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Food Supplementation Increases Reproductive Performance of Ospreys in the Lower Chesapeake Bay

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Honolulu, Hawaii

Fisheries Biology, Cal Poly Humboldt, 2020

A Thesis presented to the Graduate Faculty of The College of William & Mary in Candidacy for the Degree of
Master of Science

Biology Program

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APPROVAL PAGE

This Thesis is submitted in partial fulfillment of
the requirements for the degree of

Master of Science

Michael Academia

Approved by the Committee August 2022

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COMPLIANCE PAGE

Research approved by

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ABSTRACT

The Atlantic States Marine Fisheries Commission (ASMFC), the governing body responsible for managing fisheries on the U.S. East Coast, formally adopted the use of Ecological Reference Points (ERPs) for Atlantic menhaden, *Brevoortia tyrannus*. Scientists and stakeholders have long recognized the importance of menhaden and predators such as ospreys, *Pandion haliaetus*, that support the valuable ecotourism industry and hold cultural significance. Landings in the reduction fishery are at their lowest levels and menhaden is facing potential local depletion. Mobjack Bay, located within the lower Chesapeake Bay, has been a focus of osprey research since 1970 and represents a barometer for the relationship between osprey breeding performance and menhaden availability. Since local levels of menhaden abundance were not available, we conducted a supplemental feeding experiment on osprey pairs during the 2021 breeding season. Our main objective was to determine if the delivery rate of menhaden had an influence on nest success and productivity. Nest success (χ^2 = 5.5, df = 1, P = 0.02) and productivity (β = 0.88, SE = 0.45, pseudo r^2 = 0.14, CI = 0.049, 1.825, P = 0.048) were significantly higher within the treatment group. The added average biomass/d/nest (β = 0.03, SE = 0.01, pseudo r^2= 0.60, CI = 0.01, 0.05, P = 0.02) and energy content/d/nest (β = 0.02, SE = 0.005, pseudo r^2 = 0.64, CI = 0.006, 0.03, P = 0.02) had an influence on pairs reaching maintenance reproductive rates (1.15 young/pair). Reproductive rates within the control group were low and unsustainable suggesting that current menhaden availability is too low to support a demographically stable osprey population.

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This master thesis is dedicated to my family and friends who have helped me countless times along the way…

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INTRODUCTION

According to the Magnusson-Stevens Fishery Management and Conservation Act, overfishing should not occur in the United States of America and thus, federal policy firmly reinforces the implementation of Ecosystem-Based Fisheries Management (EBFM) which is an approach that considers trophic interactions and aims to promote the health and resilience of the ecosystem (McLeod and Leslie 2009, Link 2010, NMFS 2016). Apex predators are essential indicators within this management approach and may provide more sensitive measures of changing fish populations because of their dietary dependencies (Furness 1982, Diamond and Devlin 2003). Monitoring fish-eating bird populations may be both more cost effective and better suited to the problem of understanding fish populations within an ecosystem (Cairns 1987). Bird metrics may play an increasing role in the assessment of prey availability, especially in areas where conventional fisheries data are insufficient (Cairns 1987). Bird populations may serve as an early warning system for changes in fish populations that have ecosystem implications (Kabuta and Laane 2003, Cury et al. 2005).

The Atlantic States Marine Fisheries Commission (ASMFC), the governing body responsible for managing fisheries on the U.S. East Coast, formally adopted the use of Ecological Reference Points (ERPs) for Atlantic menhaden, *Brevoortia tyrannus*. Historical estimates of menhaden were limited and the harvest effects did not produce sufficient information on important predator species. Therefore, the ASMFC developed an interest in establishing ERPs to set quotas and evaluate menhaden’s status and role as a forage species (Drew et al. 2021). Scientists and stakeholders have long recognized the importance of predators, such as bald eagles, *Haliaeetus* *leucocephalus*, ospreys, *Pandion haliaetus*, bottlenose dolphins, *Tursiops truncates*, and humpback whales, *Megaptera novaeanglia*, that support the valuable ecotourism industry and hold cultural significance (Butler et al. 2010, Glass and Watts 2009, Gannon and Waples 2004, Smith et al. 2015, Drew et al. 2021).

Atlantic menhaden are a schooling fish that can be found along nearshore coasts along the Atlantic Ocean from Nova Scotia, CAN, to Florida, USA and go through large age- and size-dependent seasonal migrations (Dryfoos et al. 1973, Nicholson 1978, Liljestrand et al. 2019). As indeterminate spawners, adults are capable of spawning multiple times in a season and inhabit estuarine and coastal areas such as Chesapeake Bay (Ahrenholz 1991, Southeast Data Assessment and Review [SEDAR] 2020). As juveniles, they spend their first spring and summer in estuaries and by late fall, they join with other subadults and adults and migrate to nearshore coastal waters (Anstead et al. 2021, Southeast Data Assessment and Review [SEDAR] 2020).

Menhaden support the largest fishery in the U.S. East Coast by volume and is used for bait and reduced to fish oil and meal which are used for animal feed, fertilizer, and human health supplements (Anstead et al. 2021). The reduction fishery began in the mid-1800s with the use of purse seine gear and peaked in 1956 with over 20 menhaden reduction factories along the Atlantic Coast (Southeast Data Assessment and Review [SEDAR] 2020). Later in the 1960s, factories north of Chesapeake Bay shut down due to the scarcity of fish. Currently, landings in the reduction fishery are at their lowest levels (Southeast Data Assessment and Review [SEDAR] 2020) and at Chesapeake Bay, populations of menhaden are facing potential local depletion. ASMFC defined localized depletion in Chesapeake Bay “as a reduction in menhaden population density below the level of abundance that is sufficient to maintain its basic ecological, economic, and social/cultural functions” (Annis et al. 2009). Due to present-day fishing pressure, menhaden populations within Chesapeake Bay are not being sustained as an adequate forage base. Localized depletion has not been officially defined or evaluated by managers because estimates of the standing stock within Chesapeake Bay have been unavailable and thresholds for exploitation cannot be resolved.

Known as the fish hawk, the osprey was selected as an appropriate non-finfish ERP to evaluate localized depletion of menhaden and food limitation within Chesapeake Bay. The ERP Work Group emphasized the research need for diet data collection and demographic responses of non-finfish predators (Atlantic States Marine Fisheries Commission [ASMFC] 2017). According to Buccheister et al. (2017), the nearshore piscivorous birds such as ospreys are sensitive to the overfishing of menhaden. Ecologically, ospreys are generalized specialists (Beirregaard et al. 2014). Specialized in that they are obligate piscivores and generalized in that they predate upon many species of fish. Ospreys surface plunge at a maximum depth of one meter and are more susceptible to a decrease in fish density than other birds such as pursuit divers that search for prey while swimming on the water surface and dive to deeper depths (Ashmole 1971, Cramp and Simmons 1979). Piscivory and plunge diving influences an ecological indicator’s response to fish supply perturbations (Einoder 2009). Reduced prey availability and fluctuations in environmental conditions are more evident in the foraging behavior and breeding success of a specialist (Furness et al 1984, Montevecchi 1993). Moreover, shallow divers and surface feeders are more vulnerable, are considered more sensitive indicators than pursuit divers, and show greater variation in breeding performance (Monoghan et al. 1992, Montevecchi, 1993, Scott et al. 2006). As one of the more recognized raptors, ospreys have been used as an ecotoxicological sentinel species of environmental health due to their reproductive responses to natural and anthropogenic pressures and life history traits (Johnson et al. 2008, Grove et al. 2009, Henny et al. 2008). Ospreys exhibit strong nest fidelity and their reproductive status is observable by ground, boat, or aerial surveys which makes them a valuable and efficient sentinel of the ecosystem (Ogden et al. 2014) and an exemplary ERP for menhaden.

The Chesapeake Bay supports one of the largest osprey breeding populations in the world (Henny 1983, Watts and Paxton 2007). As with many similar populations, ospreys in the Chesapeake Bay experienced dramatic declines in the post-World War II era due to reproductive suppression (Truitt 1969, Wiemeyer 1971, Kennedy 1971, Reese 1977) induced by environmental contaminants (Via 1975, Wiemeyer et al. 1975). The population sustained a low point by 1973 when Henny et al. (1974) estimated its size to be 1,450 breeding pairs. From 1973 to 1995, the population more than doubled in size to nearly 3,500 pairs (Watts et al. 2004) and is now believed to be between 8,000-10,000 pairs. However, the population has experienced spatial variation in recovery (Watts et al. 2004, Watts and Paxton 2007). For example, average doubling time for the population on low-salinity, upper reaches of tributaries, was less than four years while doubling time on higher-salinity reaches of the lower Chesapeake Bay exceeded 40 years (Watts et al. 2004). This variation reflects the extent of the earlier decline, immigration from other regions of the Chesapeake Bay, and the local demography of pairs that may have been influenced by prey availability. In this study, our aim was to evaluate the reproductive performance of a local population of ospreys within the lower Chesapeake Bay, Virginia.

Mobjack Bay has been a focus of osprey research since 1970 and represents a barometer for the relationship between osprey breeding performance and menhaden availability (Glass 2008). During the mid-1970s when menhaden abundance was high, there was little evidence of food limitation reflected in osprey reproductive performance and brood sizes (Stinson 1976). By the mid-1980s, several signs of food limitation were present within the population including an increase in foraging effort or hunting time by adult males, lower provisioning rates, sibling aggression, and subsequent brood reduction (McLean 1986). By the early 2000s, the proportion of menhaden in the diet had dropped by 40%, brood reduction wasn't unusual, and reproductive rates had dropped to precarious levels (Glass 2008).

Our primary objective was to test if reproductive success for ospreys in Mobjack Bay was limited by fish availability. We conducted a supplemental feeding experiment for osprey pairs nesting in Mobjack Bay during the 2021 breeding season. A clear barrier in resolving the relationship between osprey productivity and menhaden consumption is the lack of menhaden abundance data that can be scaled down to the local level. If such data were available, we could monitor osprey foraging, provisioning, and productivity, and assess the functional response to available menhaden. Since such data are not available, a food manipulative experiment in the wild was performed (Piatt et al. 2007). Our secondary objective was to determine prey composition and the dietary importance of menhaden.

METHODS

Study Species – Ospreys are large, long-winged raptors with a nearly global distribution that feed exclusively on fish (Prevost 1983, Clark & Wheeler 1987, Dunne et al. 1988). Most osprey populations across North America are migratory, spend the winter months in Central or South America and begin breeding at the age of three (Henny & Wight 1969) Age-at-first-reproduction in Chesapeake Bay ospreys was recorded from 4 years (Kinkead 1985) to 5.7 years (Poole 1989, Poole et al. 2002). As the population reaches carrying capacity, age-at-first-reproduction increases (Spitzer 1980, Poole 1989). Poole (1989) estimated that pairs within the Chesapeake Bay must produce 1.15 young per year in order to offset adult mortality. On average, if the population consistently meets or exceeds this rate (demographic source) then the population would be expected to be stable to increasing (Pulliam 1988). If the reproductive rate consistently falls below this threshold (demographic sink) the population would be expected to decline in the absence of compensatory immigration.

Food Addition Experiment – We established treatment (fish addition) and control (no fish addition) nests to assess the effect of increased provisioning on demography. We added 472 g ± 7.9 (SE) of menhaden every 3.5d ± 0.2 to treatment nests from the time of hatching to six weeks of age. We delivered menhaden to nests using a telescopic pole with a mounted delivery device. We sourced fresh or frozen menhaden from a local fishing supply company and the fish were counted, weighed, coded, and separated into packages for easy deployment. We selected study nests based on accessibility and randomly assigned accessible nests to treatments. We conducted an initial survey (late March to mid-April) of the study area for osprey nests (N = 114) and recorded location (latitude, longitude), accessibility by boat, nesting stage, nest substrate, height over water, and water depth. We screened nests for initial inclusion in the study based on accessibility, height over water (to allow for ready access to the nest) and water depth (to allow for boat access and maneuverability). We only included nests within the study that survived to hatching stage. We monitored all nests included within the initial draw until clutches hatched. Nests that hatched eggs were randomly assigned to two treatment groups (Fig. 1) including a control group (N = 15) and a food addition group (N = 16).

Demography – We monitored nests twice per week from clutch completion to fledging to quantify demographic parameters including clutch size, brood size, and the number of young fledged. From observations, we determined brood reduction (number of young lost between hatching and fledging). We noted the age that nestlings died and the stage when nests failed. We consider a nest to be successful if the pair produced at least one young to fledging age. We consider productivity to be the number of young that reached fledging age (7 wks) per active nest (Steenhof and Newton 2007). We used a telescopic mirror pole to facilitate the examination of nest contents for nests that were >2 m above the water line.

Provisioning – We used trail cams (Browning Strike Force HD Pro X - BTC-5HDPX) to quantify nest provisioning rates including the average number of fish (n/day), biomass (g/day) and energy (kcal/day) for a subsample of treatment (N = 7) and control (N = 4) nests. We deployed cameras on nest structures that would accommodate them. We fastened trail cams to 1.91 cm (3/4 inch) diameter conduit and mounted conduit to the nesting structure such that cameras were positioned approximately 1 m above the nest.

Cameras were programmed to record an image every 5 min during daylight hours (05:00 to 22:00). We extracted images from the photo set that depicted fish delivered to nests and identified all fish to the lowest taxonomic level possible. Most fish were identified to the species level but others could only be identified to the genus or family level. We estimated fish length from photos within an image processing program, ImageJ with Java (https://imagej.nih.gov/ij/index.html) and compared to known lengths from reference structures (Poole et al. 2002) including adult bill (male =32.5, female = 34.6 mm) and talon (male = 28.9, female = 30.0 mm). We estimated the biomass (g) of each fish using species-specific length-mass equations from published literature and FishBase (https://fishbase.in/, Appendix 1). We converted biomass to energy (kcal) using published species-specific energy density values (Appendix 2). For species that could not be identified to species, we used length-mass equations and energy density from a representative species of the taxonomic group. We consider the provisioning of control nests to include fish provided by adults and for treatment nests to include fish provided by adults and menhaden that we added to nests.

Statistical Analysis – Data were not independent, not normally distributed, and non-homogenous therefore, we used appropriate tests. We investigated the influence of treatment (control vs food addition) on demographic parameters including nest success, clutch size, the number of young hatched, brood reduction, and productivity. We constructed a two-by-two contingency table and used Pearson's Chi-squared analysis to compare the relationship between treatment type and nest success. We used Generalized Linear Models (GLMs) to determine if there were the average differences in clutch size, the number of young hatched, brood reduction, and productivity between the treatment types. For provisioning (fish/d, biomass/d, energy content/d), we analyzed data from trail cameras to evaluate the relationship between provisioning and demographic parameters. It is important to note that our models were based on totals and/or average provisioning rates including naturally provisioned and supplemental fish.

We used Generalized Linear Mixed Models (GLMMs) with a negative binomial distribution and log link, nest and treatment type as the random effects, and food addition and total provisioning (natural and supplemented) as the fixed effects. For the influence of provisioning on demographics, we used GLMs with a negative binomial distribution and log link and compared the effects of the mean fish/d, biomass/d, and energy content/d (natural and supplemented) on productivity (both treatment groups combined, N = 11). We calculated the supplemented average biomass/d/nest and energy content/d/nest threshold needed for the production of 1.15 fledglings per nest-season (estimated break-even rate). All analyses were performed in RStudio 4.02 and we used the MASS and glmmTMB packages for model development and validated by the DHARMa package for residual diagnostics on hierarchical regression models (Venables and Ripley 2002, Brooks et al. 2017, Hartig 2021, R Core Team 2020).

RESULTS

Food Addition and Demography - For the food addition group, 13 of the 16 nests (81%) succeeded with an average productivity rate of 1.13 $\pm$ 0.18 (SE) young/active nest. The three nests that failed in this group failed on average during the first 1.38 $\pm$ 0.5 wks or when young were 10 d old. For the control group, five of the 15 nests (33%) succeeded with an average productivity rate of 0.47 young/active nest. The ten nests that failed in this group failed on average during the first 2.2 $\pm$ 0.5 wks. The age at failure (d) between the food addition and control groups was not statistically significantly different (β = -0.47, SE = 0.41, P = 0.25). The age at failure for the control group ranged from 3 - 42 d with the highest mortality experienced during the first 15.5 d $\pm$ 3.4 of the nestling period. Nest success and productivity were significantly different between the control and food addition groups (Table 1 & Fig. 2). Clutch size, the number of young hatched, and brood reduction were not significantly different between the control and food addition groups (Table 2).

Provisioning and Productivity – Food supplementation had a significant influence on the number of fish and amount of energy available to osprey broods (Table 4). A total of 241 Atlantic menhaden was supplemented to the food addition group and contributed 32,384 g that represented an estimated 61,206 kcal. This increased the average total prey biomass and energy content within the food addition group to 226.5 g/d/nest and 396.2 kcal/d/nest. The average biomass that was delivered to the control group was 166.8 g/d/nest and the average energy content was 242.2 kcal/d/nest (Table 3). For the control group, adult osprey delivered an average of 1.2 fish/d/nest compared to 1.1 fish/d/nest for the supplemented group.

Food supplementation had a significant influence on the likelihood that pairs reached the threshold reproductive rate of 1.15 young/nest (Fig. 3). The estimated average fish biomass and energetic content needed for a pair to produce the threshold reproductive rate was 202.7 g/d and 338.6 kcal/d respectively. Within the study area, pairs required supplementation of 63.4 g/d of menhaden or 121 kcal/d in order to reach the productivity threshold.

Diet composition included a diverse list of fish species (Table 3). A total of 600 fish were documented as prey by ospreys in which 81% of taxa were identified to 21 species or to at least family. Only five species including Atlantic menhaden *(39%),* Atlantic herring *(Clupea harengus) (10.3%),* Atlantic croaker *(Micropogonias undulatus) (5.8%),* gizzard shad (*Dorosoma cepedianum*) (5.7%), and spot *(Leiostomus xanthurus) (5%)* accounted for 65.5% of the fish delivered.

DISCUSSION

Supplementation of osprey nests with menhaden had a significant influence on the ability of nesting pairs to reach reproductive rates required for population maintenance. Pairs that did not receive supplementation had reproductive rates (0.47 young/nest) that were less than half of threshold levels. Within Mobjack Bay have, productivity rates shifted from reproductive surplus to reproductive deficit since the 1980s. For example, populations at various locations along the main stem of Chesapeake Bay were considered strongholds (McLean 1986, Byrd 1988). During 1983 and 1984, the average reproductive rate was 1.39 young/pair (Byrd 1987). By 1988 and 1990, average productivity had dropped to 0.91 young/pair (Byrd 1988 and 1990) and by 2005 and 2006 productivity had dropped further to 0.75 young/pair (Glass 2008). If fishing pressure on menhaden within Chesapeake Bay persists, osprey productivity rates could decline precipitously, threaten population stability, and eventually lead to widespread population collapse. Menhaden populations should be maintained at levels that will sustain a stable osprey population in which they are able to produce 1.15 young/active nest to offset mortality.

Our research suggests that food addition significantly influenced osprey provisioning rates and these rates impacted reproductive performance. Specifically, daily average biomass and energy content of the prey composition significantly influenced productivity. Lind (1976) used a model developed by Wiens and Innis (1974) and calculated that each adult osprey required 286 kcal/d and each nestling at 11-16 d old needed at least 113 – 170 kcal/d. Based on calculations in which fish with an energy content of 1 kcal/g, a nest with two young plus the female would require 794 g of fish/d in order to successfully fledge and a nest with three young would require 1048 g of fish/d (Winberg 1960). Along the U.S. Eastern Coast, Poole (1982) determined that male ospreys delivered 816 – 1426 g/d to nests that had young and nests that produced three – four young. In our study, menhaden consisted of 39% of the total diet composition and these fish have a high energy content of 1.89 kcal/g (June and Nicholson 1964). Based on the calculations of Winberg (1960), if a nest fledged two young that was supplied with 39% or 309.7 g/d or 585.3 kcal/d of menhaden, the estimated additional biomass and energy content required would be 648.2 g/d or 1,225.1 kcal/d. Similarly if a nest fledged three young and was supplied with 39% or 408.7 g/d or 772.4 kcal/d of menhaden, the estimated additional biomass and energy content required would be 855.5 g/d or 1,616.9 kcal/d. For the nests in our study, the added average biomass and energetic threshold needed for a nest to reach the reproductive break-even point are 63.4 g/d and 121 kcal/d which would be a total average of 208.1 g/d and 347.6 kcal/d (Fig. 3).

When we directly compared the provisioning rates in this study to historical studies in Mobjack Bay and the higher salinity areas of Chesapeake Bay, declines in daily fish deliveries were made evident. In 1975 and 1985, the fish delivery rate was 0.53 fish/hr/nest and 0.35 fish/hr/nest (McLean and Byrd 1991). In 2006 and 2007, ospreys in the higher salinity areas delivered an average of 0.26 fish/h/nest (Glass 2008). Our study revealed that in 2021, the fish delivery rate diminished to a mean of 0.11 fish/hr/nest. The average daily biomass delivered per nest fell from 237.1g and 172.3g in 1975 and 2007 to 144.7g in 2021 (Table 3, McLean and Byrd 1991, Glass 2008).

Brood reduction has been an effective parameter linking reproductive performance to food limitation in osprey (Glass 2008). In a 5-yr study, Reese (1977) determined nestling loss rates in the upper Chesapeake Bay ranged from 8-23%. Nestling mortality rates were 47% and 78% for the supplementation and control groups respectively in this study. Poole (1984) conducted a 4-yr study in New England and determined that 75% of nestling mortality was caused by starvation. Glass and Watts (2009) determined that brood reduction was highly significant between nests in the lower estuarine sites compared to the higher estuarine sites and these data suggested that ospreys in the higher salinity areas were experiencing more food limitation than the lower salinity areas. Brood reduction has generally been linked with the lack of food availability in other study areas (Poole 1982, Jamieson et al. 1983, Eriksson 1986, Hagan 1986, Forbes 1991, Glass and Watts 2009). Although brood reduction was higher in the control group, differences were not found to be significant in our study. This discrepancy could have been attributed to treatment effects in which the timing and intensity of the protocol was not strong enough to detect a significant signal. Perhaps if we supplemented more fish in greater frequency, we would have observed significant differences in the average brood reduction between the experimental groups.

The most compelling explanation for lower provisioning and productivity rates is localized depletion of the primary prey base. Atlantic menhaden has a higher lipid content compared to other species with a nearly a 2:1 energy content/biomass ratio (June and Nicholson 1964). Ospreys depend on menhaden and their reproductive performance is inextricably linked to the availability and abundance of this fish. In fact, previous studies have substantiated that menhaden are a vital prey item for ospreys during the breeding season particularly in the mid-Atlantic and northeastern United States (Spitzer and Poole 1980, Poole 1989, McLean and Byrd 1991, Steidl et al. 1991, Glass and Watts 2009). In 1985, this fish species consisted of 75% of the prey composition of ospreys in the lower Chesapeake Bay (McLean and Byrd 1991). Then in 2006 and 2007, menhaden declined to 32% of the prey composition (Glass 2008). In our study menhaden comprised of 39% of the total prey composition (Table 1). Assuming that the prey composition of ospreys reflects prey availability on a local level (Greene et al. 1983, Edwards 1988, Glass 2008), the current percentage of menhaden could indicate that this species has diminished in availability compared to the later portion of the 20th century.

Potential localized depletion of menhaden populations is one of the major sources of concern and conflict within Chesapeake Bay. According to the ASMFC, the coastwide stock assessment has determined that menhaden is not overfished and that no overfishing is occurring (Southeast Data Assessment and Review [SEDAR] 2020). However, a coastwide assessment does not capture spatial variation in menhaden availability or locations with persistent depletion. Seine surveys of juvenile menhaden in Maryland and Virginia indicate that low levels of abundance and recruitment have been happening since the early 1990’s and 2000’s (Atlantic States Marine Fisheries Commission [ASMFC] 2004, Southeast Data Assessment and Review [SEDAR] 2020). Our data suggests that the reliable metric that links population decline and food limitation is the osprey productivity rate. During the population decline in northern Florida, Bowman et al. (1989) determined that the productivity rate was 0.56 young/nest and this was due to insufficient food availability. When the Florida Bay population was healthy and food was abundant (Henny and Ogden 1970), the productivity rate was 1.22 young/nest which is similar to the rate acquired by the food addition group of our study at 1.13 young/nest.

EBFM evolves when ERPs are consistently monitored (Pikitch et. al. 2004). According to Amendment 3 of the Interstate Fishery Management Plan (FMP) for Atlantic menhaden (Southeast Data Assessment and Review [SEDAR] 2020, Anstead et al. 2021), ERPs are described as “a method to assess the status of menhaden not only with regard to the sustainability of human harvest, but also with the regard to their interaction with predators and the status of other prey species.” The ERP working group is tasked with developing ERPs that are menhaden-specific that can account for the abundance of menhaden and their species role as a forage fish (Amendment 3 to the FMP, Anstead et al. 2021). Ospreys are a non-finfish predator and can serve this role which can allow management to practice informed decisions to develop harvest targets, assess menhaden’s role as prey for upper trophic levels, and advance an ecosystem approach to fisheries management (EAFM) which considers multiple components of the ecosystem than just the target species (Patrick and Link 2015). The menhaden population within Mobjack Bay is not currently adequate to sustain the osprey breeding population.

Table 1. Two-way contingency table used for the Pearson's Chi-squared analysis that summarizes the relationship between treatment types and nest success during the 2021 osprey breeding season in the lower Chesapeake Bay, VA, USA ($\chi^2 = 5.5$, df = 1, P = 0.02).

	Nest Success (Nests)		
Treatment	Successful	Failed	Total
Fish Addition	13	3	16
Control	5	10	15
Total	18	13	31

Table 2. Results for GLMs used to compare demographic parameters between treatment types during the 2021 osprey breeding season in the lower Chesapeake Bay, VA, USA.

DEMOGRAPHIC PARAMETERS	**β**	**SE**	**PSEUDO r^2**	**CI**	***P***
CLUTCH SIZE	0.07	0.21	0.75	-0.34, 0.48	0.75
No. of YOUNG HATCHED	0.12	0.24	0.04	-0.33, 0.62	0.57
BROOD REDUCTION	0.20	0.31	0.02	-0.81, 0.40	0.50

Table 3. Prey composition and provisioning rates of nests under trail camera surveillance (N = 11) during the 2021 osprey breeding season in the lower Chesapeake Bay, VA, USA.

	Control				Treatment								Total					
Species	No.	% Total	Biomass (g)	Kcal	No.	Added	% Total	% Added	Biomass (g)	Added	Kcal	Added	No.	Added	% Total	% Added	Biomass (g)	Kcal
largemouth bass (*Micropterus salmoides*)	0	0	0	0	1		0.3		249.4		294.2		1		0.2		249.4	294.2
Atlantic croaker (*Micropogonias undulatus*)	17	7.3	740.6	740.6	18		4.9		562.1		562.1		35		5.8		1302.8	1302.8
Atlantic cutlassfish (*Trichiurus lepturus*)	1	0.4	97.2	97.2	1		0.3		2287.3		2287.3		2		0.3		2384.6	2384.6
Atlantic herring (*Clupea harengus*)	13	5.6	365.6	365.6	49		13.3		1574.6		1574.6		62		10.3		1940.2	1940.2
Atlantic menhaden (Brevoortia tyrannus)	67	28.9	12082.2	22830.0	165	241	44.8	40.0	28986.6	32384.5	54772.2	61206.7	232	241	38.7	29.0	41068.7	77602.2
Atlantic needlefish (*Strongylura marina*)	1	0.4	301.3	301.3	0		0		0		0		1		0.2		301.3	301.3
black drum (*Pogonias cromis*)	2	0.9	89.7	89.7	0		0		0		0		2		0.3		89.7	89.7
bluefish (*Pomatomus saltatrix*)	2	0.9	806.3	806.3	1		0.3		550.2		550.2		3		0.5		1356.5	1356.5
catfish (*Ictaluridae*)	0	0	0	0	1		0.3		206.6		212.8		1		0.2		206.6	212.8
gizzard shad (*Dorosoma cepedianum*)	13	5.6	4331.3	8662.5	21		5.7		3603.4		7206.8		34		5.7		7934.7	15869.3
oyster toadfish (*Opsanus tau*)	0	0	0	0	1		0.3		75.0		75.0		1		0.2		75.0	75.0
red drum (*Scianeops ocellatus*)	1	0.4	279.9	279.9	1		0.3		444.8		444.8		2		0.3		724.7	724.7
sculpin (*Cottoidea*)	0	0	0	0	2		0.5		116.6		116.6		2		0.3		116.6	116.6
spadefish (*Chaetodipterus faber*)	5	2.2	387.8	387.8	1		0.3		257.0		257.0		6		1.0		644.8	644.8
Spanish mackerel (*Scomberomorus maculatus*)	8	3.4	3004.1	3004.1	1		0.3		6.6		6.6		9		1.5		3010.7	3010.7
spot (*Leiostomus xanthurus*)	17	7.3	966.6	966.6	13		3.5		566.7		566.7		30		5.0		1533.4	1533.4
spotted seatrout (*Cynoscion nebulosus*)	12	5.2	3973.1	3933.4	12		3.3		2819.5		2791.3		24		4.0		6792.6	6724.7
striped bass (*Morone saxatilis*)	7	3.0	2632.6	2422.0	9		2.4		614.0		564.8		16		2.7		3246.5	2986.8
summer flounder (*Paralichthys dentatus*)	9	3.9	528.0	443.5	6		1.6		396.4		333.0		15		2.5		924.4	776.5
unknown	55	23.7	1761.8	1761.8	61		16.6		1572.2		1572.2		116		19.3		3334.0	3334.0
weakfish (*Cynoscion regalis*)	1	0.4	131.7	130.4	0		0.0		0		0		1		0.2		131.7	130.4
white perch (*Morone americana*)	1	0.4	210.9	248.9	4		1.1		414.8		489.4		5		0.8		625.7	738.3
Total	232		32690.7	47471.58	368				45303.7		74677.7		600				77994.4	122149.3
Total w/ Added					**609**				**77688.2**		**135884.4**		**841**				**110378.9**	**183356.0**
Average p Day	4.7		667.16	968.81	7.5				924.6		1524.0		12.2				1591.8	2492.8
Average p Day w/ Added					**12.4**				**1585.5**		**2773.2**		**17.2**				**2252.6**	**3742.0**
Average p Day p Nest	1.2		166.8	242.2	1.1				132.1		217.7		1.1				144.7	226.6
Average p Day p Nest w/ Added					**1.8**				**226.5**		**396.2**		**1.7**				**204.8**	**340.2**

Table 4. Results of GLMMs with treatment effects on provisioning rates per d of nests under trail camera surveillance (N = 11) during the 2021 osprey breeding season in the lower Chesapeake Bay, VA, USA.

TREATMENT EFFECTS	**β**	**SE**	**z VALUE**	**CI**	***P***
FISH (number of fish/d)	0.25	0.02	13.4	0.21, 0.29	< 0.001
BIOMASS (g of fish/d)	0.002	0.0004	4.65	0.001, 0.003	< 0.001
ENERGY CONTENT (kcal of fish/d)	0.001	0.0002	5.22	0.008,0.002	< 0.001

Figure 1. Map of the experimental area of Mobjack Bay on the lower eastern region of Chesapeake Bay, VA, USA. The locations of the control group (N = 15) and the food addition group (N = 16).

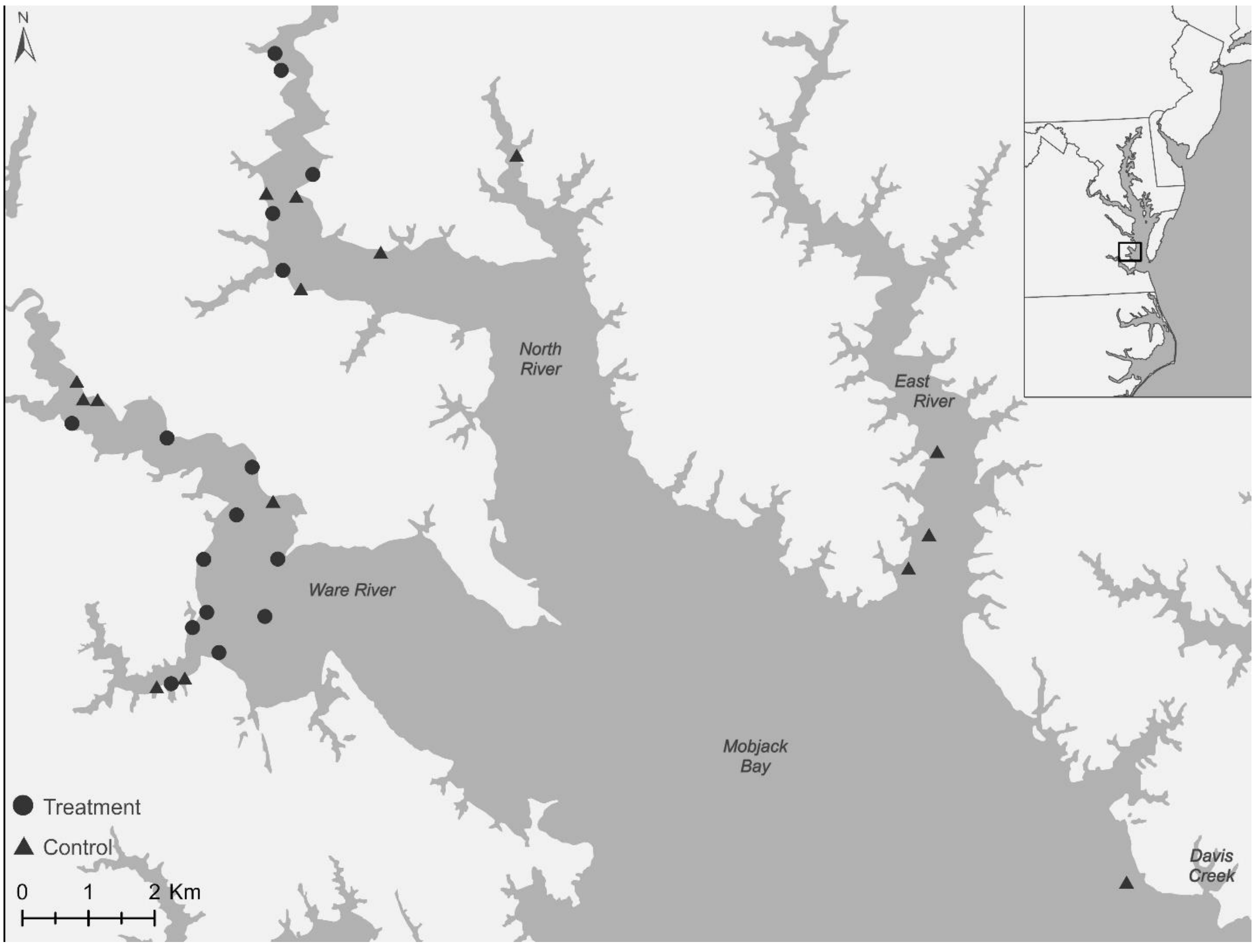


Figure 2. Productivity between the control group (N = 15) and the treatment group (N = 16) of ospreys during the 2021 breeding season in the lower Chesapeake Bay, VA, USA (β = 0.88, SE = 0.45, pseudo R^2 = 0.14, CI =0.049, 1.825, P= 0.048). Violin shapes represent the density of data distribution and the middle horizonal line of the box plots represent the median values.

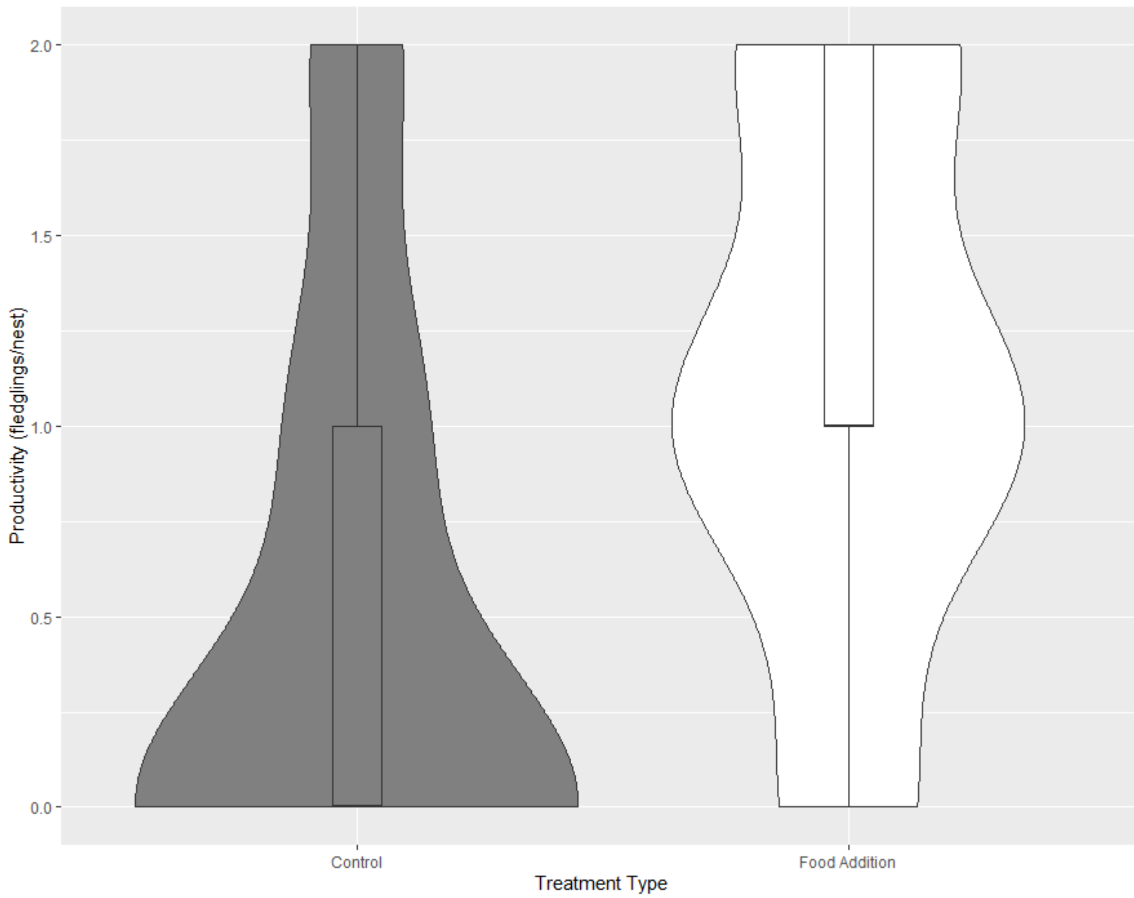
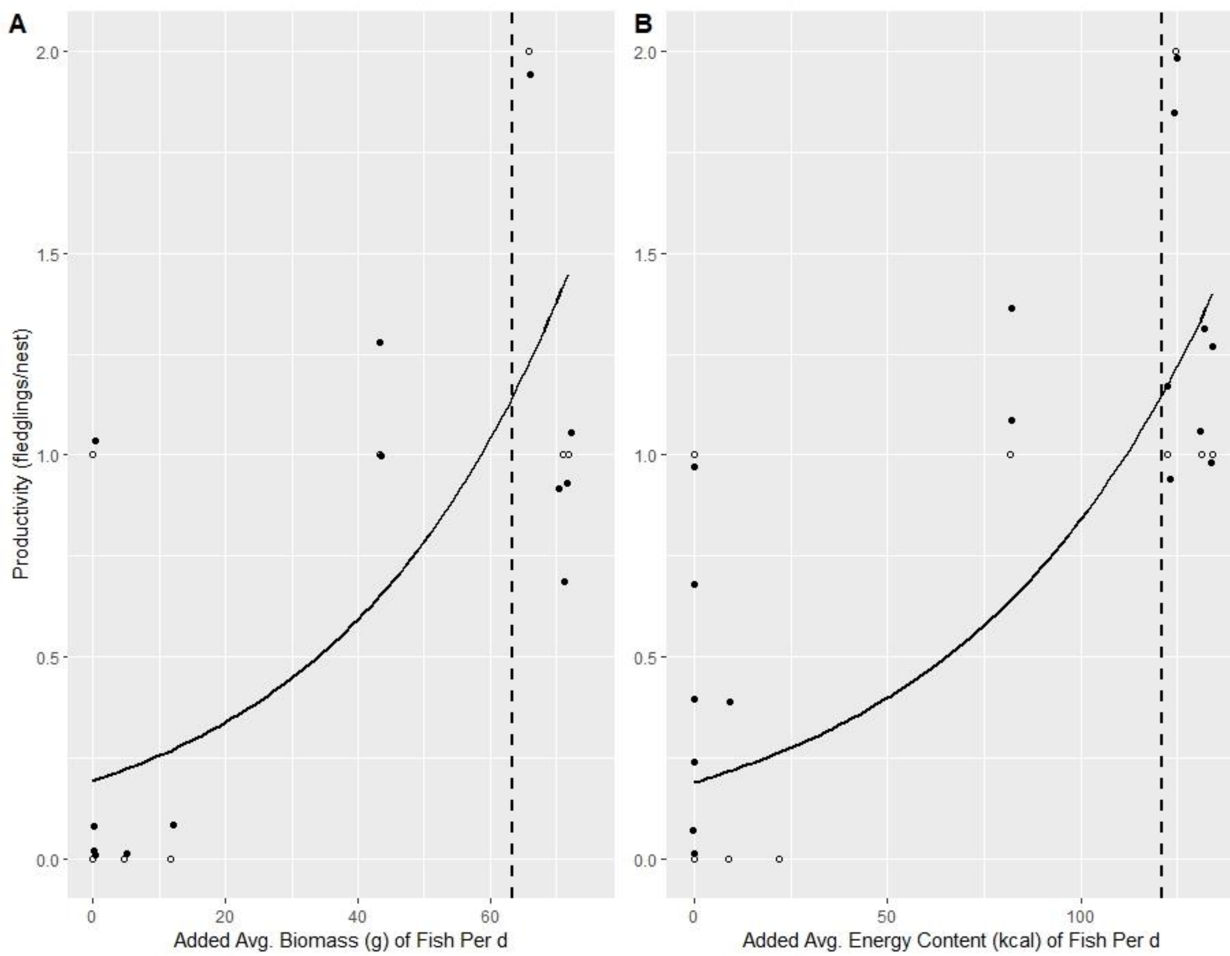


Figure 3. GLM's of the influence of the added (A) avg. biomass/d/nest (β = 0.03, SE = 0.01, Pseudo R^2 = 0.60, CI = 0.01, 0.05, P = 0.02) and (B) avg. energy content/d/nest (kcal) (β = 0.02, SE = 0.005, Pseudo R^2 = 0.64, CI = 0.006, 0.03, P = 0.02) for osprey pairs under trail camera surveillance after seven weeks post hatch of the first egg in 2021 breeding season in the lower Chesapeake Bay, VA, USA. Dotted lines indicate the supplemented average biomass (63.4 g) and energy content (121 kcal) thresholds needed per d for to produce 1.15 young per nest-season.



Appendix 1. Length-weight conversion equations for all taxa identified in the osprey diet during the 2021 breeding season in the lower Chesapeake Bay.

SPECIES	BIOMASS CONVERSION	REFERENCE
Atlantic croaker (*Micropogonias undulatus)*	$M = 0.0052*L^{3.148}$	Wilk et al. 1978
Atlantic cutlassfish (*Trichiurus lepturus)*	$M = 0.001*L^{3.40}$	https://fishbase.in/
Atlantic herring (*Clupea harengus)*	$M = 0.0075*L^{3.03}$	Hubold 1978
Atlantic menhaden (*Brevoortia tyrannus)*	$M = 0.0191*L^{3.00}$	June and Nicholson 1964
Atlantic needlefish (*Strongylura marina)*	$M = 0.011*L^{3.108}$	https://fishbase.in/
black drum (*Pogonias cromis)*	$M = 0.0114*L^{3.05}$	https://fishbase.in/
bluefish (*Pomatomus saltatrix)*	$M = 0.0281*L^{2.80}$	https://fishbase.in/
catfish (*Ictaluridae)*	$M = 0.0185*L^{3.00}$	Crawford 1993
gizzard shad (*Dorosoma cepedianum)*	$M = 0.0182*L^{2.89}$	Lagler and Van Meter 1951
largemouth bass (*Micropterus salmoides)*	$M = 0.0158*L^{2.96}$	Swingle 1965
oyster toadfish (*Opsanus tau)*	$M = 0.0068*L^{3.16}$	https://fishbase.in/
red drum (*Scianeops ocellatus)*	$M = 0.01001*L^{3.028}$	https://fishbase.in/
sculpin (*Cottoidea)*	used oyster toadfish	https://fishbase.in/
spadefish (*Chaetodipterus faber)*	$M = 0.0926*L^{2.684}$	https://fishbase.in/
spanish mackerel (*Scomberomorus maculatus)*	$M = 0.0884*L^{2.982}$	https://fishbase.in/
spot (*Leiostomus xanthurus)*	$M = 0.0092*L^{3.072}$	Dawson 1965
spotted seatrout (*Cynoscion nebulosus)*	$M = 0.0131*L^{3.000}$	Crawford 1993
striped bass (*Morone saxatilis)*	$M = 0.0061*L^{3.153}$	Mansueti 1961
summer flounder (*Paralichthys dentatus)*	$M = 0.0102*L^{2.994}$	Smith and Daiber 1977
unknown	used Atlantic herring	https://fishbase.in/
weakfish (*Cynoscion regalis)*	$M = 0.0088*L^{3.000}$	Crozier and Hecht 1913
white perch (*Morone americana)*	$M = 0.0125*L^{3.020}$	St. Pierre and Davis 1972

Appendix 2. Mass-energy conversion equations for all taxa identified in the osprey diet during the 2021 breeding season in the lower Chesapeake Bay.

SPECIES	ENERGY CONVERSION	REFERENCE
Atlantic croaker (*Micropogonias undulatus)*	E = 100*(M/100)	Frimodt 1995
Atlantic cutlassfish (*Trichiurus lepturus)*	used Atlantic herring	
Atlantic herring (*Clupea harengus)*	E = 190*(M/190)	Frimodt 1995
Atlantic menhaden (*Brevoortia tyrannus)*	E = 189*(M/100)	Frimodt 1995
Atlantic needlefish (*Strongylura marina)*	used Atlantic herring	
black drum (*Pogonias cromis)*	used Atlantic croaker	
bluefish (*Pomatomus saltatrix)*	used Atlantic herring	
catfish (*Ictaluridae)*	E = 103*(M/100)	Frimodt 1995
gizzard shad (*Dorosoma cepedianum)*	E = 200*(M/100)	Watt and Merrill 1975
largemouth bass (*Micropterus salmoides)*	used white perch	
oyster toadfish (*Opsanus tau)*	used summer flounder	
red drum (*Scianeops ocellatus)*	used Atlantic croaker	
sculpin (*Cottoidea)*	used summer flounder	
spadefish (*Chaetodipterus faber)*	used Atlantic croaker	
Spanish mackerel (*Scomberomorus maculatus)*	used Atlantic herring	
spot (*Leiostomus xanthurus)*	used Atlantic croaker	
spotted seatrout (*Cynoscion nebulosus)*	E = 99*(M/100)	Frimodt 1995
striped bass (*Morone saxatilis)*	E = 92*(M/100)	Frimodt 1995
summer flounder (*Paralichthys dentatus)*	E = 84*(M/100)	Frimodt 1995
unknown	used Atlantic herring	
weakfish (*Cynoscion regalis)*	E = 99*(M/100)	Frimodt 1995
white perch (*Morone americana)*	E = 118*(M/100)	Watt and Merrill 1975

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