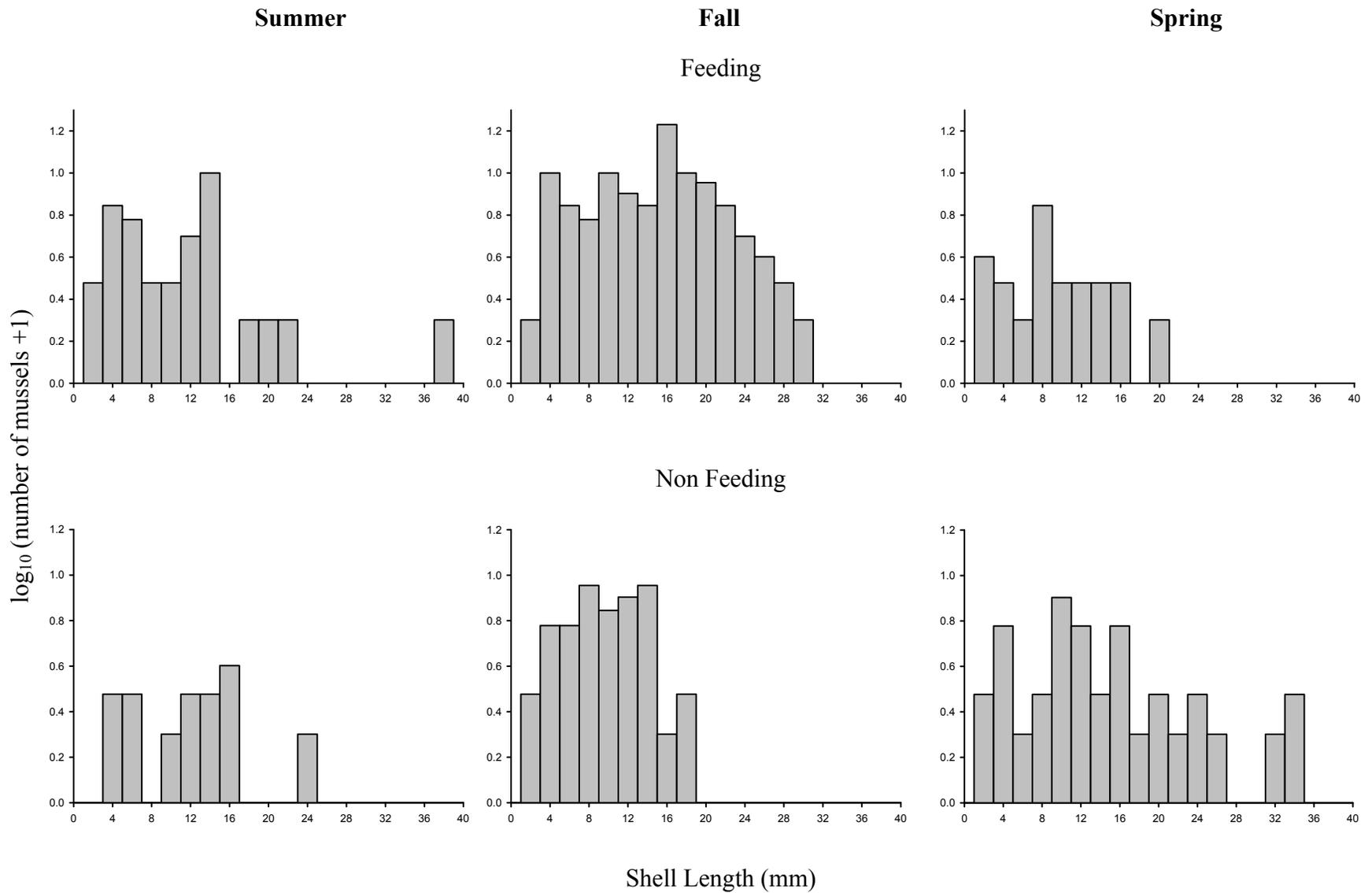


Figure 15. Size distribution of hooked mussel (*Ischadium recurvum*) sorted into 2 mm size classes for surf scoter Feeding and Non Feeding sites during sampling in summer, fall, and spring.

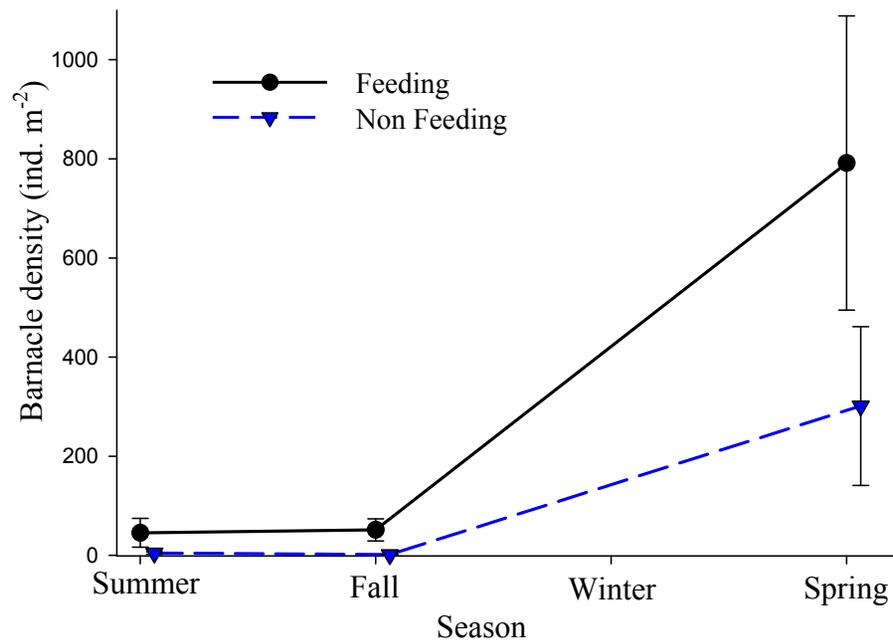


Figure 16. Seasonal densities (ind. m⁻² ± SE) of bay barnacle (*Balanus improvisus*) at surf scoter Feeding and Non Feeding sites.

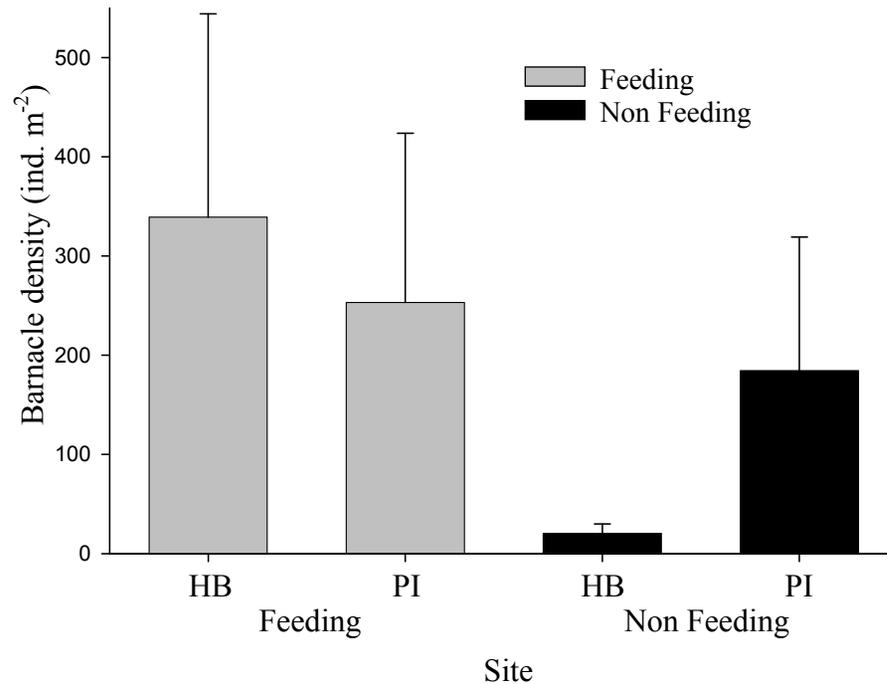


Figure 17. Location of bay barnacle (*Balanus improvisus*) densities (ind. m⁻² ± SE) at surf scoter Feeding and Non Feeding sites. No barnacles were found at Shady Side (not shown). HB = Herring Bay and PI = Poplar Island.

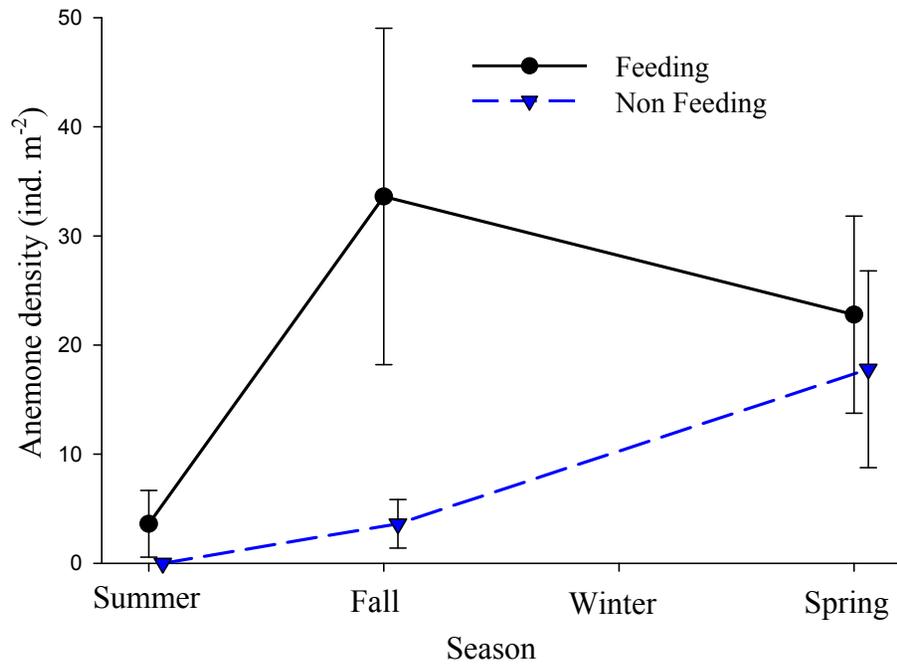


Figure 18. Seasonal densities (ind. m⁻² ± SE) of white anemones (*Diadumene leucolena*) at surf scoter Feeding and Non Feeding sites.

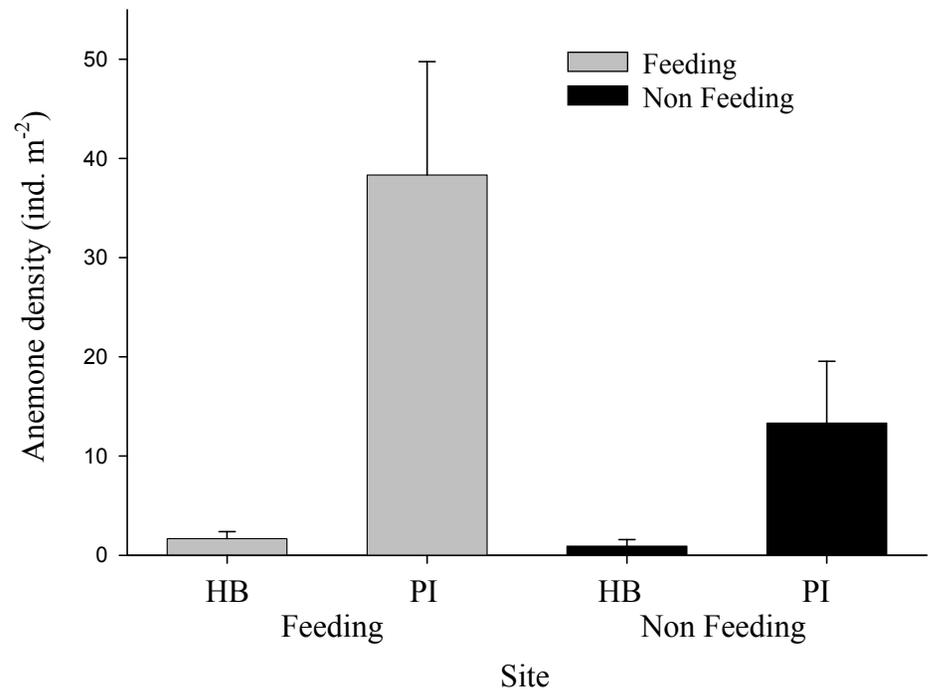


Figure 19. Location of white anemone (*Diadumene leucolena*) densities (ind. m⁻² ± SE) at surf scoter Feeding and Non Feeding sites. No anemones were found at Shady Side (not shown). HB = Herring Bay and PI = Poplar Island.

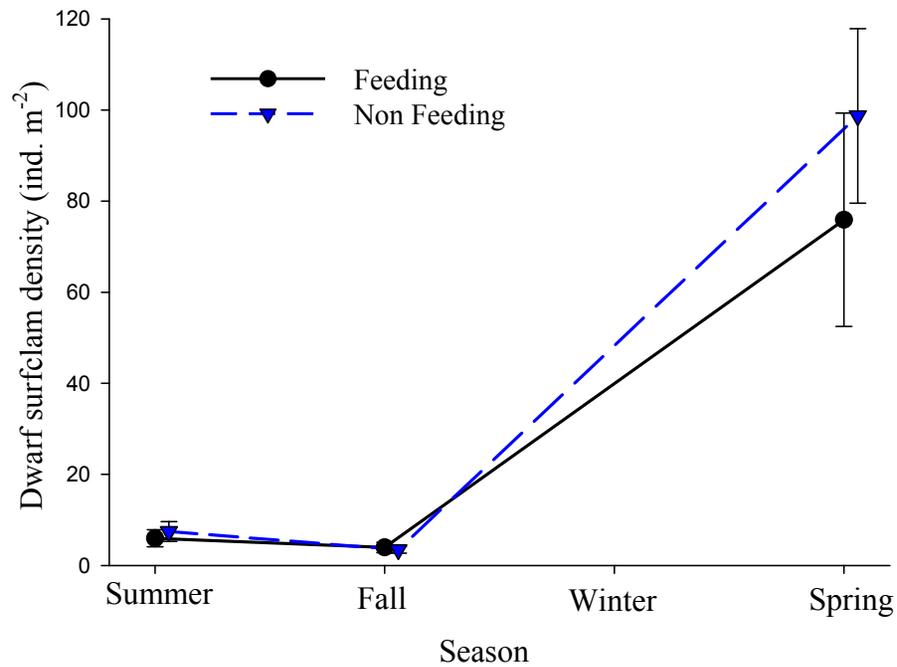


Figure 20. Seasonal densities (ind. m⁻² ± SE) of dwarf surfclam (*Mulinia lateralis*) at surf scoter Feeding and Non Feeding sites.

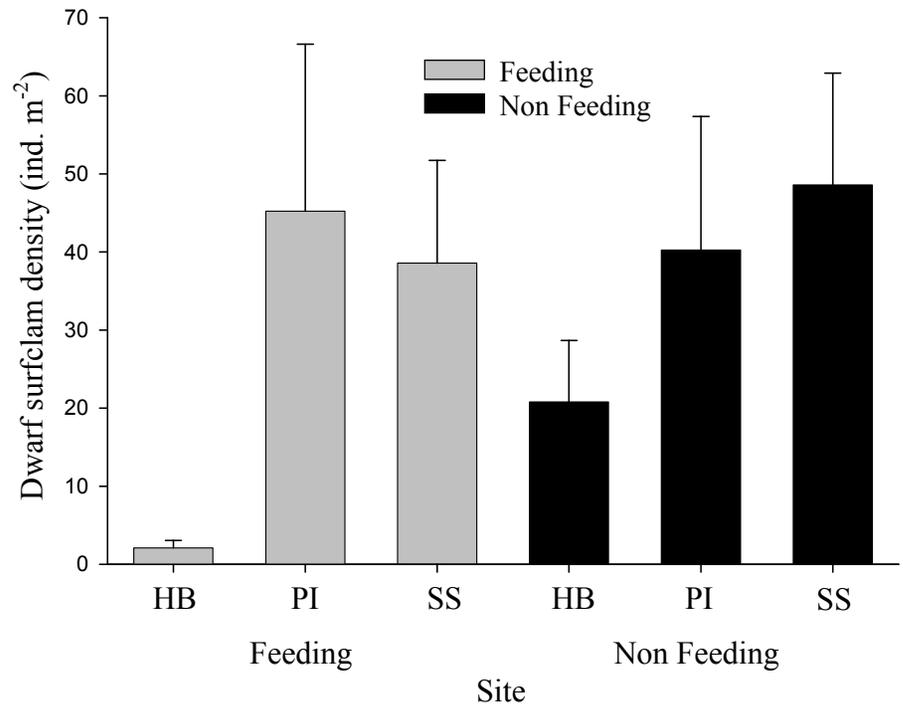


Figure 21. Location of dwarf surfclam (*Mulinia lateralis*) densities (ind. m⁻² ± SE) at surf scoter Feeding and Non Feeding sites. HB = Herring Bay, PI = Poplar Island, and SS = Shady Side

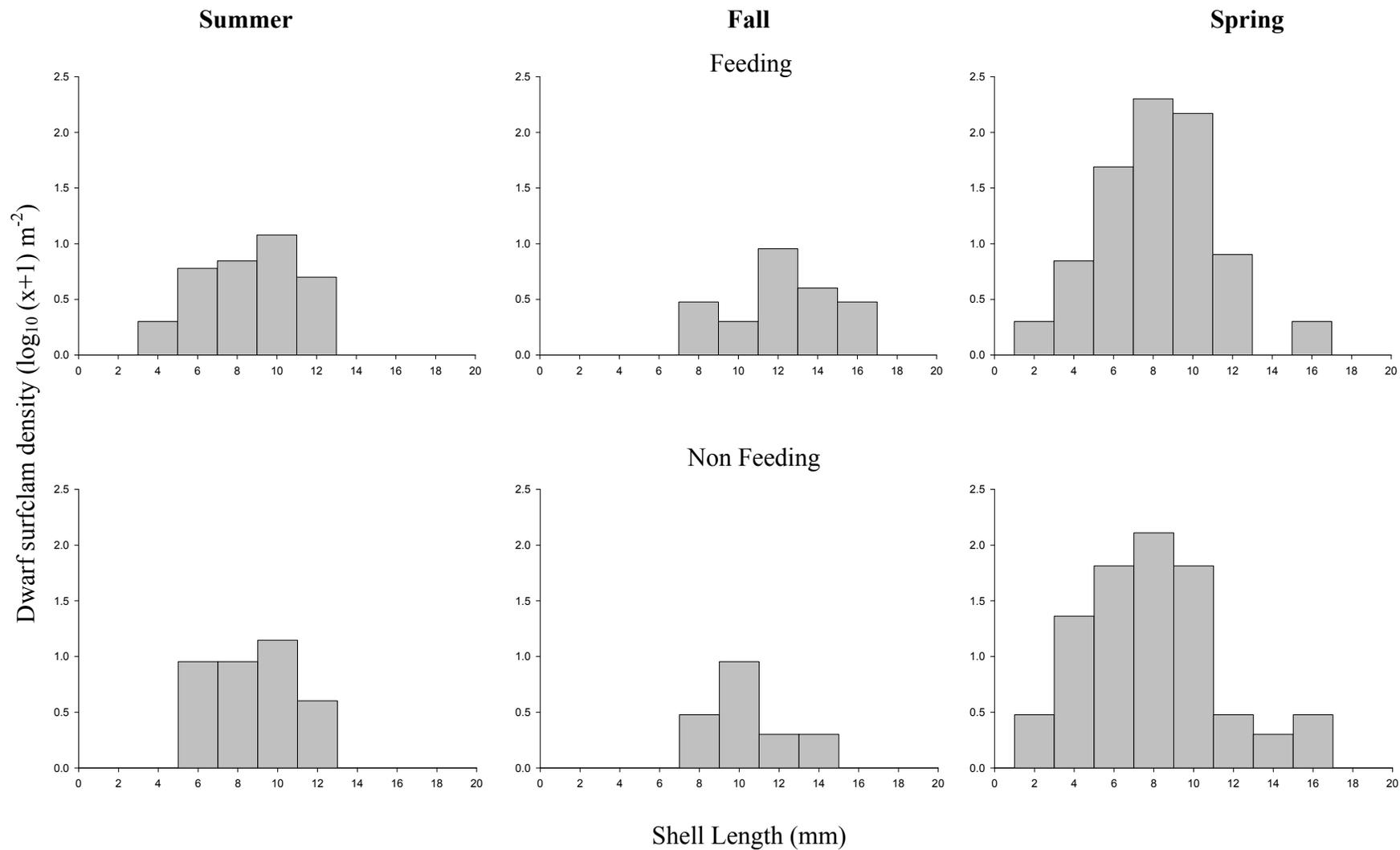


Figure 22. Size distribution of dwarf surfclam (*Mulinia lateralis*) sorted into 2 mm size classes for surf scoter Feeding and Non Feeding sites during sampling in summer, fall, and spring.

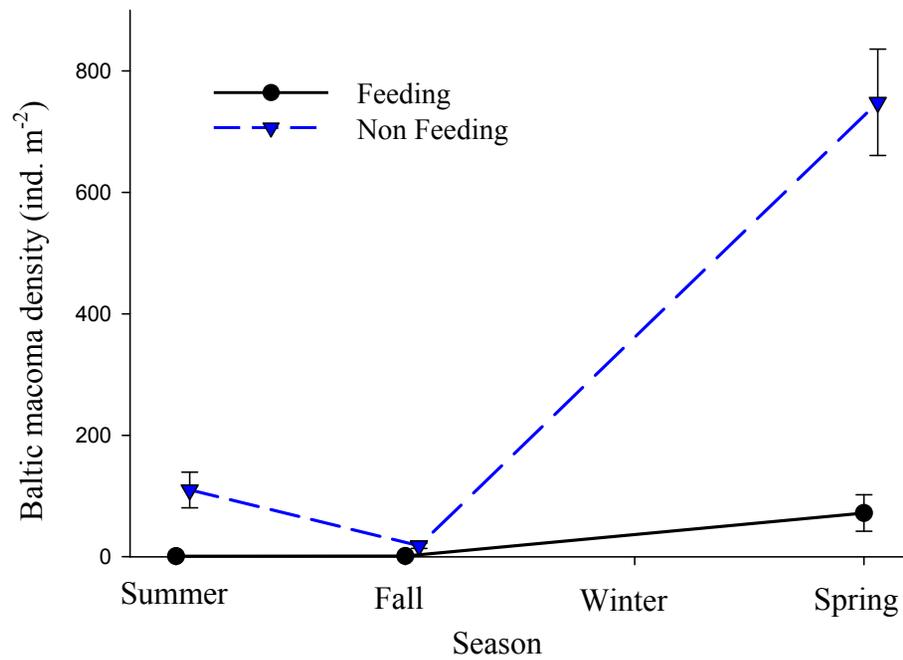


Figure 23. Seasonal densities (ind. m⁻² ± SE) of Baltic macoma (*Macoma balthica*) at surf scoter Feeding and Non Feeding sites.

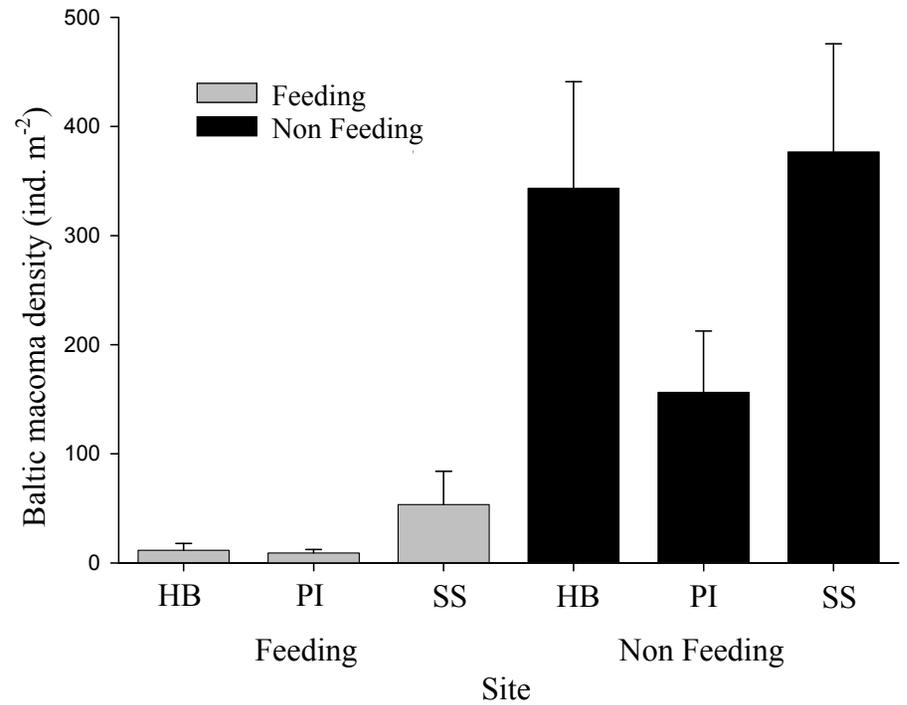


Figure 24. Location of Baltic macoma (*Macoma balthica*) densities (ind. m⁻² ± SE) at surf scoter Feeding and Non Feeding sites. HB = Herring Bay, PI = Poplar Island, and SS = Shady Side.

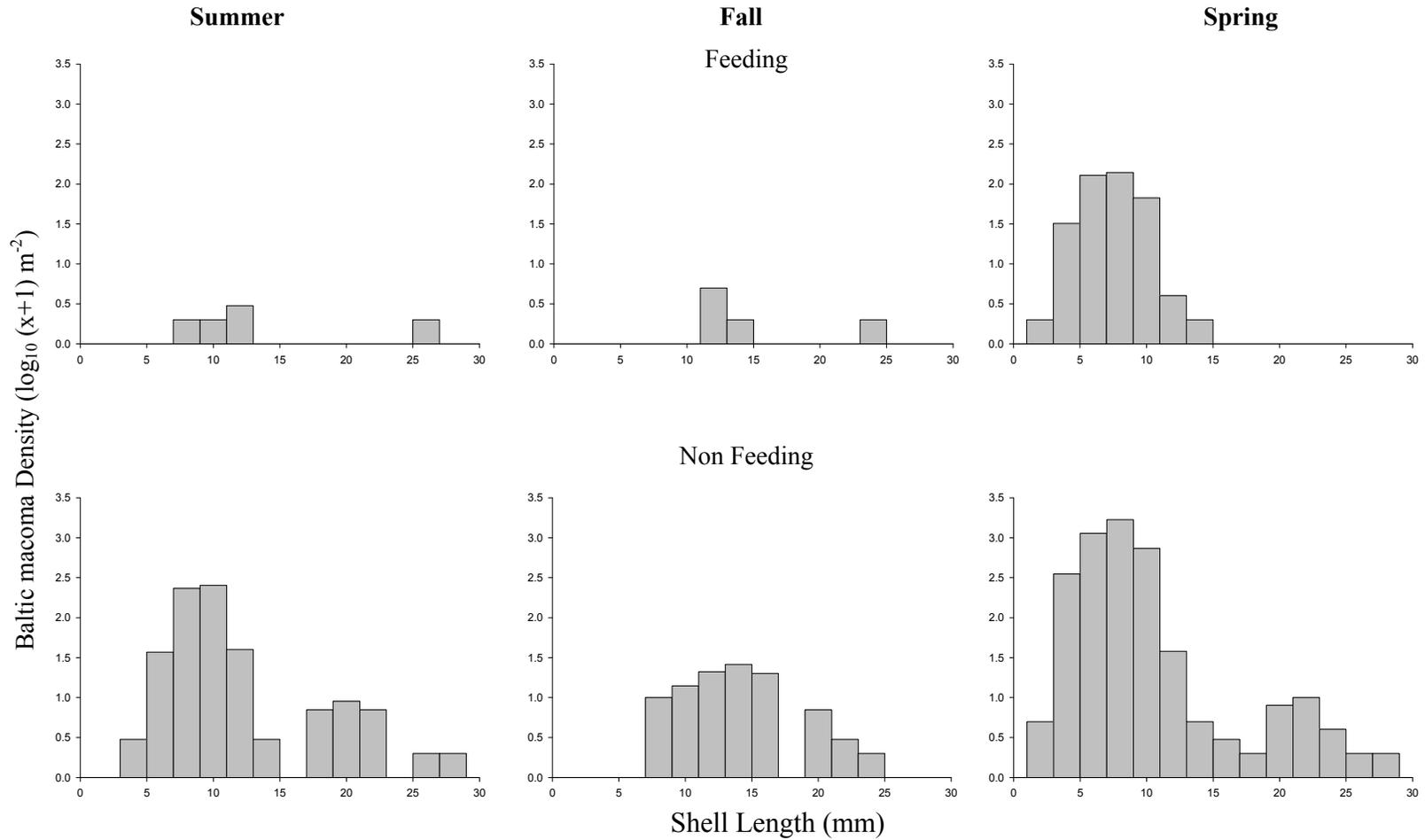


Figure 25. Size distribution of Baltic macoma (*Macoma balthica*) sorted into 2 mm size classes at surf scoter Feeding and Non Feeding sites during summer, fall, and spring from 2004-2005.

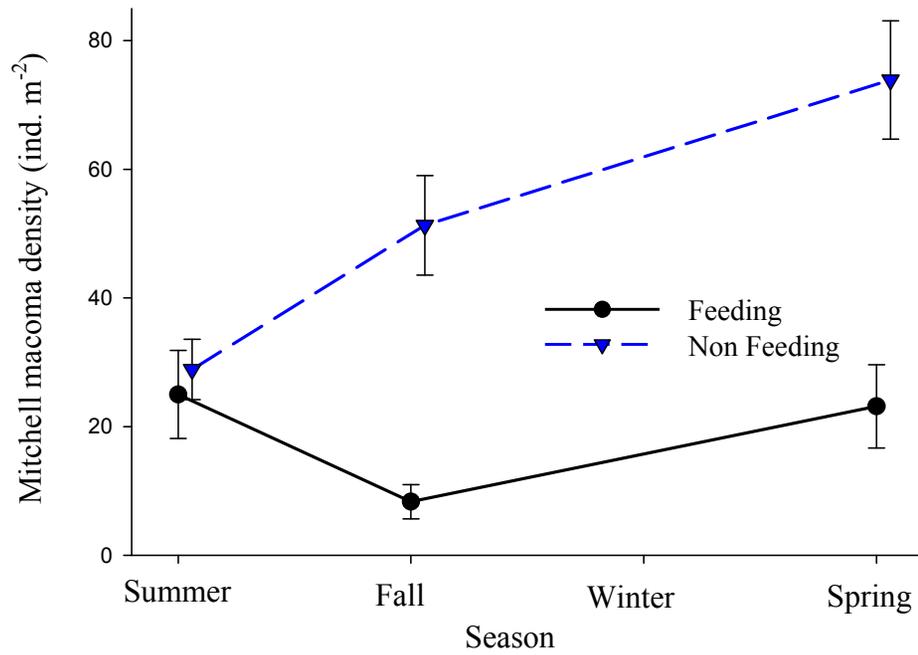


Figure 26. Seasonal densities (ind. m⁻² ± SE) of mitchell macoma (*Macoma mitchelli*) at surf scoter Feeding and Non Feeding sites.

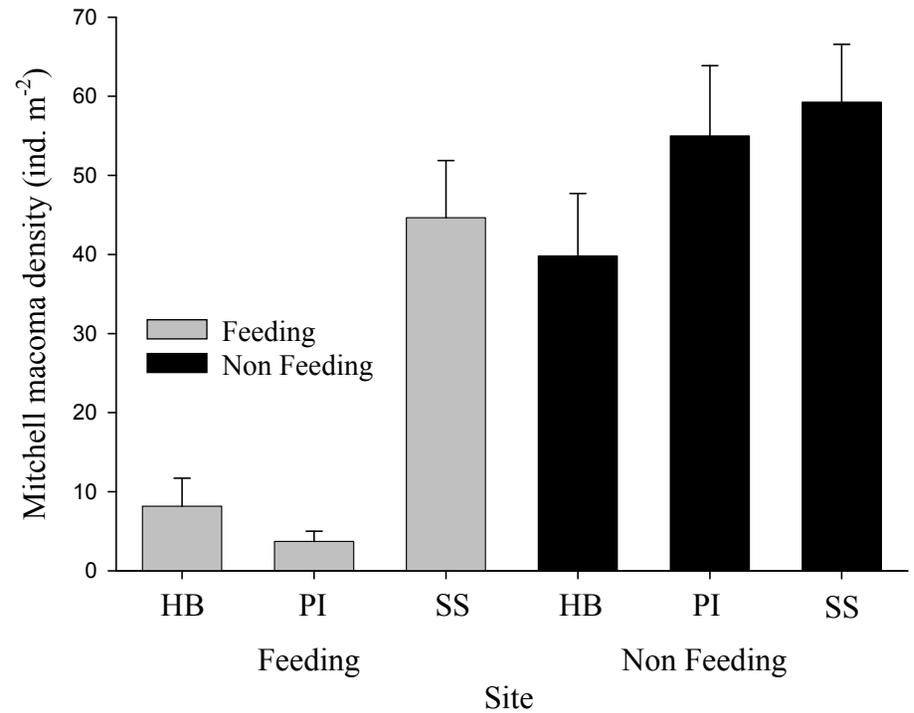


Figure 27. Location of mitchell macoma (*Macoma mitchelli*) densities (ind. m⁻² ± SE) at surf scoter Feeding and Non Feeding sites. HB = Herring Bay, PI = Poplar Island, and SS = Shady Side.

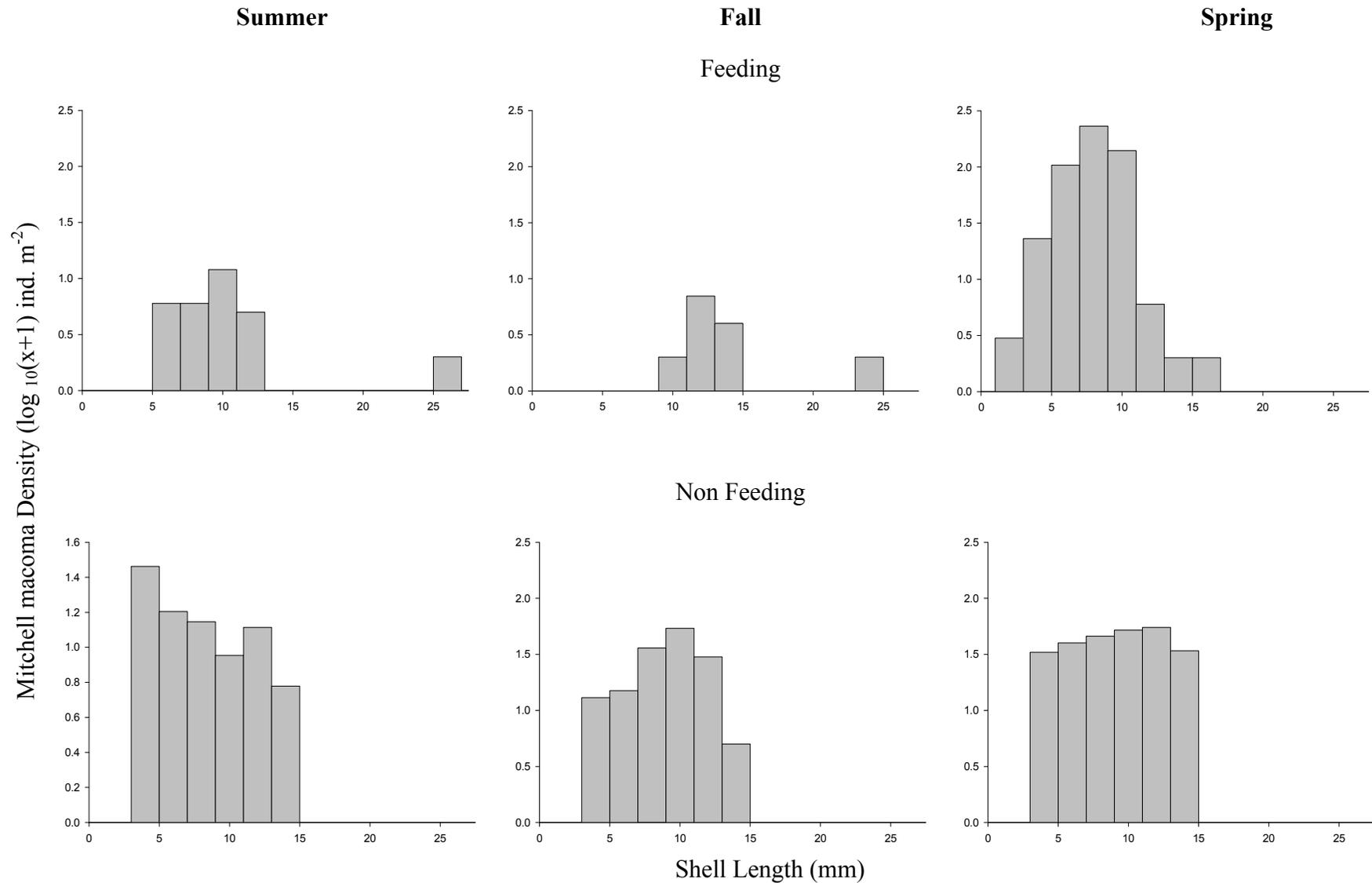


Figure 28. Size distribution of mitchell's macoma (*Macoma mitchelli*) sorted into 2 mm size classes for surf scoter Feeding and Non Feeding sites during summer, fall, and spring from 2004-2005

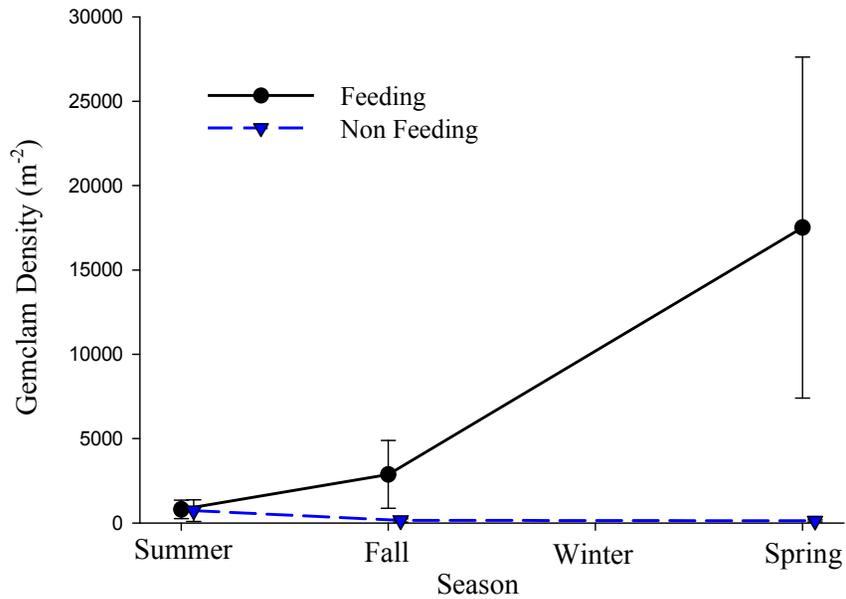


Figure 29. Seasonal densities (ind. m⁻² ± SE) of gemclam (*Gemma gemma*) at surf scoter Feeding and Non Feeding sites.

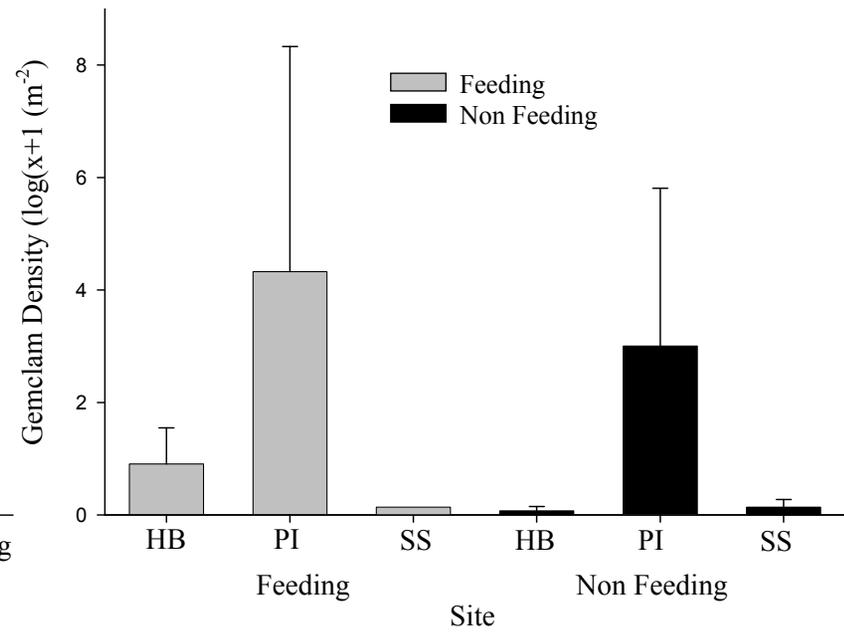


Figure 30. Location of gemclam (*Gemma gemma*) densities (ind. m⁻² ± SE) at surf scoter Feeding and Non Feeding sites. HB = Herring Bay, PI = Poplar Island, and SS = Shady Side.

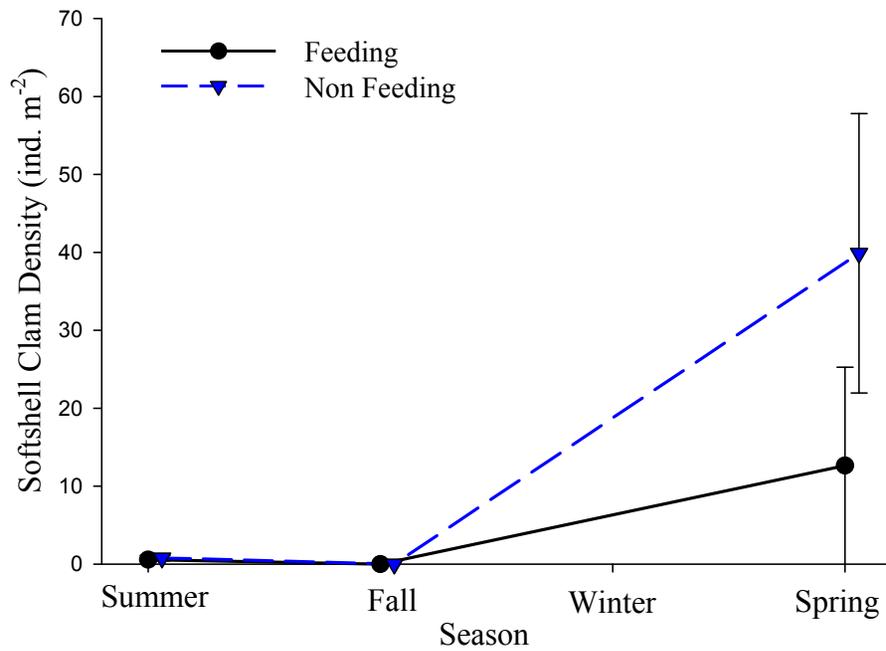


Figure 31. Seasonal densities (ind. m⁻² ± SE) of softshell clam (*Mya arenaria*) at surf scoter Feeding and Non Feeding sites.

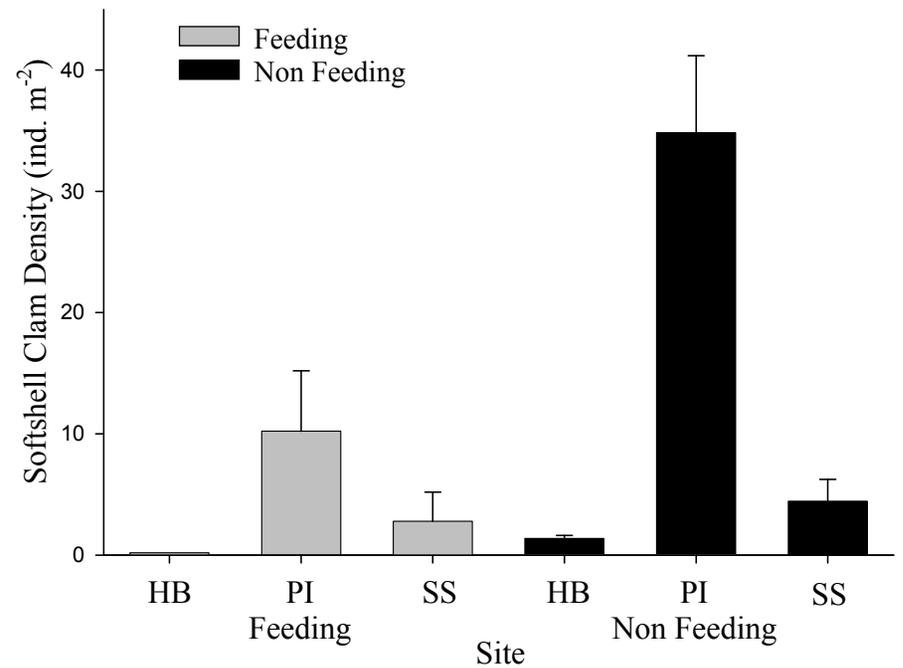


Figure 32. Location of softshell clam (*Mya arenaria*) densities (ind. m⁻² ± SE) at surf scoter Feeding and Non Feeding sites. HB = Herring Bay, PI = Poplar Island, and SS = Shady Side.

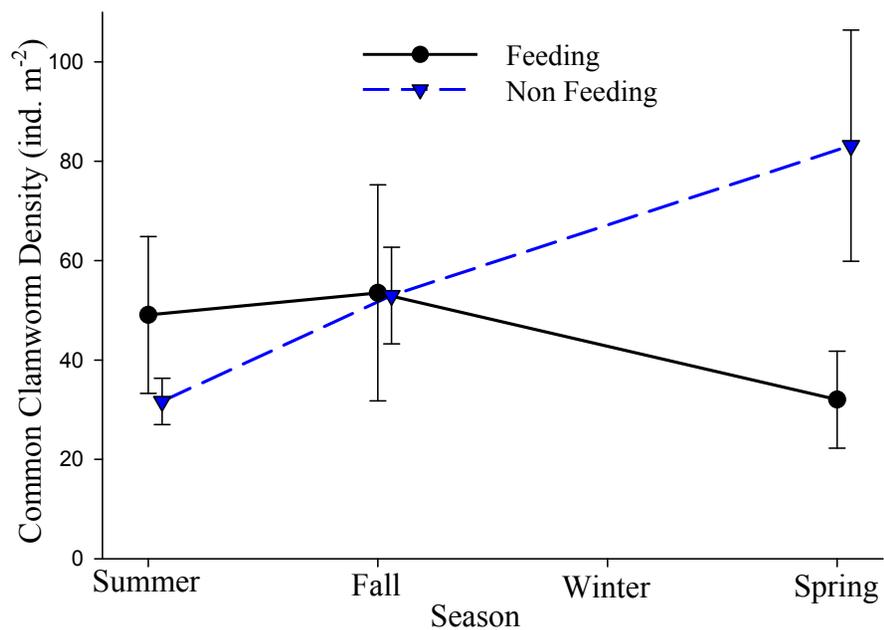


Figure 33. Seasonal densities (ind. m⁻² ± SE) of common clamworm (*Neanthes succinea*) at surf scoter Feeding and Non Feeding sites.

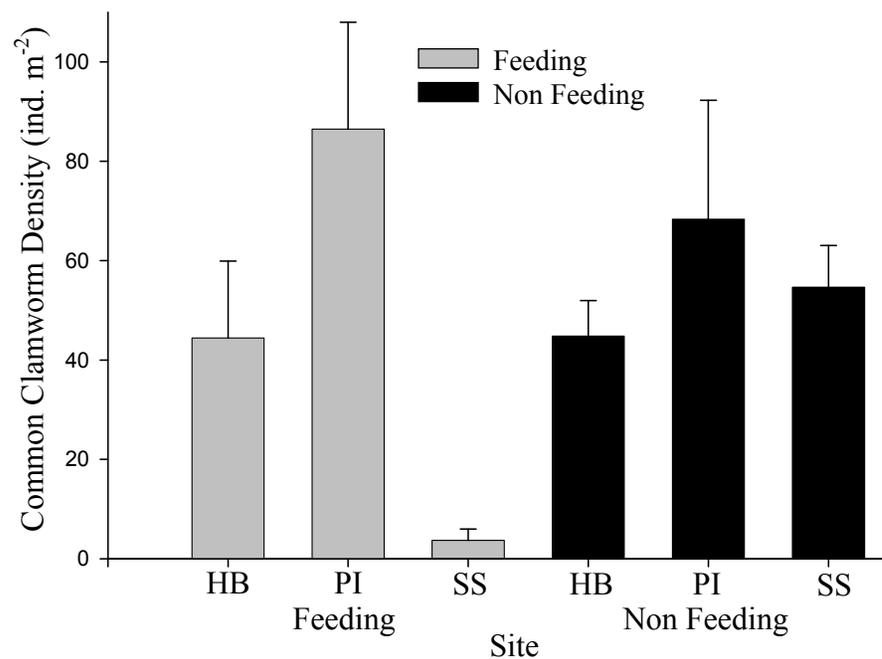


Figure 34. Location of common clamworm (*Neanthes succinea*) densities at surf scoter Feeding and Non Feeding sites. HB = Herring Bay, PI = Poplar Island, and SS = Shady Side.

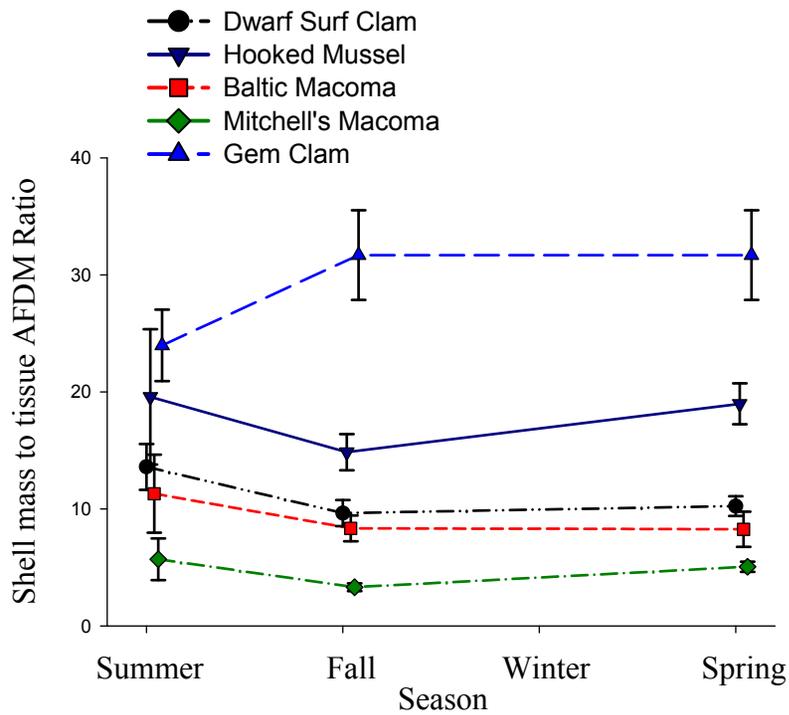


Figure 35. Seasonal ratio of shell mass (ash + ash-free) to tissue ash-free dry mass for five species of bivalves from the mesohaline portion of the mainstem Chesapeake Bay.

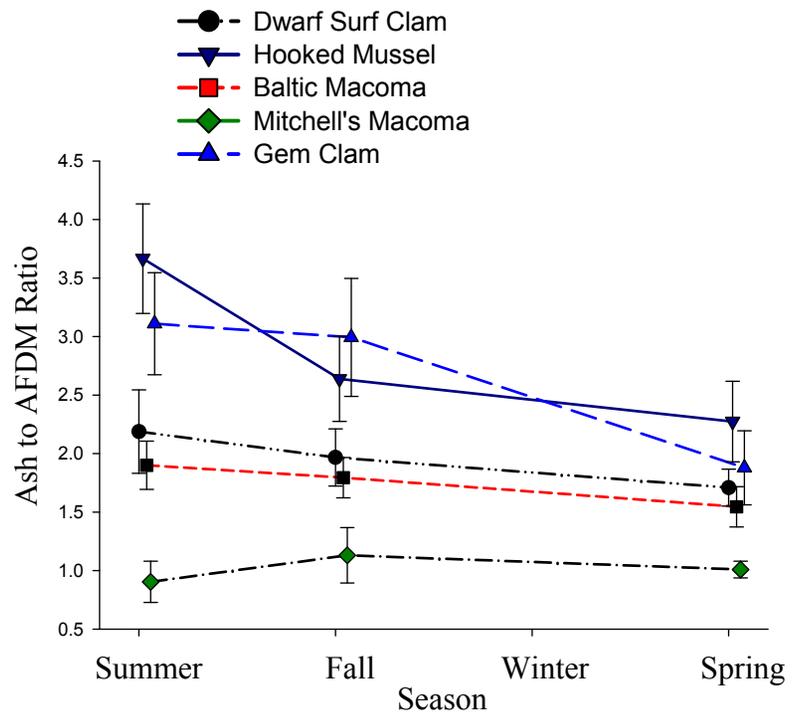


Figure 36. Seasonal ratio of total ash mass to ash-free dry mass for five species of bivalves from the mesohaline portion of the mainstem Chesapeake Bay.

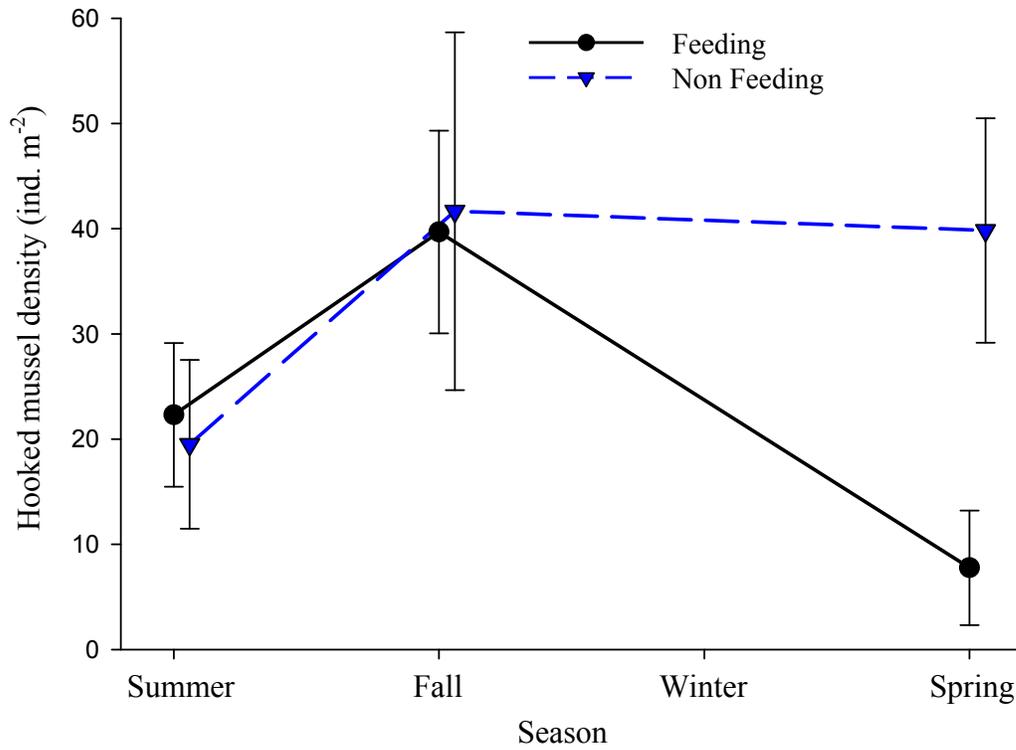


Figure 37. Habitat stratified, hard bottom only, seasonal densities (ind. m⁻² ± SE) of hooked mussel (*Ischadium recurvum*) at surf scoter Feeding and Non Feeding sites.

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