

ABSTRACT

Title of Dissertation/Thesis: FISH ASSEMBLAGE STRUCTURE IN
MARYLAND'S COASTAL LAGOON
COMPLEX

Robert F. Murphy, Master of Science, 2005

Dissertation/Thesis Directed By: Professor David H. Secor, Marine, Estuarine, and
Environmental Science Program

Fish assemblage structure of Maryland's coastal lagoon complex was analyzed for spatial and seasonal patterns for the period 1991-2000. Non-metric multidimensional scaling ordinated sites from Maryland's state trawl and seine surveys into discrete groups associated with each embayment. Dominant species from both surveys included *Callinectes sapidus*, *Anchoa mitchilli*, *Leiostomous xanthurus*, *Bairdiella chrysoura*, and *Brevoortia tyrannus*. One hundred two species were identified in the surveys, with total species richness highest in Chincoteague and lowest in Assawoman and Sinepuxent. The seine survey had the lowest catch-per-site in Chincoteague Bay while the other three embayments were of similar magnitude. The trawl survey had the lowest catch-per-site in Sinepuxent Bay (3,079) and the highest in Assawoman Bay (27,146). There was clear seasonality in assemblage structure with peak abundance and diversity in the summer compared to other seasons. Factors influencing the structure of Maryland's coastal lagoon complex included proximity to oceanic exchange and trophic status.

FISH ASSEMBLAGE STRUCTURE IN MARYLAND'S COASTAL
LAGOON COMPLEX

By

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INTRODUCTION

Coastal lagoons are often highly productive systems that function as nursery areas and feeding grounds for coastal fishes and thereby potentially support important commercial and recreational fisheries. Fish assemblages in coastal lagoons and estuaries have been widely studied in both tropical and temperate regions (Warburton 1978; Pollard 1994; Whitfield 1999; Mariani 2001; Pombo and Rebelo 2002; Jung and Houde 2003; Perez-Ruzafa et al. 2004). Regardless of geographical region, such coastal habitats can play a crucial role in providing shelter (Orth et al. 1984; Heck Jr. et al. 2003) and trophic resources for various stages of many marine fish species (Perez-Ruzafa et al. 2004). Adjacent coastal systems can theoretically provide similar ecological functions, but it has been shown that coastal lagoons contain assemblages distinct from adjacent open coastal habitats (Ishitobi et al. 2000).

Fish assemblages in estuaries are largely structured by abiotic gradients (Kupschius and Tremain 2001) that include salinity (Wagner 1999; Martino and Able 2003), temperature (Maes et al. 2004), dissolved oxygen (Weisberg et al. 1996; Eby and Crowder 2004), habitat heterogeneity and structure such as reefs or submerged vegetation (Heck Jr. et al. 2003; Martino and Able 2003) and turbidity (Cyrus and Blaber 1992). Coastal lagoons are arranged differently than the more typical east coast estuaries that have strong salinity gradients due to higher freshwater input (Yanez-Arancibia et al. 1994). These systems are more influenced by oceanic exchange via inlets as opposed to watershed effects commonly examined in drowned river valley estuaries. Therefore, the fish assemblages may be structured differently with less influence from watershed effects and greater influence from oceanic proximity.

Like other lagoon systems, the coastal embayments of Maryland (USA) located on the Delmarva Peninsula (the coastal plain peninsula between the Chesapeake Bay and Atlantic Ocean, comprised of the states of Delaware, Maryland, and Virginia) support a diverse array of fish and invertebrates (Cargo 1958; Schwartz 1961; Schwartz 1964; Wazniak et al. 2004). These four coastal embayments each have distinct characteristics related to flow, area, depth, connectivity to ocean waters, and anthropogenic habitat alterations (see further description below). Still, the embayments share connections with the ocean and are well connected as a series of lagoons. The degree of similarity of fish assemblage between embayments remains an open question, and one of consequence to similar lagoon systems elsewhere. In this study, I hypothesized that the four embayments that comprise Maryland's lagoon complex support differing assemblages of finfish and blue crab populations, and that these differences are related to marine influences (here, related to the principal connection of the Ocean City Inlet). Similarly, because this is a temperate system, I hypothesized that spatial differences in assemblage structure will vary seasonally. Assemblage structure related to season and embayment must also be examined in the context of water quality parameters known to affect fish distributions in estuarine systems (i.e., temperature, salinity, dissolved oxygen, and bottom structure).

The definition of estuary has been the subject of extensive and controversial discussion, due in part to the several geomorphological features and physiochemical characteristics of these systems (Lalli and Parsons 1997). Here the term estuarine coastal lagoon is used to indicate a shallow body of water lying parallel to the neritic shore and separated from it by a barrier (e.g., barrier island) except for small connections where

freshwater enters from land or inlets to the ocean (Lankford 1977). In this, they differ geologically from typical estuaries, which are defined in terms of the hydrological conditions and gradients of salinity and temperature (density) (Fairbridge 1980). The particular abiotic attributes of lagoons (shallowness, high turbidity, nature of the substrate, temperature fluctuation, salinity, and oxygen levels) associated with its high biotic productivity often offer excellent conditions for colonization by many marine species of fish (Pollard 1994; Yanez-Arancibia et al. 1994; Nixon et al. 2001; Poizat et al. 2004; Vega-Cendejas and Hernandez de Santillana 2004). Fish utilize coastal embayments during different phases of their life-cycle: as a nursery, juveniles apparently benefit from abundant food supply and protection from predators (Ross and Epperly 1985; Perez-Ruzafa et al. 2004); as a permanent home for sedentary species (Szedlmayer and Able 1996; Mariani 2001); and as a migratory route for diadromous species (Mariani 2001; Paperno et al. 2001). Much information is now available on the life cycles of fishes in relation to the lagoon-estuarine environment (Day Jr. and Yanez-Arancibia 1985; Henderson and Margetts 1988; Yanez-Arancibia et al. 1994) and various explanations have been proposed to link abundances with environmental parameters (Nixon 1982; Whitfield 1999; Martino and Able 2003).

Previous investigations examining ichthyofaunal assemblages in coastal areas have identified several environmental parameters that affect structural components of the assemblage (Whitfield 1999; Desmond et al. 2002; Perez-Ruzafa et al. 2004). The primary variables affecting species in nearshore regions are salinity (Wagner 1999; Martino and Able 2003; Poizat et al. 2004), temperature (Desmond et al. 2002), chlorophyll a (Perez-Ruzafa et al. 2004), oxygen (Eby and Crowder 2004), and habitat

heterogeneity, particularly with regard to structural components such as vegetation or reef systems (Whitfield 1999; Martino and Able 2003). In typical drowned river valley estuarine systems, environmental gradients are often steep due to the large watershed to surface area ratio and associated freshwater influence. This can cause dramatic shifts in salinity and temperature that estuarine organisms must either adapt to, or avoid (Vernberg 1982). Coastal lagoons, on the other hand, are less influenced by high freshwater flow (less stratification) but are influenced by winds, tides, and location of oceanic input (Mariani 2001; Poizat et al. 2004). Assemblages in these systems, therefore, must contain those species capable of thriving under dynamic conditions or shift structure as conditions vary. Additionally, many species use estuaries facultatively at different phases of their life cycles that include nursery (Gillanders et al. 2003), reproduction (Hagan and Able 2003), and opportunistic foraging (Franzoi et al. 1993). Many of these life history traits are ontogenetic in nature and thus a species will occupy multiple niches throughout its life (Able and Fahay 1998). Therefore, one would expect seasonal turnovers in species in temperate estuaries and lagoons and that the phenology (seasonal appearance of species) can be a characteristic attribute of lagoon/estuarine systems.

Yáñez-Arancibia (1994) noted differential use of lagoon-estuarine systems along a latitudinal gradient, pointing out that there are a greater number of fish species in tropical and sub-tropical lagoon-estuarine ecosystems than in comparable temperate or boreal systems (Yanez-Arancibia 1985). However, some large temperate-latitude estuaries, such as Chesapeake Bay, Delaware Bay, and Narraganset Bay, also have a large number of fish species (Day Jr. et al. 1989; Jung and Houde 2003). The theory of island biogeography (MacArthur and Wilson 1967), which relates species number to the size of

habitats could possibly provide a framework for these exceptions. Additionally, considering estuaries and lagoons as nutrient sinks, the agricultural paradigm of eutrophication promoting secondary production may also play a role in increasing fish production and diversity (Lee and Jones 1991; Pauly and Yanez-Arancibia 1994; Jeppesen et al. 2000; Nixon and Buckley 2002).

Like other coastal environments, the coastal lagoons of Maryland, Delaware, and Virginia play an important role in the regional economies of these three states due to increased tourism and residents moving into the watershed and increasing demands on the natural resources for both commercial and recreational purposes (Derickson and Price 1973; Maxted et al. 1997; Wazniak et al. 2004). As urban and suburban development proceeds and society places greater demands on the shared public resources, it is essential that we better understand the complexities of the biota that inhabit these diverse lagoons. The last comprehensive analysis of the ichthyofauna from Maryland's lagoonal system (Schwartz 1961; Schwartz 1964) approached the issue of fish utilization of the Delmarva coastal bays from primarily a qualitative perspective and did not attempt a quantitative analysis of assemblage structure. Still, Schwartz (1964) speculated that there were differences in the faunal composition of the northern bays (Assawoman and Isle of Wight) versus the southern bays (Sinepuxent and Chincoteague) due largely to the stronger currents deflected north from the inlet at Ocean City (Fig 1). Because of the various influences on the specific embayments including eutrophication (Boynton et al. 1996), abiotic variability (Wazniak et al. 2004), habitat variation including submerged macrophyte coverage (Wazniak et al. 2004), wetland acreage, anthropogenic alteration (Maxted et al. 1997), and differing geomorphologies (Pritchard 1960; Cerco et al. 1978),

it is likely that each embayment would support a distinct, identifiable fish and macroinvertebrate community. Furthermore, due to the noted transient nature of many of the species likely to use these systems, I characterized seasonal changes in the assemblage.

I used several quantitative measures of assemblage structure, including individual species abundance, indices of diversity and species richness, and multivariate approaches to evaluate spatial and seasonal patterns of species co-occurrence. This analysis was supported by a twelve-year survey conducted by the state of Maryland. Community ecology has struggled with the definition of biodiversity and its measurements and causal factors (Hurlbert 1971; Peet 1974; Lamshead et al. 1983; Weisberg et al. 1996; Wagner 1999). An analysis of large data sets spanning several years with fixed stations provides a rare opportunity to use multiple metrics of diversity in order to gain insight into assemblage dynamics (Peet 1975; Desmond et al. 2002; Mouillot et al. 2005). In this study, I used several assemblage metrics each as a lens, which can potentially describe multiple attributes of the community and its spatial and temporal structure.

Coastal lagoons by definition are shallow systems and the Delmarva embayments are typical in this regard (mean depths of embayments <2 m). Nonetheless, there are distinct “profundal” habitats in the channels (largely tidally mediated) and large expanses of littoral habitat that grade into marshes, mussel beds, fields, lawns, docks, and bulkheads. The design of the Maryland fish survey includes two capture techniques (seine and trawl) that sample community data from these two habitat types (profundal and littoral). Although comparisons of data from differing gear types can be fraught with errors associated with sampling efficiency (for example area covered, mesh size

differences, and behavioral aspects of specific species), each gear is efficient for the intended habitat type and comparisons of species richness and concurrence of trends for species captured will be treated separately.

In the mid-Atlantic coastal lagoons of Maryland, fluctuations in salinity, temperature, and dissolved oxygen vary primarily on a seasonal basis, but important differences also occur between embayments (Boynton et al. 1996; Wazniak et al. 2004). Although biologists frequently study the influence of one or a few environmental factors at a time, it is important to recognize that many factors may act in concert to affect an organism's physiology and behavior (Vernberg 1982). Individual species are expected to have differing preferences and tolerance ranges for mixtures of environmental factors. To evaluate environmental influences on assemblage structure, I assumed that such factors would influence assemblages by their influence on dominant individual species. Multiple environmental factors were analyzed through Classification and Regression Tree (CART) analysis.

The Delmarva coastal lagoon complex represents an opportunity to analyze assemblage structure related to season, and habitat attributes specific to constituent embayments. Here, I analyzed a moderately long-term data set (12 years, 1991-2002) available from Maryland Department of Natural Resources. Objectives were:

- a) Examine interannual variability and spatial patterns in key (dominant) species and test associations between their abundance and environmental parameters.
- b) Evaluate differences in assemblage structure among the four primary embayments in Maryland and identify the discriminating species.

- c) Contrast metrics of biodiversity across the four embayments and compare these metrics to other assemblage characteristics.
- d) Characterize seasonal changes in assemblage structure.

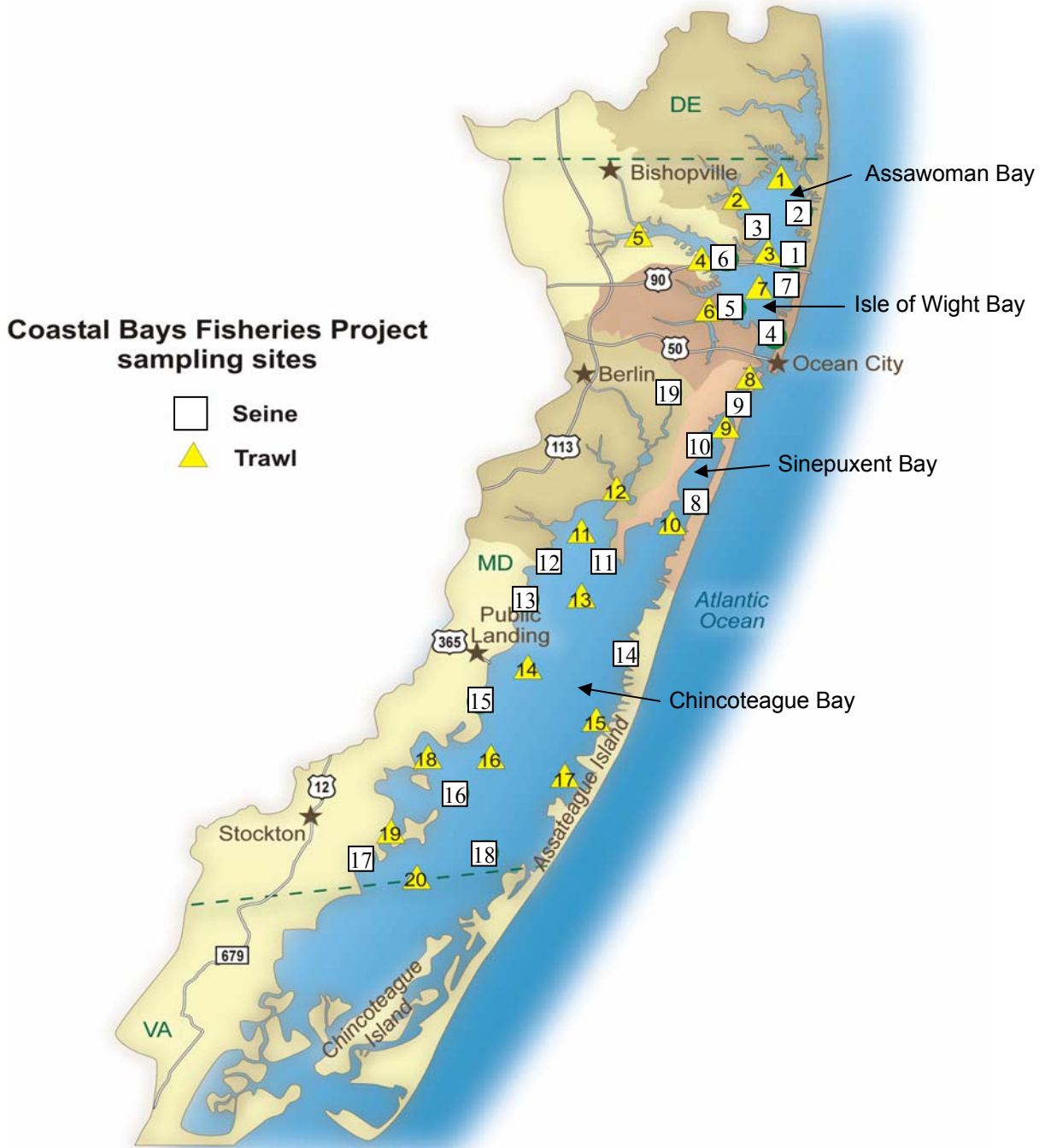
MATERIALS AND METHODS

I. Study Area

The Maryland coastal lagoon complex extends along the entire Atlantic coast of the state behind the barrier islands of Assateague and Fenwick. The system consists of four major embayments: the southern embayments bays of Chincoteague and Sinepuxent, and the northern embayments of Assawoman and Isle of Wight (Figure 1). The principal population center is the resort town of Ocean City, located on Fenwick Island. These embayments are connected to several smaller tidal creek systems with low freshwater discharge and exhibit little stratification due primarily to wind driven mixing (Cercio et al. 1978). As a result of the low relief landscape and sandy soils, rainwater permeates the ground quickly and seeps into the bays as groundwater that can have a localized effect on salinity.

The coastal bays and associated tributaries are shallow, with an average depth of 1.2 m. The watersheds of the embayments are relatively small compared to open water areas: 45,246 ha, or about 1.7 times the area of the bays (Boynton et al. 1996). By comparison, the Chesapeake Bay's watershed is 16-fold that of open-water areas. Additionally, the coastal bays maintain a constricted connection to the ocean at the Ocean City Inlet and at the southern end of Chincoteague Bay via a series of channels. Pritchard (1960) estimated the flushing rate for Chincoteague Bay at 7.5% per day. Flushing rates and oceanic exchange of the northern embayments are predicted to be higher, but still substantially less than large estuarine systems such as the Chesapeake or Delaware Bays. The combination of reduced freshwater input and limited oceanic exchange results in low water replacement times and thus these systems retain nutrients, sediments, and other

Figure 1: Map of study location with trawl (triangle) and seine (circle) sites. (courtesy MD Dept. of Natural Resources; Wazniak et al 2004)



inputs from the watershed. In terms of eutrophication, the northern coastal bays (Assawoman and Isle of Wight) are relatively enriched with nitrogen (median N [2000-2002] > 1 mg l⁻¹) while the southern bays (Sinepuxent and Chincoteague) exhibit the lowest total nitrogen concentrations (median N [2000-2002] < 1 mg L⁻¹) (Wazniak et al. 2004). Habitat availability in terms of structural complexity varies between the four systems. Over the course of this survey, seagrass coverage in each of the bays has increased dramatically (Figure 2).

II. Survey Design

Beginning in 1972, the Maryland Department of Natural Resources (DNR) initiated a biological survey of the four coastal embayments with the intent to inventory living resources utilizing these systems. The survey methodology was standardized in 1991 and this analysis consists of data collected 1991-2002. Twenty fixed trawl stations and eighteen fixed seine stations were established throughout the four embayments (Table 1).

Trawl samples were collected by a 4.8 m semi-balloon trawl with tickler chain towed at an approximate speed of 3 knots for 6 minutes per tow. The mean estimated area sampled per tow is 1864 m². Samples were sorted by species and identified in the field. Seine samples were collected using a beach seine 30.5 m in length, 1.8 m tall, with a bag (1.8 m X 1.8 m X 1.8m) with a 6.3 mm nylon knotless mesh. The seine was deployed in a quarter circle pull at most sites covering an area of 117 m²; however some sites required a modified deployment. At these sites, the area covered was adjusted to remain the same although the deployment with regard to the shore differed somewhat.

Figure 2: Seagrass coverage by embayment 1991-2002. Note the difference in scale along the y-axis for each bay. Data courtesy Virginia Institute of Marine Science. (www.vims.edu/bio/sav)



Table 1: Number of sampling stations for each bay by gear type for MD DNR coastal bays fish monitoring survey, 1991-2002.

Gear Type	Embayment			
	Assawoman	Isle of Wight	Sinepuxent	Chincoteague
Seine	3	4	3	8
Trawl	3	4	3	10

Samples were processed similarly between surveys. Water quality and environmental parameters collected at each sampling event included dissolved oxygen (mg/l), salinity, temperature, tidal state, and weather phenomena. From 1991-2000, dissolved oxygen and temperature were measured in situ using a YSI © 51B, and salinity was measured with a YSI © 33 meter. Beginning in 2000, dissolved oxygen, temperature, and salinity were all measured using a YSI © 85 meter. Trawl sites were sampled seasonally (see Table 2 for seasonal assignments) whereas seine sites were only sampled two times per year. All data was transcribed into an Access (Microsoft Office, 2000) database and data manipulation was done in Excel spreadsheets.

III. Statistical Analyses

Because sampling effort was uneven between the four embayments, catch per unit effort (CPUE; i.e., catch per tow or haul) statistics were calculated for each bay. Furthermore, CPUE was calculated for each site with sample catches pooled across years. Water quality parameters were tested for significance using a two-way ANOVA with site nested within embayment and embayment nested within season (Sokal and Rohlf 1995). Assumptions for the ANOVA (normality and homogeneity of variances) were examined both visually and using the Shapiro-Wilkes test statistic. Water quality data met the assumptions of the ANOVA model and did not require transformations.

Single Species

Table 2: Seasonal assignment for sampling month by gear type for MD DNR's coastal fish survey, 1991-2002

Gear	Month sampled	Season Assignment	Range of Dates
Seine	June, July	Early Summer	June 14 – July 27
Seine	September	Late Summer	Sept. 9 – Sept. 30
Trawl	April, May	Spring	April 1 – May 31
Trawl	June, July	Early Summer	June 10 – July 31
Trawl	August, September	Late Summer	Aug. 5 – Sept. 30
Trawl	October	Fall	Oct. 1 – Oct. 30

Variations in individual species abundance were analyzed for differences among bays, seasons, and for possible associations with environmental variables. Individual species abundance was first transformed if necessary, and then analyzed for potential bay or seasonal effects. If normality was not achieved through data transformation, a Kruskal-Wallis non-parametric test was used to determine departure from the null hypothesis (Sokal and Rohlf 1995). To investigate environmental associates with patterns of species abundance, I employed a Classification and Regression Tree (CART) analysis (Breiman et al. 1984). CART is a flexible nonparametric multivariate analysis that provides dichotomous keys for each species' abundance based on both continuous and categorical variables, including DO, salinity, temperature, embayment, submerged aquatic vegetation (SAV) coverage of each bay (specific to year). The CART algorithm constructs a structural tree (dichotomous key) by repeated splits of the subsets of the multivariate records into descendent pairs of subsets selecting the combination of parameters that most minimize error. The model was run allowing interaction of variables to better estimate confounding responses. For each dichotomous branch, a criteria coefficient is estimated (e.g., less than or greater than 20 C). If the resulting node was an interaction of a categorical and continuous variable, the coefficient was nonsensical (and therefore not presented). I included potential explanatory variables: DO, temperature, embayment, tidal state, SAV coverage, and abundance of other dominant species.

Community Analysis

The fish abundance data was transformed after determining non-normality of the dataset, due primarily to a large number of zero observations for a given species. Various

attempts at transforming the data (e.g., fourth-root and $\ln + 1$) failed to normalize the distribution, so non-parametric analyses were selected. There are several methods available for reducing the high dimensionality of species abundance data to evaluate the assemblage structure. Each of these techniques is sensitive to various aspects of structural variation and thus each has its advantages and disadvantages (Hurlbert 1971; May 1975; Peet 1975; Magurran 1991). In order to adequately assess various attributes of the community, and to make certain that defensible conclusions are drawn about emergent patterns, I used a variety of community descriptors to provide a broad view of community structure. Three univariate measures of diversity were calculated: species richness (an overall measure of niche availability) and the Shannon-Weiner index (H), which is most sensitive to changes in rare species (Peet 1974).

As a complement to the traditional univariate descriptors of community structure, k -dominance curves were generated to show structure relative to dominant species and number of individuals (Lambhead et al. 1983). K -dominance curves are constructed by plotting the log species rank (k) against the percent cumulative abundance. More diverse communities (less dominance) have less arced curves with lower slopes, and curves from different populations or samples plotted on the same graph may be compared for diversity if the curves do not intersect (Lambhead et al. 1983). These curves were constructed to compare dominance patterns between bays and seasons for each gear type.

The assemblage metrics described thus far do not take into account the individual species in the samples. Thus, they are ineffective at detecting events or factors that cause one species to occur more or less abundantly than others that share similar ecology. By using these emergent metrics of community alone, one could potentially fail to observe

structural changes in the community structure; therefore additional analyses that include species identity are considered (Rice 2000; Martino and Able 2003). Ordination and clustering techniques are multivariate procedures that provide a means for assessing spatial and temporal differences while taking species composition into account. Of the available methods, non-metric multidimensional scaling (MDS) has been shown to be a robust non-parametric technique (Fasham 1977; Clarke and Ainsworth 1993) that can ordinate non-parametric data (Clarke 1993). MDS plots are based on similarity matrices that are constructed using an abundance weighted Bray-Curtis similarity index. MDS ordination positions samples in two- (or three, if necessary) dimensional space so that the relative distance between samples reflects their relative dissimilarity, with the samples most similar to one another occurring closest together. Sample sites were used as the sampling unit (pooled across season and year) to look for patterns among embayment assemblages. The goodness of fit of the data distributed across two composite axes was indexed by the stress coefficient, which tends towards zero when the data are perfectly represented. Stress may be thought of as the distortion involved in representing multi-dimensional data in two dimensions (Field et al. 1982). MDS plots are deemed an acceptable representation of the data when stress values are < 0.2 (Clarke 1993).

To examine the underlying structure of the MDS plots, a cluster analysis were performed. The cluster analysis represents the same data as the MDS; therefore any inconsistencies between the two approaches should be attributed to the inadequacy of the two-dimensional ordination (i.e., the MDS approach). The scale on the dendrogram of the cluster analysis represents dissimilarity.

Differences in assemblage structure between embayments or seasons identified by MDS was further analyzed by another non-parametric analysis: permutation-based, one-way analysis of similarity (ANOSIM) (Clarke 1993). Where appropriate, R-statistic values were used to determine the dissimilarity of groups. R values were protected for pairwise comparisons and represent the extent of similarity between two groups. Values close to 1 indicated very different composition between samples, while values near zero are expected to exhibit little difference. ANOSIM was used to test the null hypothesis that within seasons, no changes in community structure were observed between the embayments, and secondly to test that within single embayment assemblages, there was no difference between seasons.

To establish which species most contributed to community structure, the SIMPER (similarity percentage analysis) approach was used (Clarke 1993). This method identifies which species (due to their relative abundances) are contributing most to the overall dissimilarity between defined (a priori) groups. Groups were defined by either specific embayment or season.

ANOVA and CART analyses were carried out in Systat version 11.0. Multivariate community analyses including cluster analysis, MDS, ANOSIM, and SIMPER procedures were carried out in the Community Analysis Package (Pisces Conservation, Ltd., 1997). Species richness and diversity calculations were performed using the software program Species Richness and Diversity (Pisces Conservation, Ltd., 1997).

RESULTS

Water quality

Between April 1991 and October 2002, salinity of all the embayments varied between 8.1 in April 2000 (Isle of Wight) to 37.4 in September 2002 (Chincoteague), and water temperatures from 9.9° C in April 2000 (Sinepuxent) to 32.3° C in July 1999 (Chincoteague). Dissolved oxygen concentrations ranged from 1.8 mg l⁻¹ in August 2001 (Isle of Wight) to 11.2 mg l⁻¹ in September 2000 (Isle of Wight). Salinity and temperature were significantly different between bays and seasons for both littoral and deep water environments (Table 3 and Figure 3) whereas DO differed only between seasons. Salinity was the only parameter that differed between sites (as a nested component of the ANOVA). In addition, mean salinity was highest in Sinepuxent (mean = 29.3) and lowest in Isle of Wight (mean = 26.8). Pairwise comparisons between bays indicated no difference between Assawoman and Isle of Wight. Sinepuxent bay was typically cooler (mean temp = 20.5 C) than the other three embayments; temperatures did not differ significantly among these three bays (mean temp = 21.7 C). Each embayment had seasonal differences in all water quality parameters (Table 4).

Catch characteristics

From 1991-2002, the surveys conducted 1744 trawl and 435 seine deployments collecting 286,486 and 445,890 specimens, and 96 and 89 species, respectively (Table 5). Trawl samples were dominated by eight species comprising 95% of the catch. These were in descending order of abundance: *Callinectes sapidus*, *Anchoa mitchilli*, *Leiostomus xanthurus*, *Cynoscion regalis*, *Clupea harengus*, *Micropogonias undulatus*, *Paralichthys dentatus*, *Bairdiella chrysoura*, and *Brevoortia tyrannus* (for common

Table 3: Results of a two-factor nested model ANOVA comparing water quality data collected in littoral (seine) and profundal (trawl) environments. Hierarchy of nesting indicated by parentheses. Significance levels are indicated by * $p < 0.05$, ** $p \leq 0.01$, *** $p \leq 0.001$, NS non-significant at the $\alpha = 0.05$ level.

Source of variation	df	DO	Trawl	
			Salinity	Temperature
Season	3	***	***	***
Bay	3	NS	***	***
Site (Bay(Season))	65	NS	***	NS
Error	1658			
Seine				
Season	1	*	***	***
Bay	3	NS	**	**
Site (Bay(Season))	28	NS	NS	NS
Error	402			

Figure 3: Temperature and salinity across years for profundal (trawl) survey (a) and littoral (seine) survey (b). Values are pooled across all embayments for a composite mean. Bars represent standard error. Temperature is given in °C and salinity as parts per thousand.

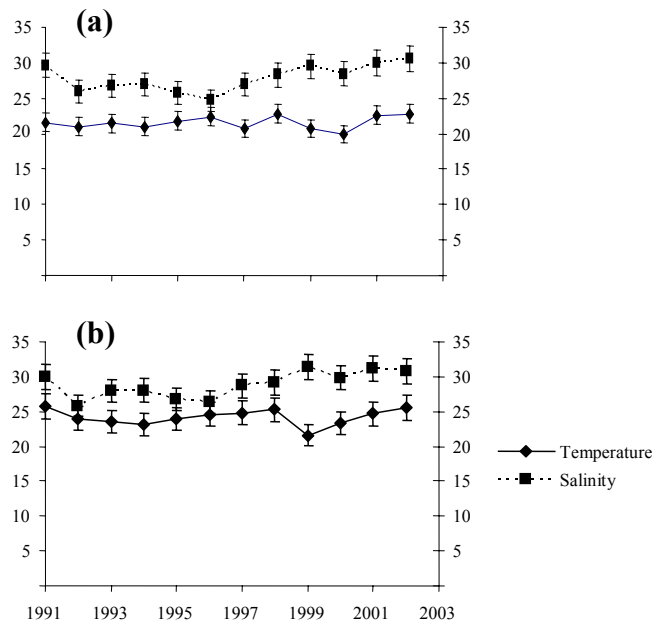


Table 4: Mean water quality values collected during trawl survey across all seasons by bay, and by bay and season.

<u>Bay</u>	<u>Mean (\pmSE)</u>		
	<u>Temperature C</u>	<u>Salinity</u>	<u>Dissolved Oxygen (mg L⁻¹)</u>
Assawoman	21.48 (0.32)	26.8 (0.23)	6.26 (0.125)
Isle of Wight	21.78 (0.277)	26.7 (0.20)	6.34 (0.11)
Sinepuxent	20.54 (0.327)	29.3 (0.24)	6.26 (0.129)
Chincoteague	21.9 (0.183)	28 (0.136)	6.34 (0.074)
<u>Spring</u>			
Assawoman	23.0 (0.49)	27.0 (0.38)	5.89 (0.219)
Isle of Wight	21.8 (0.45)	26.7 (0.34)	6.34 (0.197)
Sinepuxent	21.7 (0.51)	29.2 (0.38)	6.03 (0.228)
Chincoteague	19.2 (0.31)	28.1 (0.241)	6.93 (0.142)
<u>Early Summer</u>			
Assawoman	20.6 (0.48)	27.0 (0.39)	6.37 (0.187)
Isle of Wight	21.8 (0.43)	27.3 (0.35)	6.51 (0.172)
Sinepuxent	18.6 (0.49)	30.4 (0.40)	6.35 (0.196)
Chincoteague	24.8 (0.31)	26.2 (0.238)	5.96 (0.118)
<u>Late Summer</u>			
Assawoman	20.4 (0.48)	26.5 (0.37)	6.65 (0.193)
Isle of Wight	21.7 (0.43)	26.3 (0.34)	6.28 (0.176)
Sinepuxent	20.3 (0.49)	28.8 (0.38)	6.26 (0.193)
Chincoteague	23.4 (0.30)	29.4 (0.24)	6.0 (0.13)
<u>Fall</u>			
Assawoman	22.7 (0.65)	26.9 (0.51)	5.72 (0.286)
Isle of Wight	21.8 (0.59)	26.9 (0.46)	6.09 (0.259)
Sinepuxent	22.8 (0.67)	28.5 (0.52)	6.41 (0.286)
Chincoteague	18.5 (0.43)	28.7 (0.33)	6.97 (0.18)

Table 5: List of species caught and CPUE (no. per gear deployment) for each embayment and gear type, 1991-2002

<u>Common Name</u>	<u>Species</u>	<u>Assawoman</u>		<u>Chincoteague</u>		<u>Isle of Wight</u>		<u>Sinepuxent</u>	
	<u>Scientific Name</u>	<u>Trawl</u>	<u>Seine</u>	<u>Trawl</u>	<u>Seine</u>	<u>Trawl</u>	<u>Seine</u>	<u>Trawl</u>	<u>Seine</u>
	Trakidae								
SMOOTH DOGFISH SHARK ¹	<i>Mustelus canis</i>	0.0000	0.0000	0.0000	0.0000	0.0027	0.0000	0.0000	0.0000
	Rajidae								
CLEARNOSE SKATE ¹	<i>Raja eglanteria</i>	0.0217	0.0000	0.0024	0.0000	0.0356	0.0000	0.0153	0.0000
	Gymnuridae								
SMOOTH BUTTERFLY RAY	<i>Gymnura micrura</i>	0.0000	0.0000	0.0036	0.0052	0.0055	0.0000	0.0038	0.0000
	Dasyatidae								
SOUTHERN STINGRAY	<i>Dasyatis americana</i>	0.0144	0.1528	0.0071	0.2280	0.0164	0.0303	0.0192	0.0704
	Myliobatidae								
BULLNOSE RAY ²	<i>Myliobatis freminvillei</i>	0.0000	0.0139	0.0000	0.0052	0.0000	0.0101	0.0000	0.0282
	Rhinopteridae								
COWNOSE RAY	<i>Rhinoptera bonasus</i>	0.0000	0.0139	0.0012	0.0052	0.0000	0.0000	0.0000	0.0141
	Elopidae								
LADYFISH ²	<i>Elops saurus</i>	0.0000	0.0000	0.0000	0.0207	0.0000	0.0000	0.0000	0.0000
	Anguillidae								
AMERICAN EEL	<i>Anguilla rostrata</i>	0.0830	0.8056	0.2342	0.9067	0.0164	0.6869	0.0038	0.2535
	Congridae								
CONGER EEL ¹	<i>Conger oceanicus</i>	0.0000	0.0000	0.0012	0.0000	0.0055	0.0000	0.0038	0.0000
	Engraulidae								
BAY ANCHOVY	<i>Anchoa mitchilli</i>	85.7581	43.9722	31.8371	65.1658	64.6767	47.0808	10.7778	6.9718
STRIPED ANCHOVY	<i>Anchoa hepsetus</i>	0.3069	2.5278	0.1058	1.0933	0.3863	3.5960	0.4215	1.3944
	Clupeidae								
GIZZARD SHAD	<i>Dorosoma cepedianum</i>	0.0036	0.0278	0.0012	0.0052	0.0055	0.0202	0.0000	0.0000
ATLANTIC MENHADEN	<i>Brevoortia tyrannus</i>	0.4910	164.4444	1.3306	164.4444	1.7616	212.1111	0.0000	38.7606
ATLANTIC HERRING	<i>Clupea harengus</i>	11.4477	0.0000	3.8644	0.0000	8.8603	0.0000	0.2261	0.0563
BLUEBACK HERRING	<i>Alosa aestivalis</i>	0.1047	0.1250	0.1605	0.0933	0.1205	0.0000	0.0307	0.0000
	Synodontidae								
INSHORE LIZARDFISH	<i>Synodus foetens</i>	0.8556	0.8333	0.6100	0.4301	1.1370	1.7778	0.8429	0.8169

Species		Assawoman		Chincoteague		Isle of Wight		Sinepuxent		
<u>Common Name</u>	<u>Scientific Name</u>	<u>Trawl</u>	<u>Seine</u>	<u>Trawl</u>	<u>Seine</u>	<u>Trawl</u>	<u>Seine</u>	<u>Trawl</u>	<u>Seine</u>	
Batrachoididae										
OYSTER TOADFISH	<i>Opsanus tau</i>	0.1444	2.6806	0.2069	0.7202	0.2027	1.0101	0.0383	1.2535	
Gobiesocidae										
SKILLET FISH	<i>Gobiesox strumosus</i>	0.0072	0.0139	0.0095	0.0725	0.0055	0.0000	0.0000	0.0000	
Gadidae										
ATLANTIC POLLOCK ¹	<i>Pollachius virens</i>	0.0000	0.0000	0.0000	0.0000	0.0027	0.0000	0.0038	0.0000	
SPOTTED HAKE ¹	<i>Urophycis regia</i>	0.3502	0.0000	0.1724	0.0000	0.4082	0.0000	0.1188	0.0000	
RED HAKE ¹	<i>Urophycis chuss</i>	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0038	0.0000	
Ophidiidae										
STRIPED CUSK EEL	<i>Ophidion marginatum</i>	0.0000	0.0000	0.1332	0.0155	0.0055	0.0000	0.0230	0.0000	
Belonidae										
ATLANTIC NEEDLE FISH	<i>Strongylura marina</i>	0.0000	2.1806	0.0012	2.0052	0.0055	1.9596	0.0000	4.0423	
Hemiramphidae										
HALF BEAK ²	<i>Hyporhamphus meeki</i>	0.0000	0.5139	0.0000	0.0363	0.0000	0.0000	0.0000	0.5070	
Cyprinodontidae										
SHEEPSHEAD MINNOW	<i>Cyprinodon variegatus</i>	0.0036	2.1944	0.0143	0.6373	0.0000	0.7475	0.0000	0.4085	
Fundulidae										
BANDED KILLIFISH ¹	<i>Fundulus diaphanus</i>	0.0000	0.0000	0.0024	0.0000	0.0027	0.0000	0.0000	0.0000	
SPOTFIN KILLIFISH ²	<i>Fundulus luciae</i>	0.0000	0.0000	0.0000	0.0000	0.0000	0.0404	0.0000	0.0000	
MUMMICHOG	<i>Fundulus heteroclitus</i>	0.0000	17.4028	0.0963	8.4093	0.4493	8.2727	0.0000	17.8028	
STRIPED KILLIFISH	<i>Fundulus majalis</i>	0.0000	9.4583	0.0071	2.9016	0.0164	10.1616	0.0000	12.6479	
RAINWATER KILLIFISH	<i>Luciana parva</i>	0.0000	6.5278	0.4875	2.9741	0.0055	0.8485	0.0345	8.4085	
Atherinopsidae										
ATLANTIC SILVERSIDE	<i>Menidia menidia</i>	0.2383	105.2083	0.5505	53.7047	0.9562	252.4747	0.2567	352.0141	
INLAND SILVERSIDE	<i>Menidia beryllina</i>	0.0000	1.4583	0.0000	0.1295	0.0740	0.1515	0.0000	0.0000	
ROUGH SILVERSIDE	<i>Membras martica</i>	0.0000	4.7778	0.0083	0.5855	0.0000	3.1818	0.0000	0.2113	
Gasterosteidae										
FOURSPINE STICKLEBACK	<i>Apeltes quadracus</i>	0.0072	1.0000	0.7812	0.9637	0.0164	0.1212	0.0000	0.1690	
THREESPINE STICKLEBACK ²	<i>Gasterosteus aculeatus</i>	0.0000	0.0417	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	

Species		Assawoman		Chincoteague		Isle of Wight		Sinepuxent		
<u>Common Name</u>	<u>Scientific Name</u>	<u>Trawl</u>	<u>Seine</u>	<u>Trawl</u>	<u>Seine</u>	<u>Trawl</u>	<u>Seine</u>	<u>Trawl</u>	<u>Seine</u>	
Syngnathidae										
LINED SEAHORSE	<i>Hippocampus erectus</i>	0.0578	0.0278	0.0392	0.0155	0.1397	0.1414	0.0651	0.1268	
DUSKY PIPEFISH	<i>Syngnathus floridae</i>	0.0000	0.0139	0.0012	0.0000	0.0000	0.0000	0.0000	0.0000	
NORTHERN PIPEFISH	<i>Syngnathus floridae</i>	0.2347	2.8194	0.7313	0.9534	0.3342	1.3030	0.1456	1.0986	
Fistularidae										
BLUESPOTTED CORNETFISH	<i>Fistularia tabacaria</i>	0.0000	0.0278	0.0048	0.0000	0.0164	0.0202	0.0077	0.0423	
Triglidae										
NORTHERN SEAROBIN	<i>Prionotus carolinus</i>	0.3105	0.4583	0.1795	0.2694	0.3699	0.2525	0.8697	0.1127	
STRIPED SEAROBIN	<i>Prionotus evelans</i>	0.1191	0.0694	0.0476	0.0052	0.1068	0.0808	0.0498	0.0423	
Stromateidae										
HARVESTFISH	<i>Peprilus alepidotus</i>	0.0072	0.0000	0.0000	0.0881	0.0027	0.1515	0.0000	3.5775	
BUTTERFISH	<i>Peprilus triacanthus</i>	0.3574	0.0278	0.0119	0.0000	0.1726	0.0404	0.0038	0.0000	
Ammodytidae										
AMERICAN SAND LANCE ¹	<i>Ammodytes americanus</i>	0.0000	0.0000	0.0012	0.0000	0.0000	0.0000	0.1149	0.0000	
Carangidae										
ATLANTIC MOONFISH ¹	<i>Selene setapinnis</i>	0.0108	0.0000	0.0012	0.0000	0.0000	0.0000	0.0000	0.0000	
LOOKDOWN	<i>Selene vomer</i>	0.0325	0.1528	0.0024	0.0104	0.0548	0.2222	0.0000	0.0704	
BLUE RUNNER ²	<i>Caranx crysos</i>	0.0000	0.0000	0.0000	0.0000	0.0000	0.0101	0.0000	0.0000	
CREVALLE JACK	<i>Caranx hippos</i>	0.0794	0.1111	0.0131	0.1399	0.0822	0.3333	0.0000	0.1408	
HORSE-EYE JACK ²	<i>Caranx latus</i>	0.0000	0.0000	0.0000	0.0000	0.0000	0.0101	0.0000	0.0000	
PERMIT ¹	<i>Trachinotus falcatus</i>	0.0000	0.2083	0.0000	0.0052	0.0000	0.5253	0.0000	0.2394	
ROUGH SCAD	<i>Trachurus lathami</i>	0.0000	0.0000	0.0000	0.0000	0.0027	0.0101	0.0000	0.0000	
Rachycentridae										
COBIA	<i>Rachycentron canadum</i>	0.0000	0.0000	0.0000	0.0000	0.0110	0.0202	0.0000	0.0000	
Scombridae										
KING MACKEREL ¹	<i>Scomberomorus cavalla</i>	0.0000	0.0000	0.0048	0.0000	0.0110	0.0000	0.0000	0.0000	
SPANISH MACKEREL ¹	<i>Scomberomorus maculatus</i>	0.0181	0.0000	0.0048	0.0000	0.0164	0.0000	0.0000	0.0000	
Mullidae										
DWARF GOATFISH	<i>Upeneus parvus</i>	0.0000	0.0000	0.0012	0.0000	0.0027	0.0000	0.0000	0.0141	

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Mugilidae										
STRIPED MULLET	<i>Mugil cephalus</i>	0.0000	0.5139	0.0048	0.6995	0.0055	3.9293	0.0000	0.3380	
WHITE MULLET	<i>Mugil curema</i>	0.0000	24.5139	0.0000	3.1658	0.0795	43.8283	0.0000	8.0704	
Sphyraenidae										
NORTHERN SENNET	<i>Sphyraena borealis</i>	0.0000	0.3056	0.0000	0.0052	0.0027	0.3131	0.0000	0.3380	
Uranoscopidae										
NORTHERN STARGAZER	<i>Astroscopus guttatus</i>	0.0000	0.0139	0.0024	0.0207	0.0110	0.0000	0.0690	0.0000	
Sciaenidae										
SILVER PERCH	<i>Bairdiella chrysoura</i>	2.6895	81.9444	2.0892	26.3782	2.0329	24.9495	0.2375	21.7042	
SPOTTED SEA TROUT	<i>Cynoscion nebulosus</i>	0.0000	0.1111	0.0095	0.1399	0.0000	0.0000	0.0000	0.0704	
WEAKFISH	<i>Cynoscion regalis</i>	39.9639	0.2222	2.7729	0.4767	18.1671	0.1414	0.3793	0.0000	
SPOT	<i>Leiostomus xanthurus</i>	76.2238	76.7917	36.0357	27.6995	57.0493	41.1111	3.9042	63.6338	
NORTHERN KINGFISH	<i>Menticirrhus saxatilis</i>	0.0505	0.4444	0.0369	0.2228	0.1753	0.4444	0.3218	0.1127	
CROAKER	<i>Micropogonias undulatus</i>	12.6065	0.4861	1.9548	0.4197	8.0767	1.6061	0.2682	0.0563	
BLACK DRUM	<i>Pogonias chromis</i>	0.1047	1.0000	0.0250	0.6736	0.0822	0.9394	0.0077	0.2958	
RED DRUM ²	<i>Scianops ocellatus</i>	0.0000	0.0000	0.0000	0.0155	0.0000	0.0101	0.0000	0.0423	
Ephippidae										
SPADEFISH	<i>Chaetodipterus faber</i>	0.0108	0.0000	0.0131	0.0104	0.0055	0.0000	0.0038	0.0000	
Pomatomidae										
BLUEFISH	<i>Pomatomous saltatrix</i>	0.0650	2.6528	0.0095	0.3420	0.1041	2.2323	0.0038	0.7324	
Gobiidae										
NAKED GOBY	<i>Gobiosoma bosc</i>	0.0578	3.3333	0.4637	0.5130	0.1671	0.1111	0.0192	0.6901	
GREEN GOBY	<i>Microgobius thalassinus</i>	0.0325	0.0694	0.1451	0.0466	0.0986	0.0000	0.0038	0.0000	
Blenniidae										
STRIPED BLENNY	<i>Chasmodes bosquianus</i>	0.0000	0.1944	0.0048	0.0570	0.0137	0.0202	0.0000	0.0141	
FEATHER BLENNY	<i>Hypsoblennius hentz</i>	0.0361	0.4583	0.0476	0.2021	0.0274	0.1212	0.0575	0.3521	
Moronidae										
WHITE PERCH	<i>Morone americana</i>	0.0144	0.0000	0.0262	0.0311	0.0027	0.0000	0.0000	0.0000	
STRIPED BASS	<i>Morone saxatilis</i>	0.0000	0.0694	0.0119	0.0415	0.0027	0.0303	0.0038	0.0000	
Chaetodontidae										
SPOTFIN BUTTERFLYFISH	<i>Chaetodon ocellatus</i>	0.0072	0.0000	0.0012	0.0000	0.0000	0.0202	0.0192	0.0000	

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	Gerreidae								
SPOTFIN MOJARRA	<i>Eucinostomus argenteus</i>	0.1047	1.5139	0.0036	0.1088	0.0877	3.1010	0.0307	0.9296
	Sparidae								
SHEEPSHEAD	<i>Archosargus probatocephalus</i>	0.0072	0.1667	0.0036	0.0570	0.0027	0.0303	0.0000	0.1408
SPOTTAIL PINFISH	<i>Diplodus holbrooki</i>	0.0000	0.0000	0.0000	0.0000	0.0027	0.0505	0.0000	0.0000
PINFISH	<i>Lagodon rhomboides</i>	0.0433	8.3611	0.0488	0.2694	0.2411	2.7576	0.0345	3.2254
SCUP	<i>Stenotomus chrysops</i>	0.0108	0.0417	0.0178	0.0104	0.0110	0.0202	0.0958	0.0000
	Lutjanidae								
RED SNAPPER	<i>Lutjanus campechanus</i>	0.0181	0.9722	0.0024	0.0415	0.0055	0.1818	0.0000	0.0141
GREY SNAPPER	<i>Lutjanus griseus</i>	0.0000	0.1528	0.0012	0.0000	0.0000	0.0202	0.0000	0.0000
	Haemulidae								
PIGFISH	<i>Orthopristis chrysoptera</i>	0.0469	5.4306	0.1320	0.1710	0.0849	1.8283	0.0728	1.0282
	Labridae								
TAUTOG	<i>Tautoga onitis</i>	0.0072	0.2083	0.0107	0.0000	0.0082	0.1616	0.0230	0.2676
CUNNER ¹	<i>Tautoglabrus adspersus</i>	0.0000	0.0000	0.0012	0.0000	0.0000	0.0000	0.0038	0.0000
	Serranidae								
BLACK SEA BASS	<i>Centropristis striata</i>	0.5307	0.4861	0.3853	0.0984	0.7123	0.6566	0.5364	1.1408
GAG	<i>Mycteroperca microlepis</i>	0.0000	0.5417	0.0012	0.0052	0.0000	0.1414	0.0038	0.3380
	Balistidae								
GRAY TRIGGERFISH	<i>Balistes caprisicus</i>	0.0000	0.0139	0.0024	0.0104	0.0137	0.1919	0.0307	0.1127
	Monacanthidae								
ORANGE FILEFISH ¹	<i>Aluterus schoepfi</i>	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0038	0.0000
PLANEHEAD FILEFISH ¹	<i>Monacanthus hispidus</i>	0.0036	0.0000	0.0012	0.0000	0.0137	0.0000	0.0077	0.0000
	Diodontidae								
STRIPED BURRFISH	<i>Chilomycterus schoepfi</i>	0.0072	0.0000	0.0749	0.1554	0.0247	0.0101	0.0307	0.0000
	Tetraodontidae								
SOUTHERN PUFFER ¹	<i>Lagocephalus laevigatus</i>	0.0000	0.0000	0.0012	0.0000	0.0000	0.0000	0.0038	0.0000
NORTHERN PUFFER	<i>Sphoeroides maculatus</i>	0.2888	1.9167	0.3817	0.8912	0.6438	1.9192	0.8889	2.2817
	Cynoglossidae								
BLACKCHEEK TONGUEFISH	<i>Symphurus plagiusa</i>	0.1083	0.0278	0.0809	0.2539	0.0986	0.0808	0.0192	0.0704

Species		Assawoman		Chincoteague		Isle of Wight		Sinepuxent	
<u>Common Name</u>	<u>Scientific Name</u>	<u>Trawl</u>	<u>Seine</u>	<u>Trawl</u>	<u>Seine</u>	<u>Trawl</u>	<u>Seine</u>	<u>Trawl</u>	<u>Seine</u>
	Achiridae								
HOGCHOKER	<i>Trinectes maculatus</i>	0.6679	0.0000	1.7420	0.4041	0.5890	0.0202	0.0460	0.0000
	Pleuronectidae								
WINTER FLOUNDER	<i>Pleuronectes americanus</i>	2.0325	7.9722	0.1700	0.1658	1.1041	11.1515	0.2414	8.4930
	Scophthalmidae								
WINDOWPANE FLOUNDER	<i>Scophthalmus aquosus</i>	0.1083	0.0000	0.0071	0.0052	0.3288	0.0606	0.6360	0.0141
	Paralichthyidae								
SMALLMOUTH FLOUNDER	<i>Etropus microstomus</i>	0.0650	0.0139	0.0380	0.0104	0.6466	0.3535	0.9540	0.0986
SUMMER FLOUNDER	<i>Paralichthys dentatus</i>	5.5596	1.9583	3.8312	1.9119	4.3260	1.6263	1.4866	1.1831
	Portunidae								
BLUE CRAB	<i>Callinectes sapidus</i>	51.3032	101.8056	57.6611	66.5648	57.0192	40.5859	10.8544	69.1408

¹ Present in trawl survey only

² Present in seine survey only

names, see Table 5). Seine samples were dominated by 12 species that comprised 95% of the catch, including in descending order: *Menidia menidia*, *B. tyrannus*, *C. sapidus*, *A. mitchilli*, *L. xanthurus*, *B. chrysoura*, *Mugil curema*, *Fundulus heteroclitus*, *F. majalis*, *Pleuronectes americanus*, *Lucania parva*, and *Lagodon rhomboides*. Differences in species encountered between gear types were moderate as most species were encountered in both gear types. However, differences were noticeable in the relative abundance of some demersal species (e.g., *P. dentatus*) that were abundant in trawl but were relatively rare in seine samples. Likewise, typically littoral species (e.g. *M. curema*) were rarely encountered in the trawl survey but were abundant in seine samples.

Catch per site (CPS) differed between bays and seasons for both gear types (Table 6) with the northern bays of Assawoman and Isle of Wight exhibiting higher CPS than the southerly bays of Sinexpuxent and Chincoteague. In addition, the trawl data indicated a seasonal pattern in CPS, with lower numbers of species and organisms caught in the spring and fall than in summer months.

Single Species

The species that comprised the top 95% of the catch differed significantly among bays, seasons, and years. For example, *C. sapidus* abundance was significantly different between seasons ($p < 0.0001$) consistently showing peaks in the early summer across all years (Figure 4 and 5), but exhibiting fairly low interannual variability (Coefficient of Variation [CV] of annual means= 0.33 for trawl, 0.51 for seine). *C. sapidus* distribution

Table 6: Number of species, individuals, and mean catch per site for each embayment and season by gear type, 1991-2002

Season	Bay	<u>No. Species</u>		<u>No. Sites</u>		<u>No. Individuals</u>		<u>Mean catch per site (no. individuals)</u>	
		Trawl	Seine	Trawl	Seine	Trawl	Seine	Trawl	Seine
Spring	All	51	N/A	20	N/A	52,874	N/A	2,644	N/A
Early summer	All	76	72	20	18	135,385	134,165	6,769	7,454
Late summer	All	77	75	20	18	92,214	100,970	4,611	5,609
Fall	All	59	N/A	20	N/A	21,238	N/A	1,062	N/A
All	Assawoman	59	67	3	3	81,439	50,041	27,146	16,680
All	Isle of Wight	80	71	4	4	84,936	72,491	21,234	18,123
All	Sinepuxent	60	59	3	3	9,236	45,175	3,079	15,058
All	Chincoteague	83	71	10	8	126,100	67,428	12,610	8,429

Figure 4: Blue crab abundance transformed ($\ln x + 1$) across years (a) and embayments (b) for trawl survey. Bars indicate standard error.

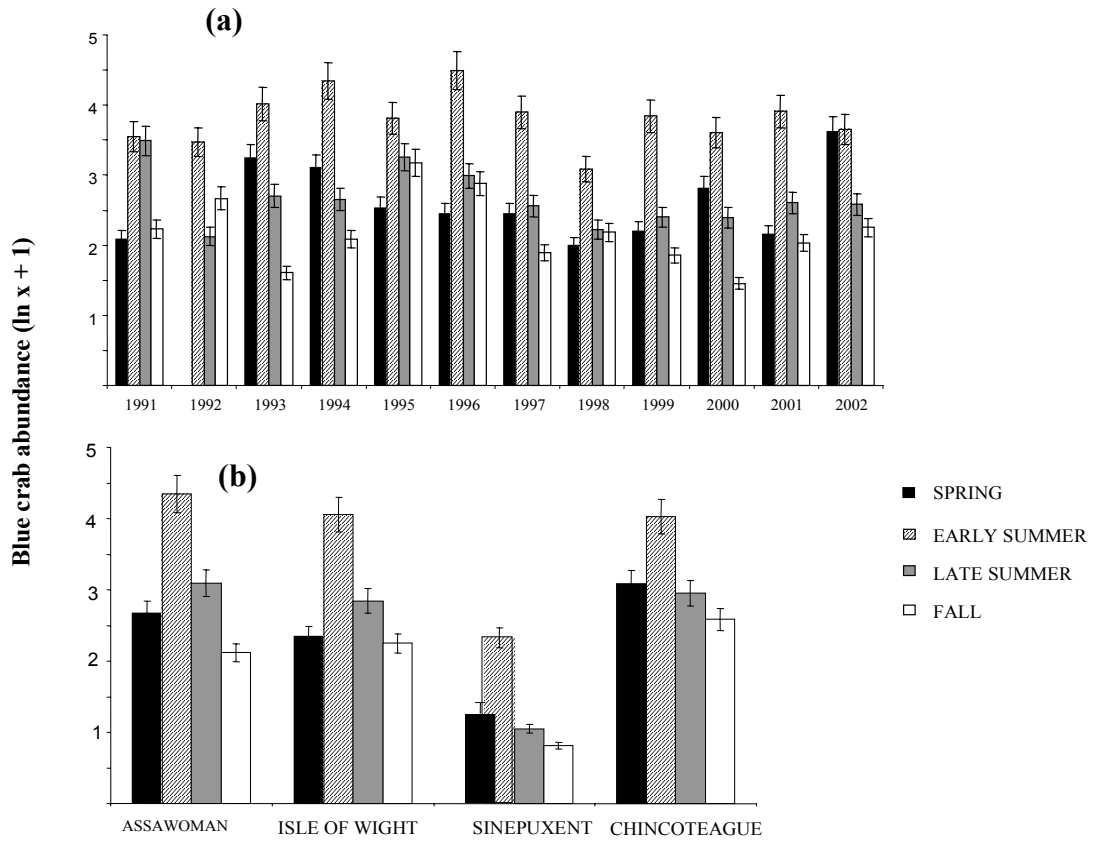
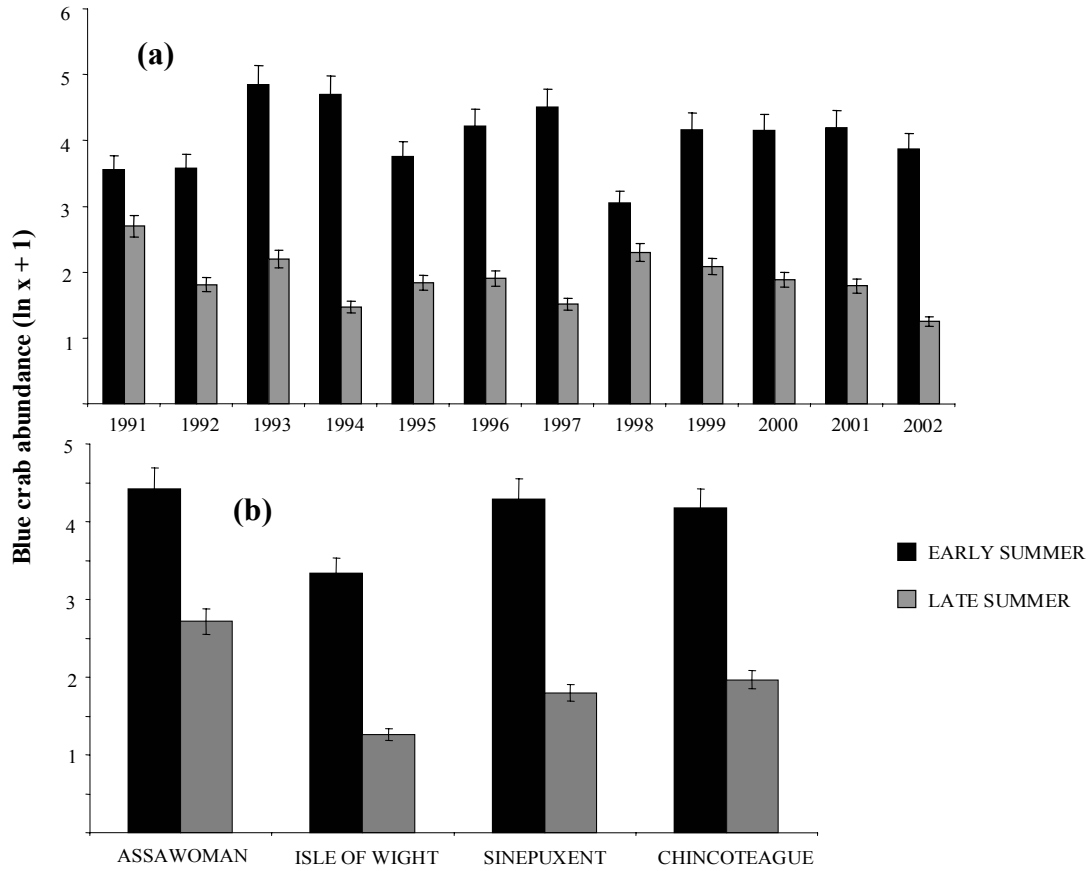


Figure 5: Blue crab abundance transformed ($\ln x + 1$) across years (a) and embayments (b) for seine survey. Bars indicate standard error.



was also different between bays ($p < 0.0001$, both surveys) with c. half the abundance observed (trawl) in Sinepuxent than that observed in other bays (Figure 4b).

A. mitchilli abundances showed moderate interannual variation ($CV_{\text{Trawl}} = 0.60$, $CV_{\text{Seine}} = 0.66$) with a low point across both gear types in 2001 (Figures 6 and 7).

Significant differences occurred between seasons ($p < 0.001$) and bays ($p < 0.0001$) for both gear types. Similar to *C. sapidus*, *A. mitchilli* mean abundance was c. 50% lower in Sinepuxent than other Bays for both gear types. *L. xanthurus*, a schooling demersal species, showed high interannual variation in abundance ($CV_{\text{Trawl}} = 1.68$, $CV_{\text{Seine}} = 1.49$; Figures 8 and 9). No significant differences occurred between bays ($p = 0.20$) for the seine gear, but a significant difference between bays was observed for trawl samples, where Sinepuxent exhibited lower abundance ($p < 0.0001$). Both gear types indicated significant differences in *L. xanthurus* abundances between seasons, with highest abundances during early summer ($p < 0.0001$).

C. regalis, a demersal species, was not a dominant species from the seine samples (rank in CPUE=35), but was abundant in the trawl survey, with significant seasonal and embayment differences ($p < 0.0001$ each) (Figure 10). Abundance tended to peak in summer months, and the northern bays of Assawoman and Isle of Wight had c. 2-fold higher abundances of weakfish than the Chincoteague or Sinepuxent. This species exhibited moderate interannual variation in abundance ($CV = 0.74$).

C. harengus was a dominant spring-time species in the trawl survey, but was very rare in the seine survey. The species only occurred in spring (Figure 11) and Sinepuxent Bay showed significantly lower abundance than other embayments ($p < 0.01$). *C.*

Figure 6: Bay anchovy abundance transformed ($\ln x + 1$) across years (a) and embayments (b) for trawl survey. Bars indicate standard error.

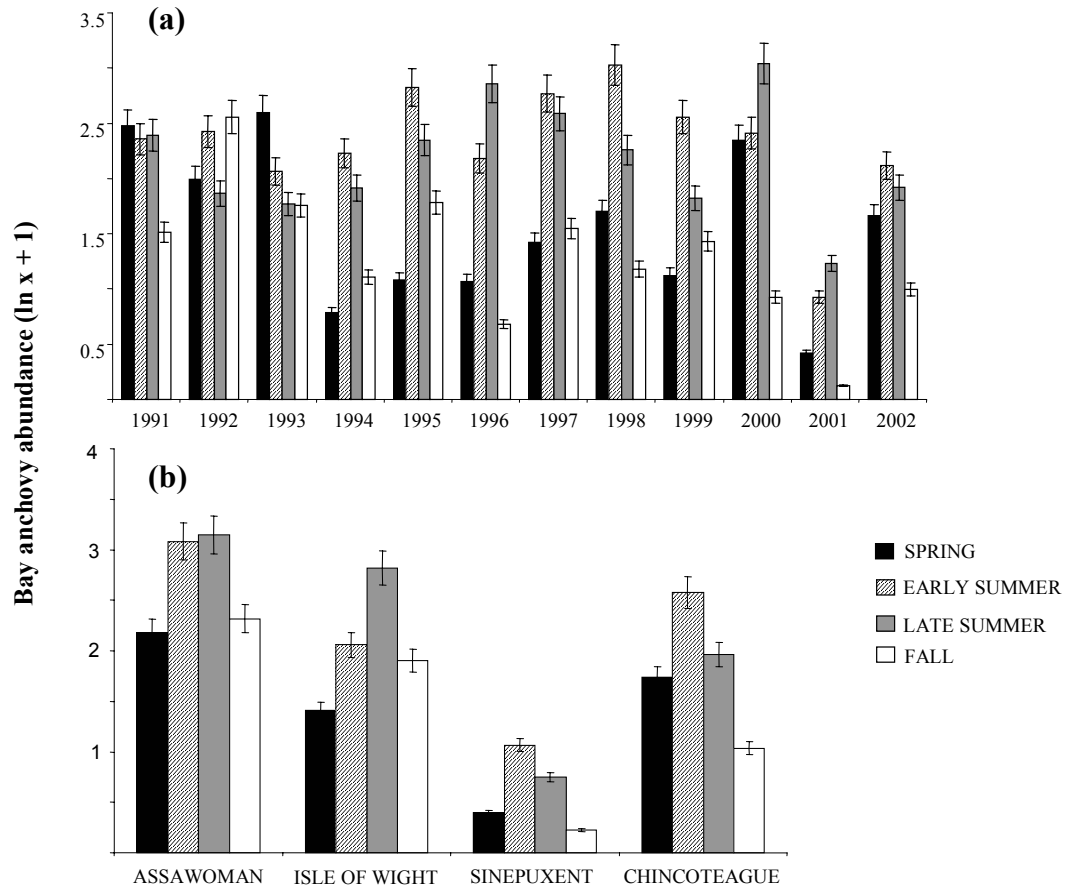


Figure 7: Bay anchovy abundance transformed ($\ln x + 1$) across years (a) and embayments (b) for seine survey. Bars indicate standard error.

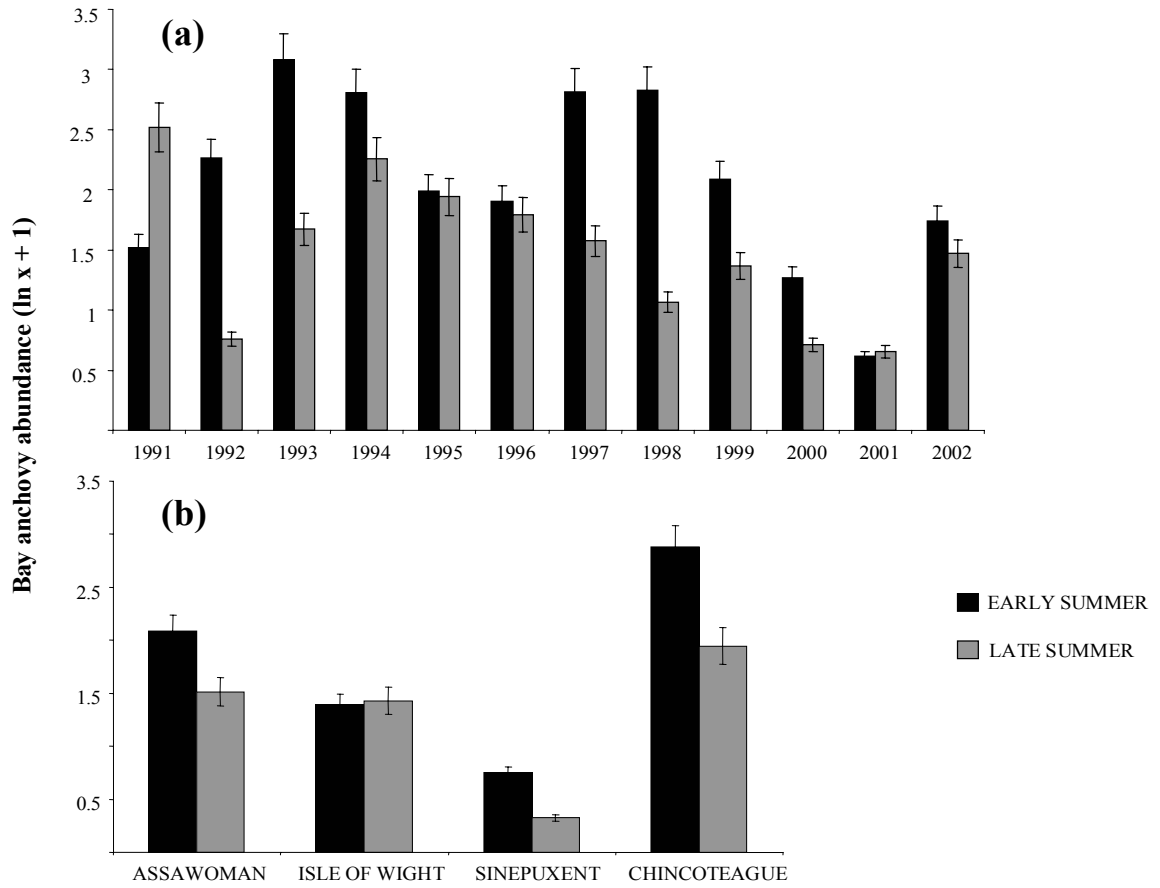


Figure 8: Spot abundance transformed ($\ln x + 1$) across years (a) and embayments (b) for trawl survey. Bars indicate standard error.

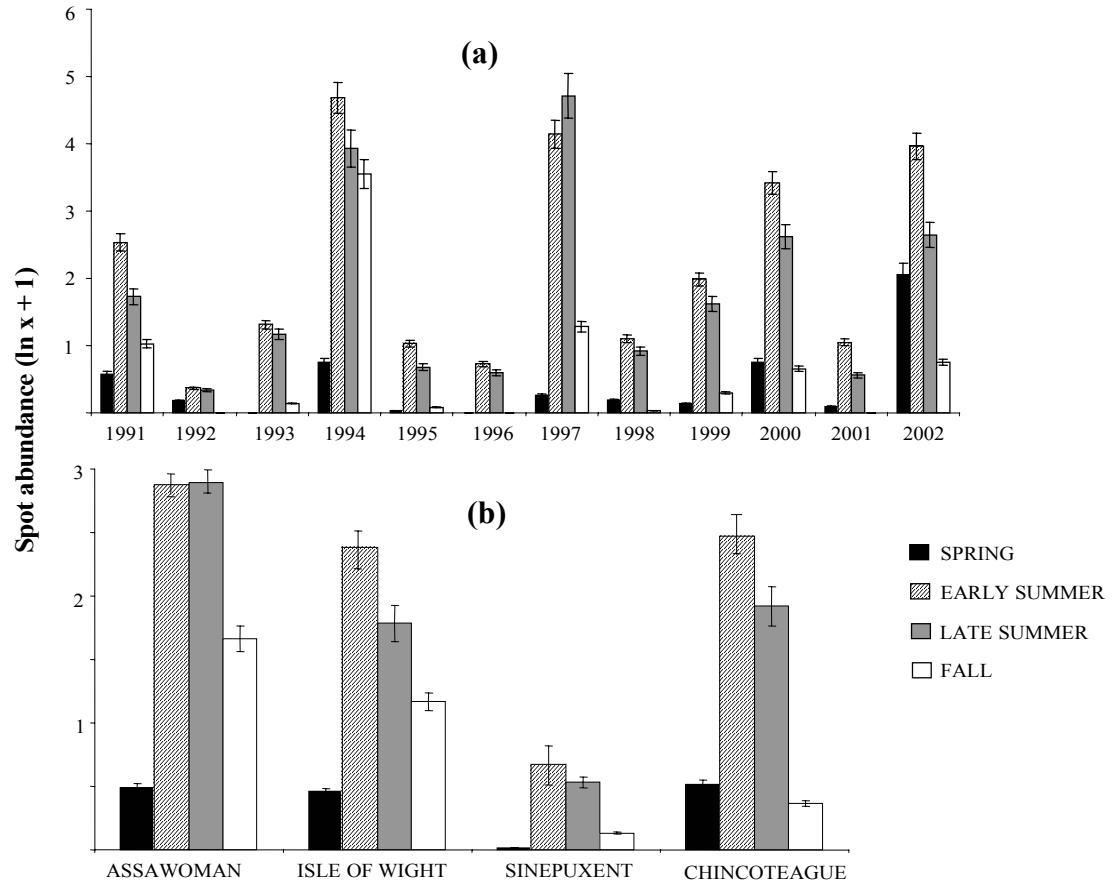


Figure 9: Spot abundance transformed ($\ln x + 1$) across years (a) and embayments (b) for seine survey. Bars indicate standard error.

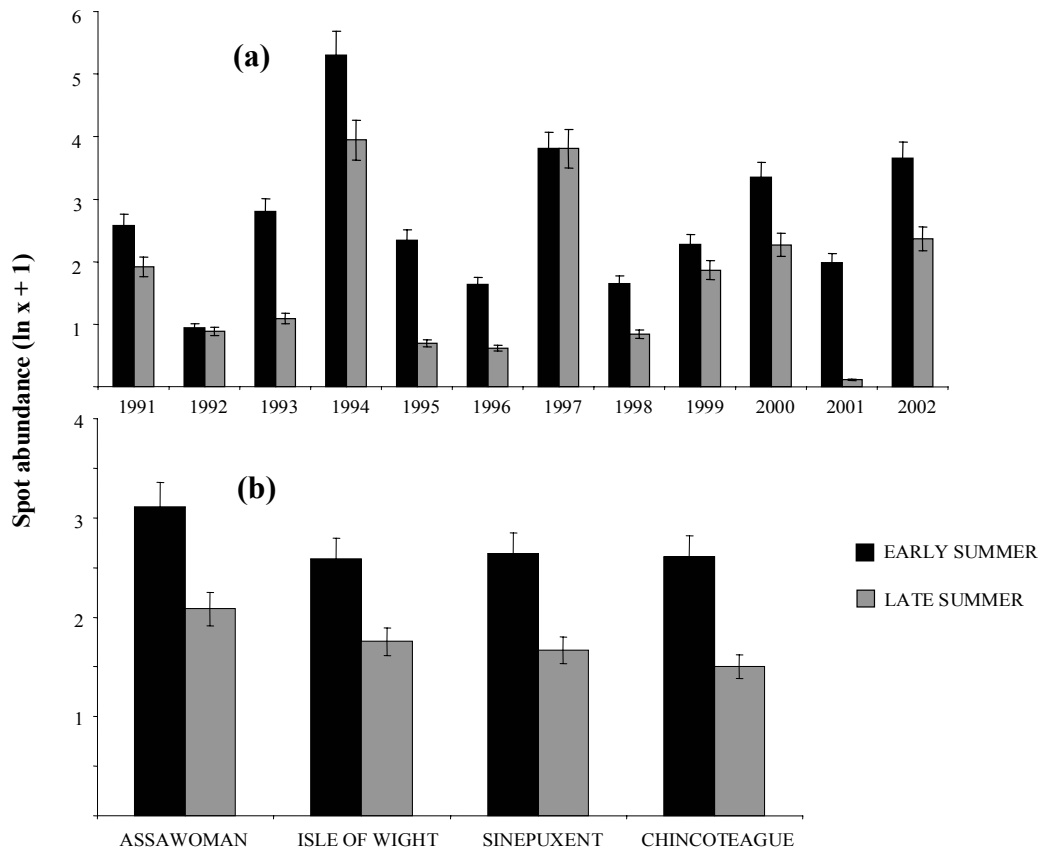


Figure 10: Weakfish seasonal abundance for trawl across years (a) and embayments (b). Bars represent standard error.

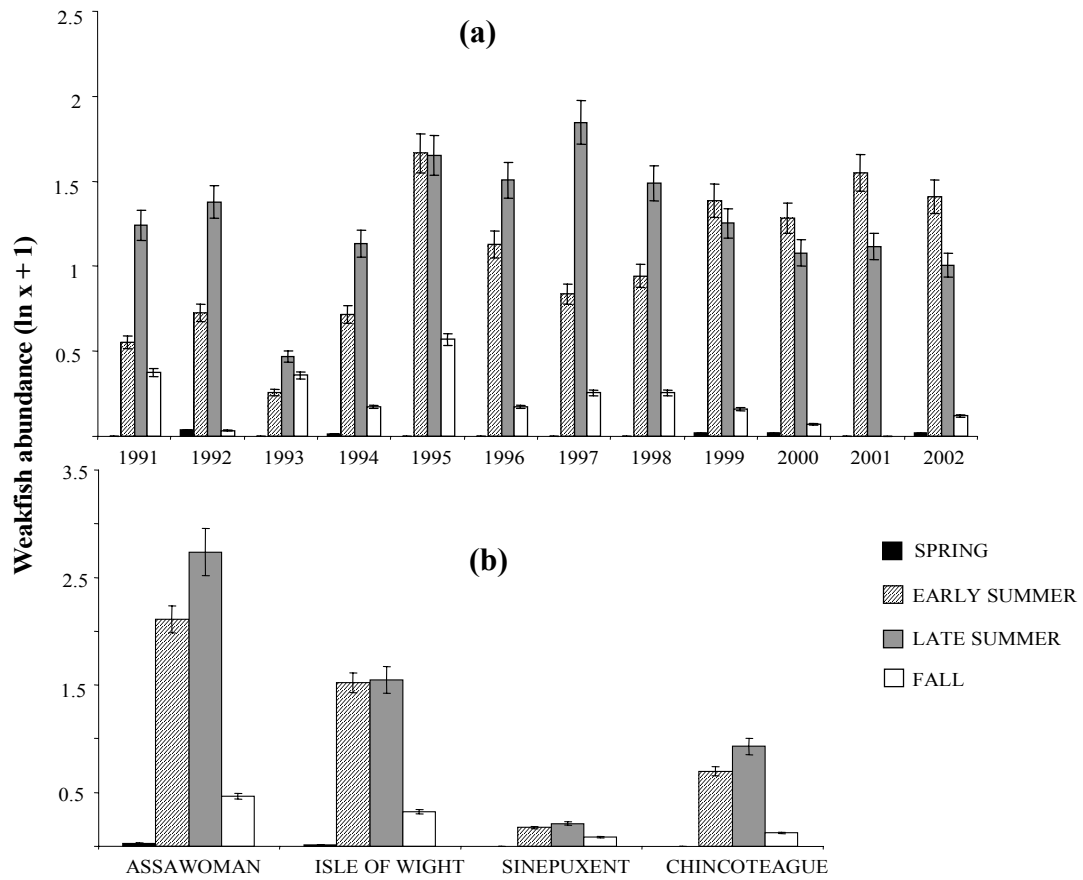
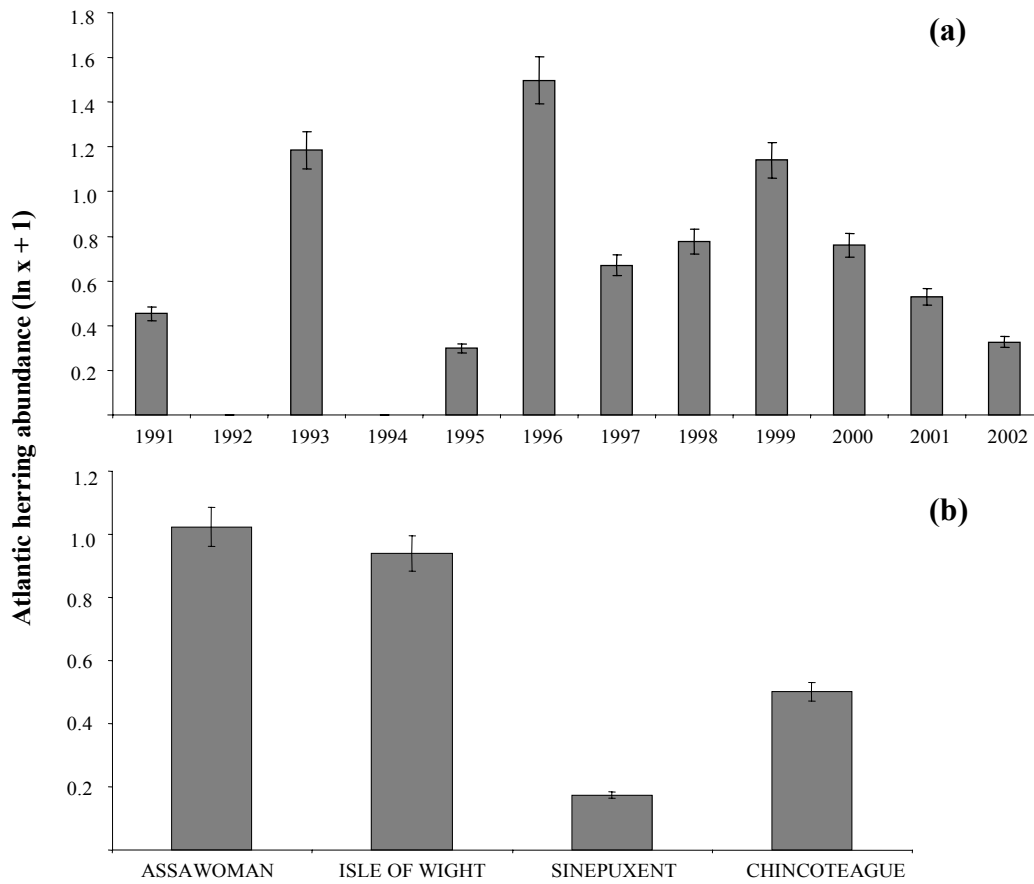


Figure 11: Atlantic herring abundance for trawl across years (a) and embayments (b). This species was only present in the spring (only season displayed), and very rare in the seine survey. Bars represent standard errors.



harengus showed fairly high interannual variation ($CV_{\text{Trawl}} = 0.99$) and was completely absent from the survey in 1992 and 1994.

M. undulatus was a dominant species in the trawl survey (rank = 6), but occurred less frequently in the seine survey (rank=31). Its distribution was significantly different between bays and seasons (Kruskal-Wallis non-parametric test, $p < 0.0001$ for each factor). Overall, *M. undulatus* abundance was generally highest in the fall, but this pattern was variable between years (Figure 12; CV among years was 0.72). Abundances in the northern two bays were 2-fold than those in the southern two bays. *P. dentatus* also showed dominance in the trawl data (rank=7) relative to the seine data (rank=17), with highest abundances occurring early summer (seasonal effect; K-W, $p < 0.0001$) (Figure 13). Its distribution varied significantly between bays (K-W, $p < 0.0001$), with generally lower abundances in the Sinepuxent than elsewhere. The species exhibited relatively low intrannual variation across the bays (CV= 0.44).

B. chrysoura was a dominant species collected by both gear types and differed in abundance significantly by season (K-W, $p < 0.0001$ for seine and trawl) with a peak abundance in late summer (Figures 14 and 15). There was a significant difference in trawl abundances by bay (K-W, $p < 0.0001$), but no significant differences occurred for the seine survey. Similar to other dominant species, trawl CPUE's were >50% less in Sinepuxent than the other bays. This species exhibited moderate interannual variation ($CV_{\text{Trawl}} = 0.61$, $CV_{\text{Seine}} = 0.84$). *B. tyrannus* was important in both surveys, but was of particularly high abundance (based on CPUE-Table 5) in the seine survey. *B. tyrannus* abundance (Figure 16 and 17) tended to peak early summer (seine and trawl). CPUE was significantly influenced by embayment in both trawl and seine survey (K-W, $p < 0.001$, p

Figure 12: Croaker seasonal abundance for trawl by years (a) and embayments (b). Croaker was infrequently captured in the seine survey. Bars represent standard errors.

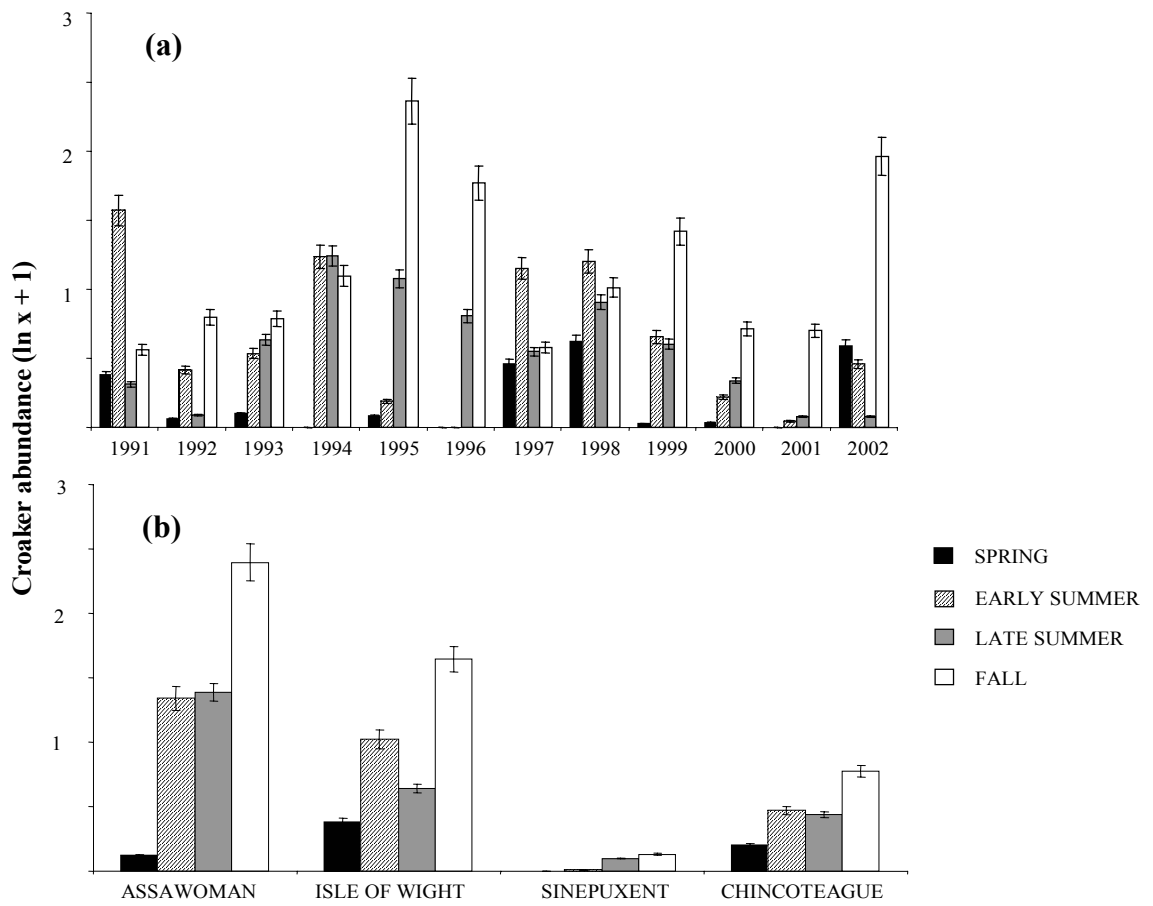


Figure 13: Summer flounder seasonal abundance for trawl by embayment and across years (a) and embayments (b). Bars represent standard error.

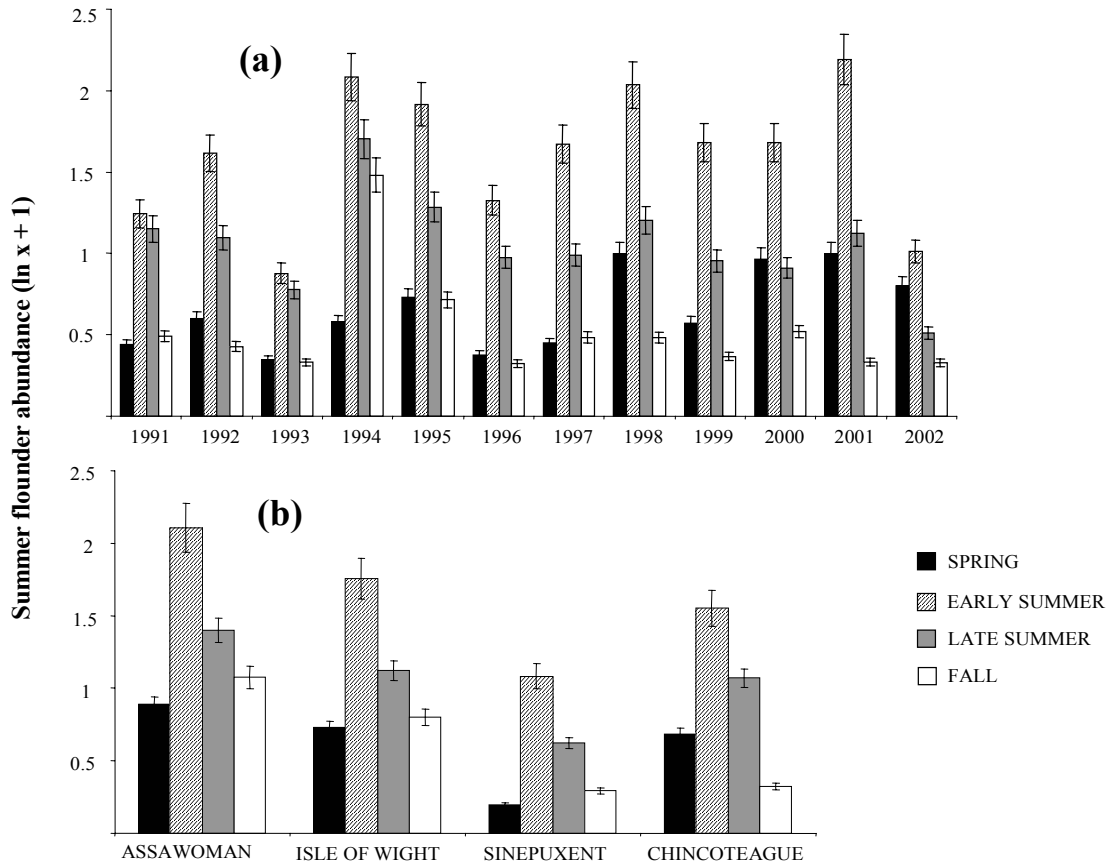


Figure 14: Silver perch seasonal abundance for trawl across years (a) and embayments (b). Bars represent standard error.

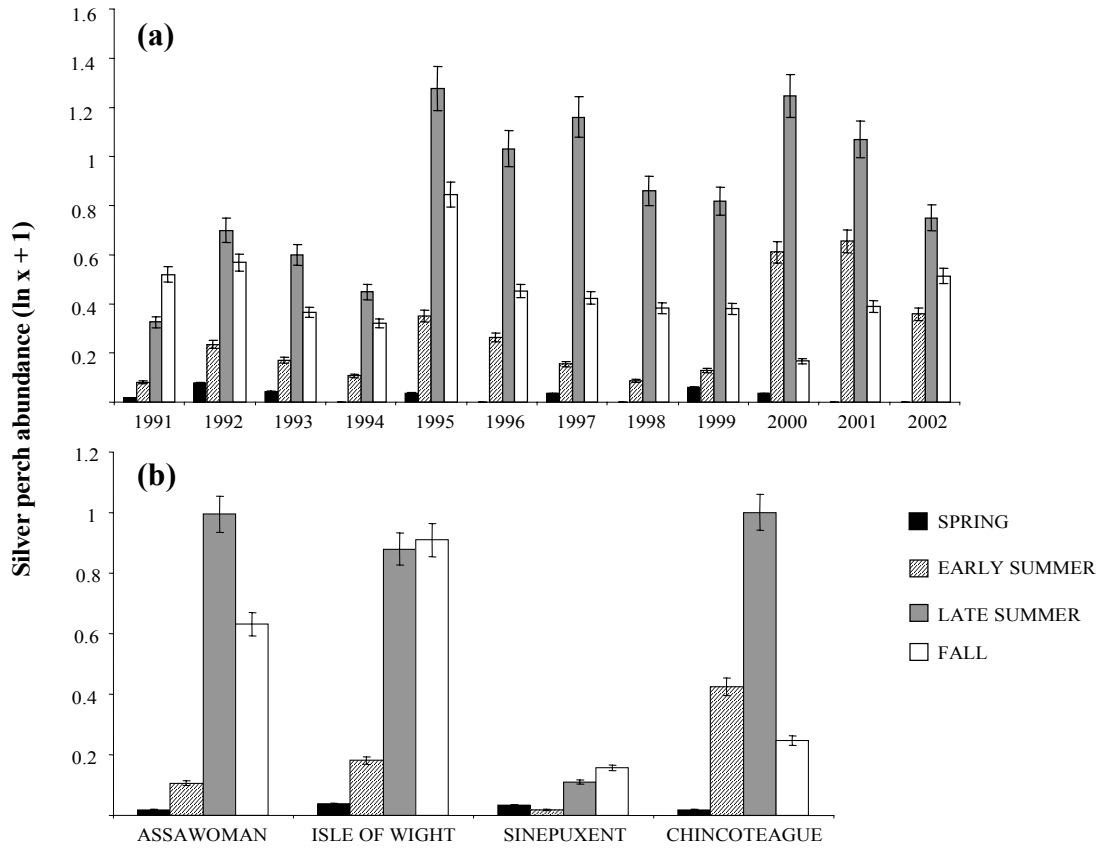


Figure 15: Silver perch seasonal abundance for seine across years (a) and embayments (b). Bars represent standard error.

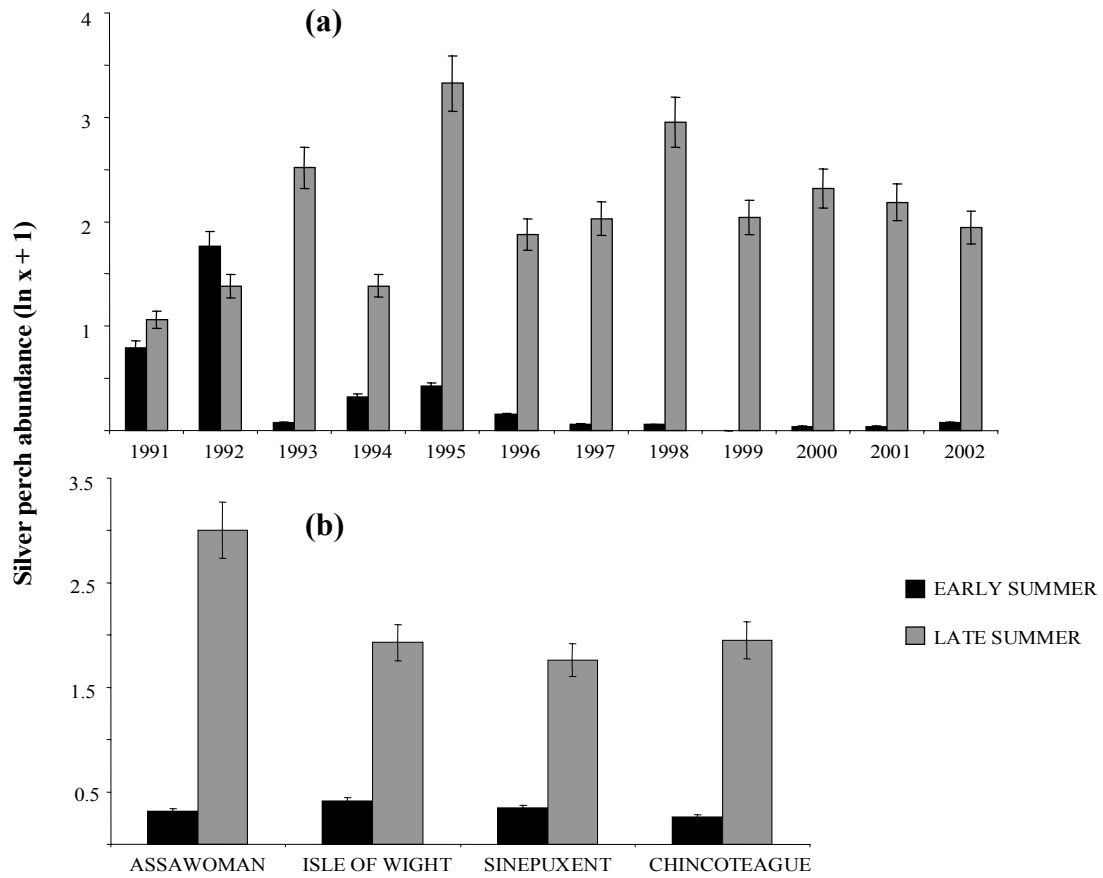


Figure 16: Atlantic menhaden seasonal abundance for seine across years (a) and embayments (b). Bars represent standard error.

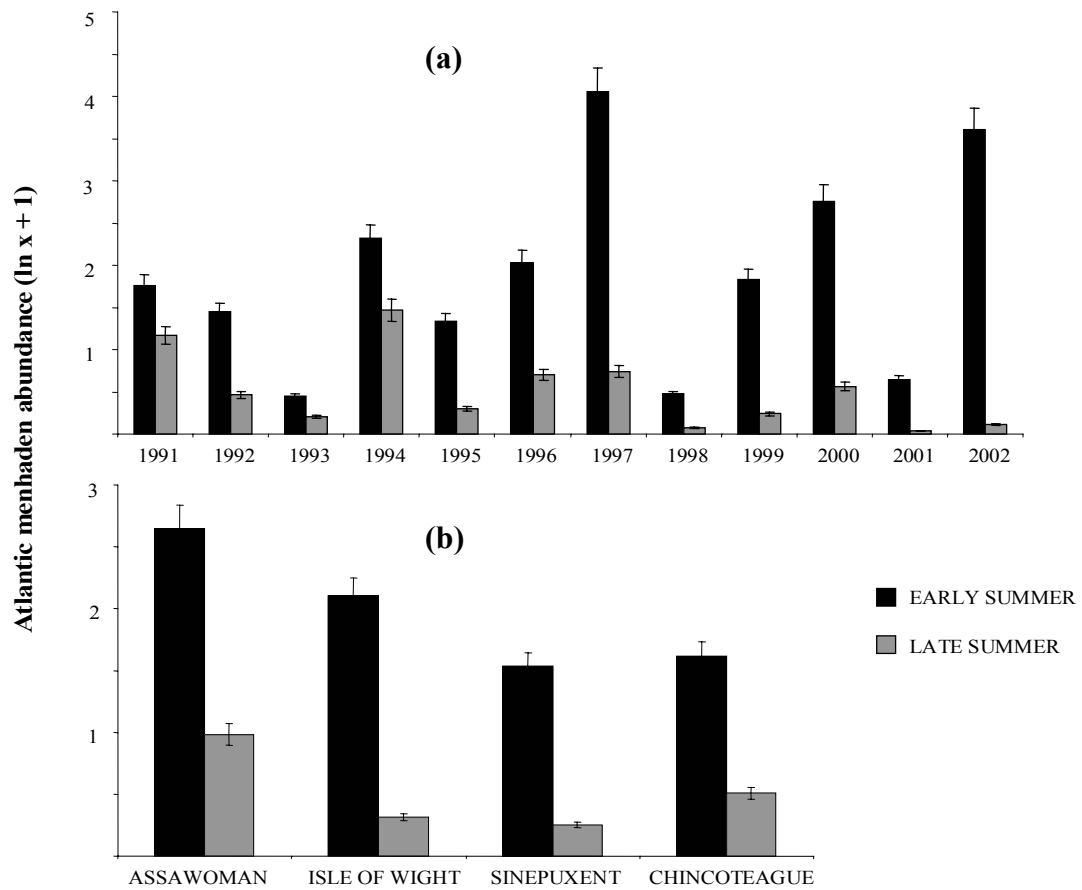
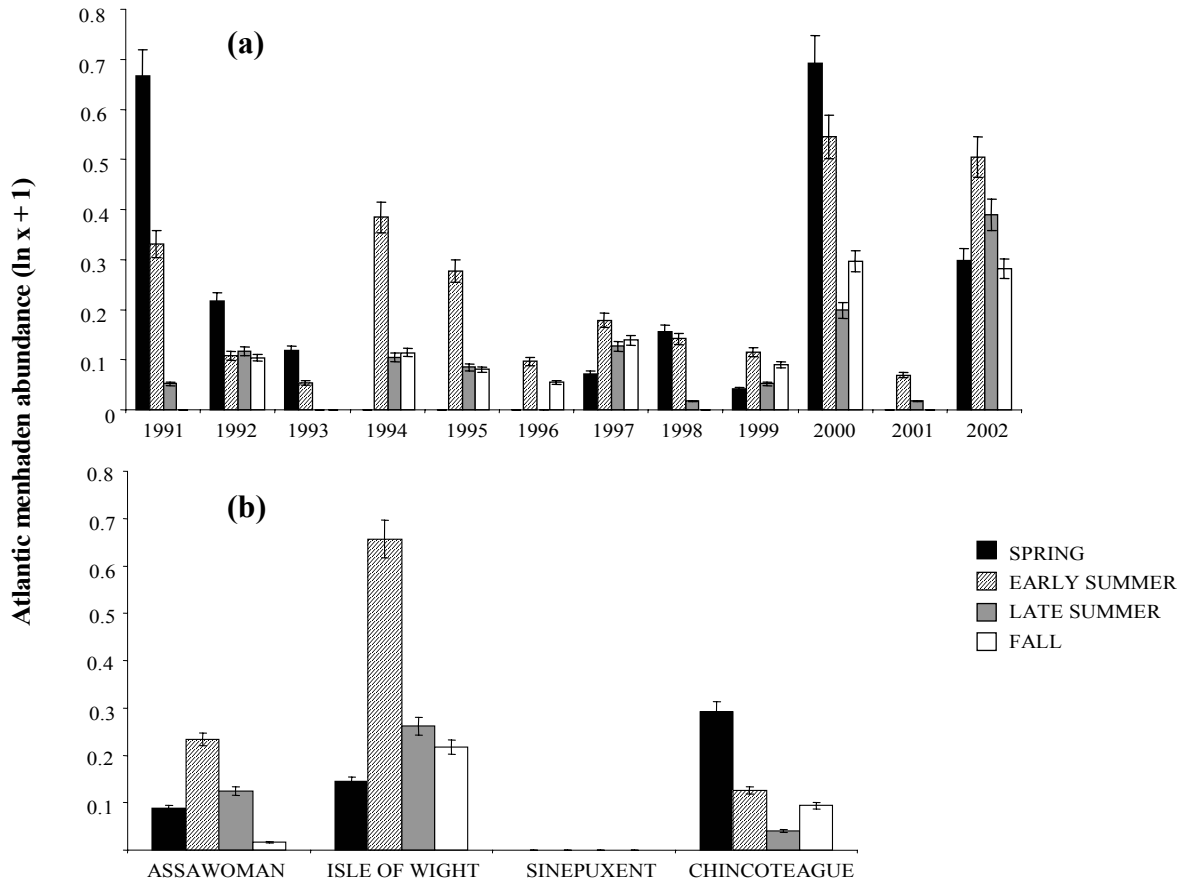


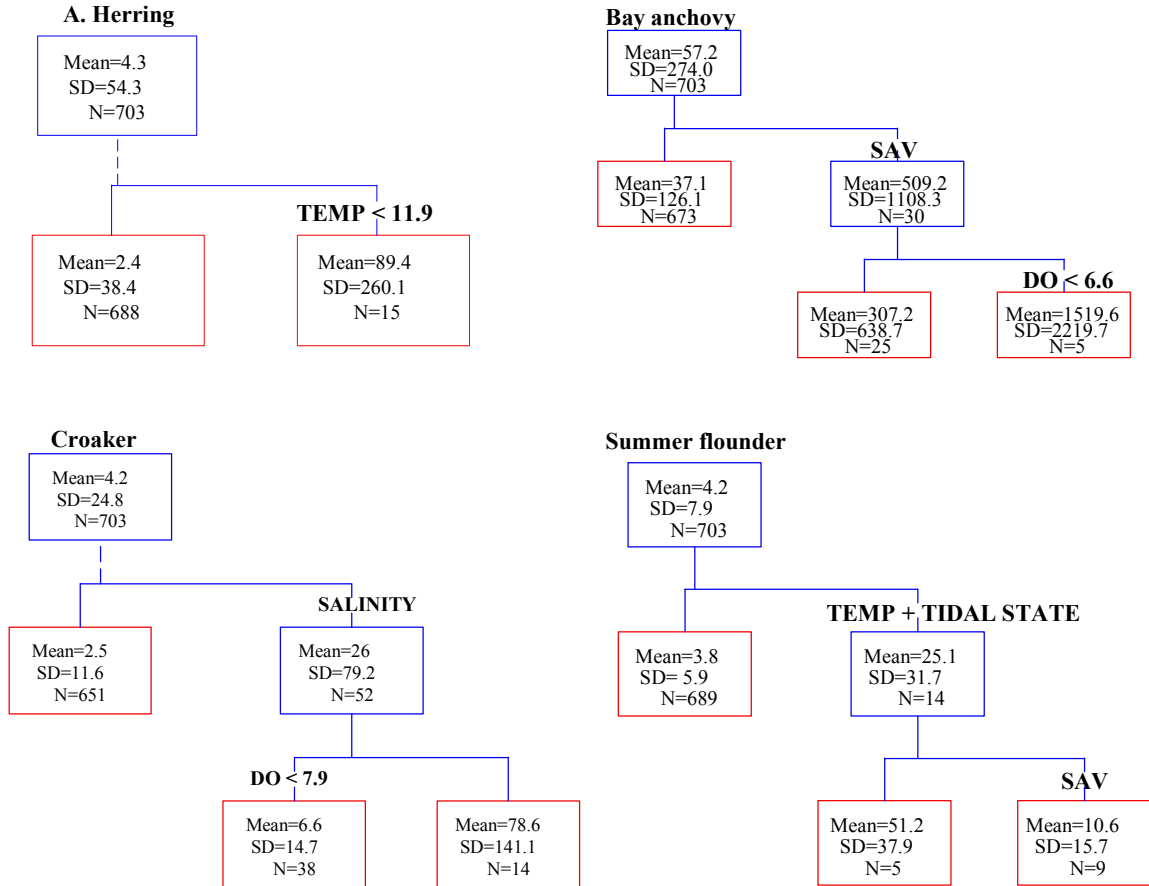
Figure 17: Atlantic menhaden seasonal abundance for trawl across years (a) and embayments (b). Bars represent standard error.



<0.01, respectively) and by season in trawl (K-W, $p < 0.001$) and seine (Mann-Whitney U-test, $p < 0.0001$). No menhaden were observed for the Sinepuxent trawl sample, and the Isle of Wight showed significantly higher abundances than elsewhere. Seine data indicated that menhaden tended to occur at higher abundance in the northern two bays. Interannual variations for menhaden were relatively high ($CV_{\text{Trawl}} = 1.19$, $CV_{\text{Seine}} = 1.14$).

CART analysis provided some insight into possible explanatory variables affecting abundances, but did so for only a minority of the dominant species. For the trawl data (focusing on the top species = 95% of catch), there were no significant explanatory variables for *C. sapidus*, *L. xanthurus*, *C. regalis*, *B. chrysoura*, or *B. tyrannus*. *A. mitchilli* abundance was partially explained by SAV coverage and dissolved oxygen levels (Figure 18). At those sites containing SAV, mean abundance was >10-fold higher. D.O. levels less than 6.6 mg l^{-1} were associated with c. 3-fold higher abundances of *A. mitchilli*. On the other hand, *C. harengus* abundance was related to temperature, which can be seen as a proxy for season because *C. harengus* was absent during the warmer seasons of early and late summer and fall. *M. undulatus* abundance appeared to be driven by an interaction of salinity and year and dissolved oxygen. Although there appeared to be a discrete seasonal component to croaker abundance (Fig. 12), the CART procedure did not indicate season as a significant explanatory variable. Lastly, *P. dentatus* abundance was best explained by an interaction between SAV coverage and tidal state. Like *M. undulatus*, summer flounder showed seasonal variability (Fig. 13) that was not detected by the CART procedure.

Figure 18: Classification and regression tree (CART) diagrams for bay anchovy, Atlantic herring, croaker, and summer flounder in the trawl survey, 1991-2002. For each box: Mean = mean abundance for parameters at that node; SD = standard deviation; n = number of samples meeting the criteria of listed parameter values. Caution should be used when interpreting these trees as some of the interactive parameters are not scaled in a way that permits classification of habitat variables and therefore the explanatory variable is directionless.



Assemblages and Embayments

The MDS plots for trawl data indicated fairly tight ordination of specific embayments (Figure 19a) based on relative abundances of all species encountered and grouped by site. The stress statistics were all low, signifying that the two-dimensional plots were representative of the relationship between sites. Within the trawl data (Fig. 19a), sites 20 and 12 were discrete from other Chincoteague sites, while site 7 (Isle of Wight) was separate from the other sites of that embayment. The northern bays of Assawoman and Isle of Wight ordinated consistently along axis 1 and broadly overlapped whereas Sinepuxent ordinated along both axes, remaining separate from the other embayments. Chincoteague sites generally clustered together with dispersion along both axes. Within the seine data, sites 18 and 14 showed separation from other Chincoteague sites (Fig. 19b), and site 4 was discrete from other Isle of Wight sites, while all the Sinepuxent sites were fairly equidistant. Overall, the trawl MDS plots showed better segregation of embayments than the seine plots, as indicated by the lower stress value. When data for each embayment were compared on a seasonal basis, separations of sites according to bays remained apparent (Figure 20). The ordination plots showed varying degrees of overlap between bays on a seasonal basis. The fall season showed the greatest separation between bays whereas there was greater overlap in the spring. Both early and late summer seasons had similar patterns with regard to separation of embayments, with sites 7, 11, 12, and 20 consistently ordinating outside the clusters of their defined system.

The cluster analysis (Bray-Curtis similarity) in general showed similarity of sites within embayments (Figure 21). For the trawl dendrogram, all northern bays sites except site 7 split at the node indicating ~ 30% dissimilarity. This same node included site 12

Figure 19: Non-metric multidimensional scaling (MDS) plots of trawl (a) and seine (b) data by site (numbered), 1991-2002. Site identification numbers are referenced in Fig. 1. Squares = Assawoman, Ovals = Isle of Wight, Diamonds = Sinepuxent, Triangles = Chincoteague. Stress values for each plot are indicated.

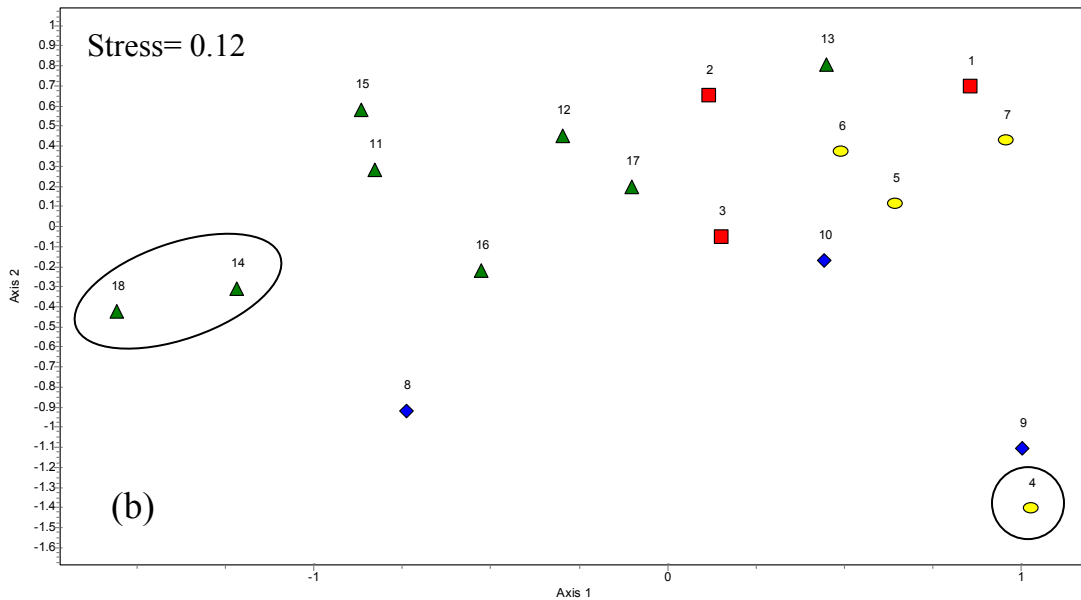
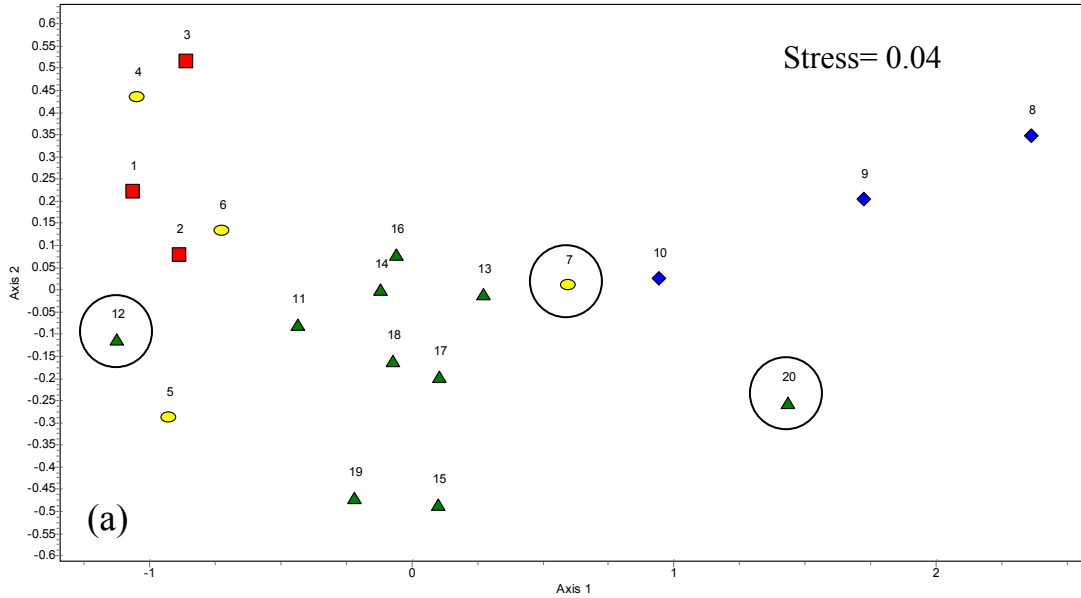


Figure 20: Non-metric multidimensional scaling plots of trawl data (1991-2002) by season and site (numbered). Squares = Assawoman, Ovals = Isle of Wight, Diamonds = Sinepuxent, Triangles = Chincoteague. Stress values for each plot are indicated.

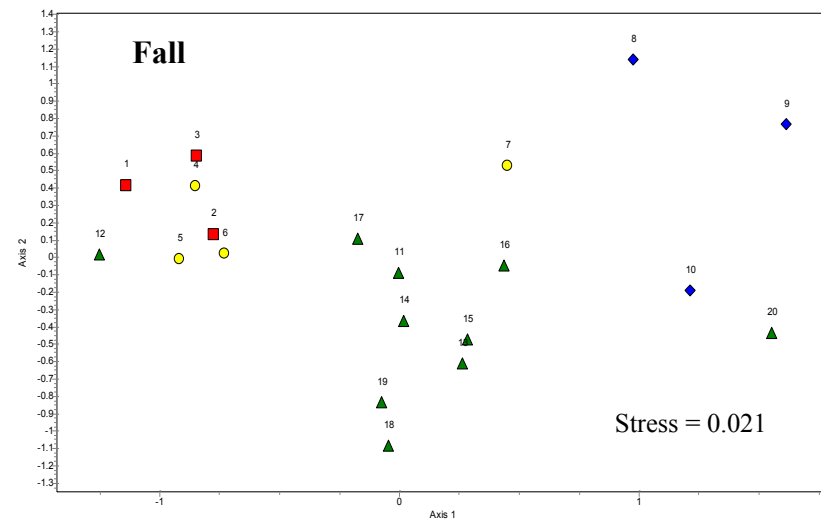
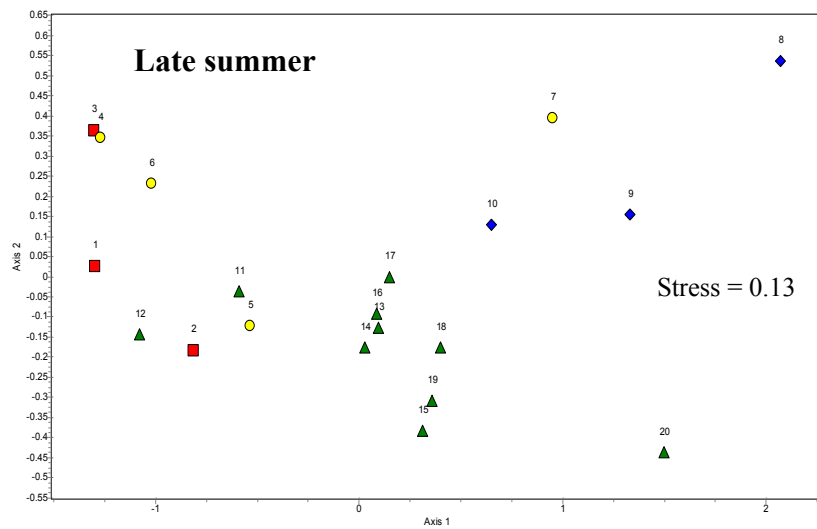
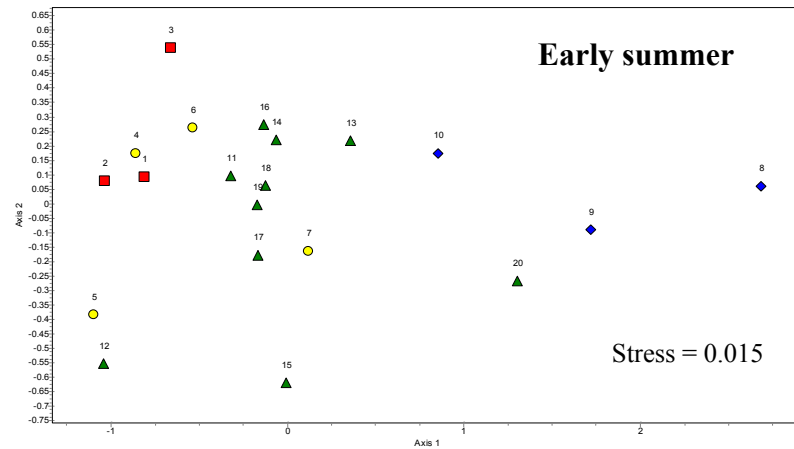
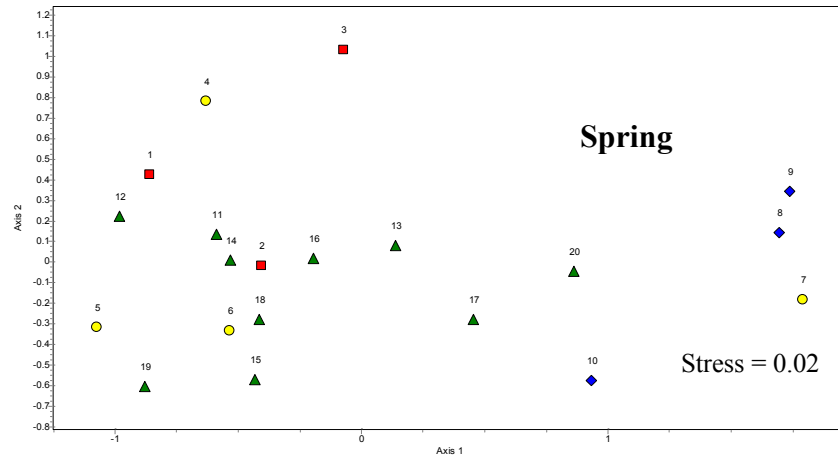
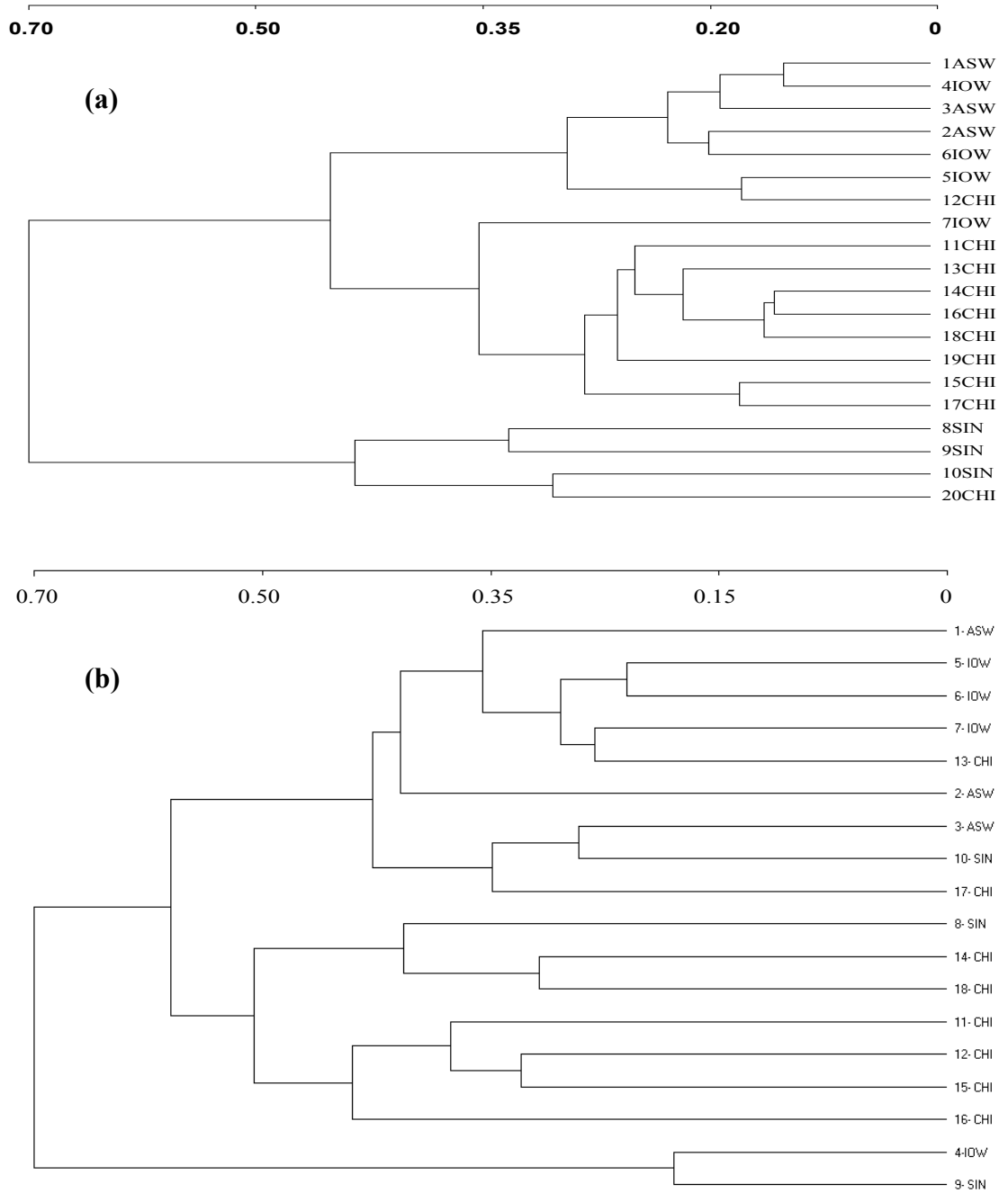


Fig. 21: Cluster dendrogram for all sites from trawl (a) and seine (b) data, 1991-2002 using agglomerative clustering and the Bray-Curtis similarity coefficient as distance measure. The upper line represents the dissimilarity (percent) along the dendrogram (e.g. 0 = no dissimilarity). Numbers are sites (referenced in Fig. 1), with the following embayment codes: ASW = Assawoman, IOW = Isle of Wight, SIN = Sinepuxent, CHI = Chincoteague.



from Chincoteague Bay. The Sinepuxent sites all separated at the node associated with ~ 47% dissimilarity. The seine dendrogram followed the MDS plots in that the nodes for each of the site cluster showed a lesser degree of dissimilarity. The northern bay sites (except sites 3 and 4) all split from a node at ~ 41% dissimilarity. Outlier sites included trawl sites 7 (IOW), 10 (SIN), 12, and 20 (CHI) and seine sites 1 (ASW), 4 (IOW), 8, 9 (SIN), and 13 (CHI) (for bay abbreviations, see Figure 20).

The separation of sites by embayment was supported by the ANOSIM analysis (trawl: $R = 0.45$, $p < 0.004$; seine: $R = 0.28$, $p < 0.03$) (Table 7). For the trawl, pairwise comparisons (Bonferroni adjusted significance level to account for multiple [6] comparisons) were significant between Assawoman and Chincoteague ($R = 0.42$, $p < 0.04$), and between Sinepuxent and each of the other bays indicating Sinepuxent is distinct in terms of its profundal community. The seine survey ANOSIM analysis also indicated a significant difference between bays ($R = 0.28$, $p < 0.04$), although the only pairwise differences were between Chincoteague and Isle of Wight and Chincoteague and Sinepuxent. MDS plots of trawl data by season showed moderate separation between early and summer samples, but broad overlap across other seasons (Figure 22). The ANOSIM procedure confirmed this pattern in that there was a significant difference between seasons, but a low R value ($R = 0.30$; $p < 0.001$) indicated high overlap (Table 8).

The SIMPER analysis of trawl data showed that each bay had distinctive species in terms of their relative abundances. For example, the higher relative abundances of *A. mitchilli*, *L. xanthurus*, and *C. sapidus* distinguished the fauna of Assawoman Bay (Table 9) from other bays. In general the relative abundance of these three species, in addition to *C. regalis* and *P. dentatus* were diagnostic among the bays. Note that these dominant

Table 7: Analysis of Similarity by embayment for trawl and seine survey, 1991-2002. Pariwise tests between bays are protected at $\alpha = 0.005$ level. See text for discussion of interpretations for R values. The number in parentheses by each embayment indicates number of sites used in analysis.

ANOSIM By Bay			
<i>TRAWL</i>			
Sample R: 0.45			
p < 0.004			
<u>1st Group</u>	<u>2nd Group</u>	<u>p</u>	<u>Sample R</u>
Assawoman (3)	Chincoteague(10)	0.04	0.42
Assawoman (3)	Isle of Wight (4)	0.85	-0.19
Assawoman (3)	Sinepuxent (3)	0.05	1
Chincoteague (10)	Isle of Wight (4)	0.08	0.25
Chincoteague (10)	Sinepuxent (3)	0.01	0.7
Isle of Wight (4)	Sinepuxent (3)	0.05	0.78
<i>SEINE</i>			
Sample R: 0.28			
p < 0.04			
<u>1st Group</u>	<u>2nd Group</u>	<u>p</u>	<u>Sample R</u>
Assawoman (3)	Chincoteague(8)	0.14	0.18
Assawoman (3)	Isle of Wight (4)	0.22	0.09
Assawoman (3)	Sinepuxent (3)	0.25	0.11
Chincoteague (8)	Isle of Wight (4)	0.008	0.49
Chincoteague (8)	Sinepuxent (3)	0.02	0.46
Isle of Wight (4)	Sinepuxent (3)	0.31	0.07

Figure 22: Non-metric multidimensional scaling plot of trawl data by season. Spring = diamond, early summer = square, late summer = oval, fall = triangle. Stress values for each plot indicate adequacy of two-dimensional relationships, where lower values (< 0.2) indicates true relationships of the data points.

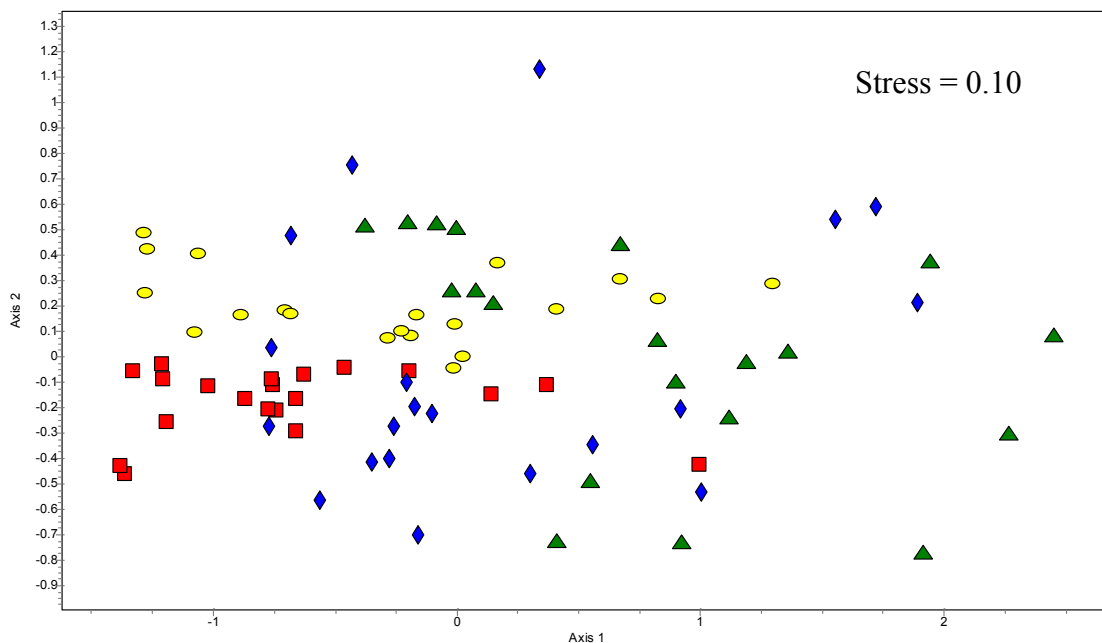


Table 8: Analysis of Similarity on trawl data (1991-2002) for seasonal comparisons. Pairwise tests are protected at $\alpha = 0.005$ level. Numbers in parentheses indicate number of sites used for each seasonal comparison.

ANOSIM by Season			
Sample R: 0.30			
p < 0.001			
<u>1st Group</u>	<u>2nd Group</u>	<u>p</u>	<u>Sample R</u>
Early Summer (20)	Fall (20)	0.001	0.53
Early Summer (20)	Late Summer (20)	0.001	0.22
Early Summer (20)	Spring (20)	0.001	0.31
Fall (20)	Late Summer (20)	0.001	0.24
Fall (20)	Spring (20)	0.001	0.25
Late Summer (20)	Spring (20)	0.001	0.26

Table 9: Results from Similarity Percentage (SIMPER) analysis on trawl data, 1991-2002. Diagnostic species are listed in descending order of percent contribution to overall bay similarity. Those species that contribute to the top 90% ranked species are shown. Mean abundance is mean number of individuals per embayment. For common names, see Table 5.

<u>Typifying Species</u>	<u>Mean abundance</u>	<u>% contribution</u>	<u>Cumulative %</u>
<u>Assawoman</u>			
<i>A. mitchilli</i>	7930.3	30.2	30.2
<i>L. xanthurus</i>	7038.0	27.2	57.4
<i>C. sapidus</i>	4739.7	19.3	76.7
<i>C. regalis</i>	3690.0	12.2	88.9
<i>M. undulatus</i>	1164.0	3.6	92.5
<u>Isle of Wight</u>			
<i>L. xanthurus</i>	5205.8	30.0	30.0
<i>C. sapidus</i>	5201.0	26.4	56.4
<i>A. mitchilli</i>	5892.8	24.6	81.0
<i>C. regalis</i>	1657.7	7.4	88.4
<i>P. dentatus</i>	395.0	2.9	91.3
<u>Sinepuxent</u>			
<i>C. sapidus</i>	944.3	31.9	31.9
<i>A. mitchilli</i>	937.7	23.1	55.0
<i>L. xanthurus</i>	339.7	9.5	64.5
<i>P. dentatus</i>	129.0	7.3	71.7
<i>S. maculatus</i>	77.4	4.2	75.9
<i>P. carolinus</i>	75.6	4.1	80.0
<i>S. foetens</i>	73.3	3.7	83.7
<i>E. microstomus</i>	83.0	3.0	86.7
<i>S. aquosus</i>	55.0	2.1	88.8
<i>C. striata</i>	46.7	2.0	90.8
<u>Chincoteague</u>			
<i>C. sapidus</i>	4849.3	43.6	43.6
<i>L. xanthurus</i>	3030.6	24.7	68.3
<i>A. mitchilli</i>	2677.5	20.7	89.0
<i>P. dentatus</i>	322.2	2.7	91.7

species were consistently higher in relative abundance in the northern two bays than in the southern two bays. In this analysis, Sinepuxent required the greatest number of species (10) to meet the 90% similarity measure.

Analysis of the seine survey data by the SIMPER procedure showed that six species (*B. tyrannus*, *C. sapidus*, *M. menidia*, *A. mitchilli*, *B. chrysoura*, *L. xanthurus*) were diagnostic for each embayment (Table 10). Two additional species (*Fundulus heteroclitus* and *F. majalis*) were necessary to statistically distinguish Sinepuxent. *M. menidia* (the most abundant species collected in the seine survey) abundance was greatest in Sinepuxent (CPUE = 352) and lowest in Chincoteague (CPUE = 53.7). *B. tyrannus*, the second most abundant species in the seine survey, was most abundant in Assawoman (CPUE = 81.9) compared to Sinepuxent (CPUE = 21.7) and Chincoteague (CPUE = 26.4). *C. sapidus* abundance was highest in Assawoman (CPUE = 101.2) and Sinepuxent (CPUE = 69.1) and lowest in Chincoteague (CPUE = 66.6) and Isle of Wight (CPUE = 40.6).

Diversity

Species richness varied between bays and seasons, and across years (Figs.23 and 24) for both gear types. For the trawl data, there was a significant difference between bays (ANOVA, $F = 58.5$, $p < 0.0001$) and season ($F = 153.3$, $p < 0.0001$). The northern bays of Assawoman (mean = 7.6 and 13.5 for trawl and seine, respectively) and Isle of Wight (mean = 7.3 and 11.6 for trawl and seine, respectively) were higher in species richness compared to those of Sinepuxent (mean = 4.8 and 10.1 for trawl and seine, respectively) and Chincoteague (mean = 6.2 and 10.6 for trawl and seine, respectively)

Table 10: Results from Similarity Percentage (SIMPER) analysis on seine data, 1991-2002. Diagnostic species are listed in descending order of percent contribution to overall bay similarity. Those species that contribute to the top 90% ranked species are shown. Mean abundance is mean number of individuals per embayment. For common names, see Table 5.

<i>Typifying Species</i>	Mean abundance	% contribution	Cumulative %
<u>Assawoman</u>			
<i>B. tyrannus</i>	3946.6	30.7	30.7
<i>C. sapidus</i>	2443.3	18.6	49.3
<i>M. menidia</i>	2525.0	15.4	64.8
<i>L. xanthurus</i>	1843.0	14.9	79.7
<i>B. chrysoura</i>	1966.6	9.4	89.0
<i>A. mitchilli</i>	1055.3	2.6	91.6
<u>Isle of Wight</u>			
<i>M. menidia</i>	6248.7	31.6	31.6
<i>B. tyrannus</i>	5249.7	31.5	63.1
<i>L. xanthurus</i>	1017.5	9.6	72.7
<i>M. curema</i>	1084.7	7.3	80.0
<i>C. sapidus</i>	1004.5	5.2	85.2
<i>B. chrysoura</i>	617.5	4.0	89.2
<i>A. mitchilli</i>	1165.2	2.6	91.8
<u>Sinepuxent</u>			
<i>M. menidia</i>	8331.0	42.0	42.0
<i>L. xanthurus</i>	1506.0	24.7	66.7
<i>C. sapidus</i>	1636.3	17.9	84.6
<i>B. tyrannus</i>	917.4	2.5	87.1
<i>F. heteroclitus</i>	421.4	2.0	89.1
<i>F. majales</i>	299.3	2.0	91.1
<u>Chincoteague</u>			
<i>C. sapidus</i>	1605.9	25.6	25.6
<i>A. mitchilli</i>	1572.1	20.2	45.8
<i>M. menidia</i>	1295.6	19.5	65.2
<i>L. xanthurus</i>	668.3	12.9	78.2
<i>B. tyrannus</i>	1796.5	8.8	86.9
<i>B. chrysoura</i>	636.4	4.1	91.1

Figure 23: Mean number of species per haul for 1991-2002 MD DNR coastal finfish trawl survey by season (a) and embayment (b). Bars represent standard error.

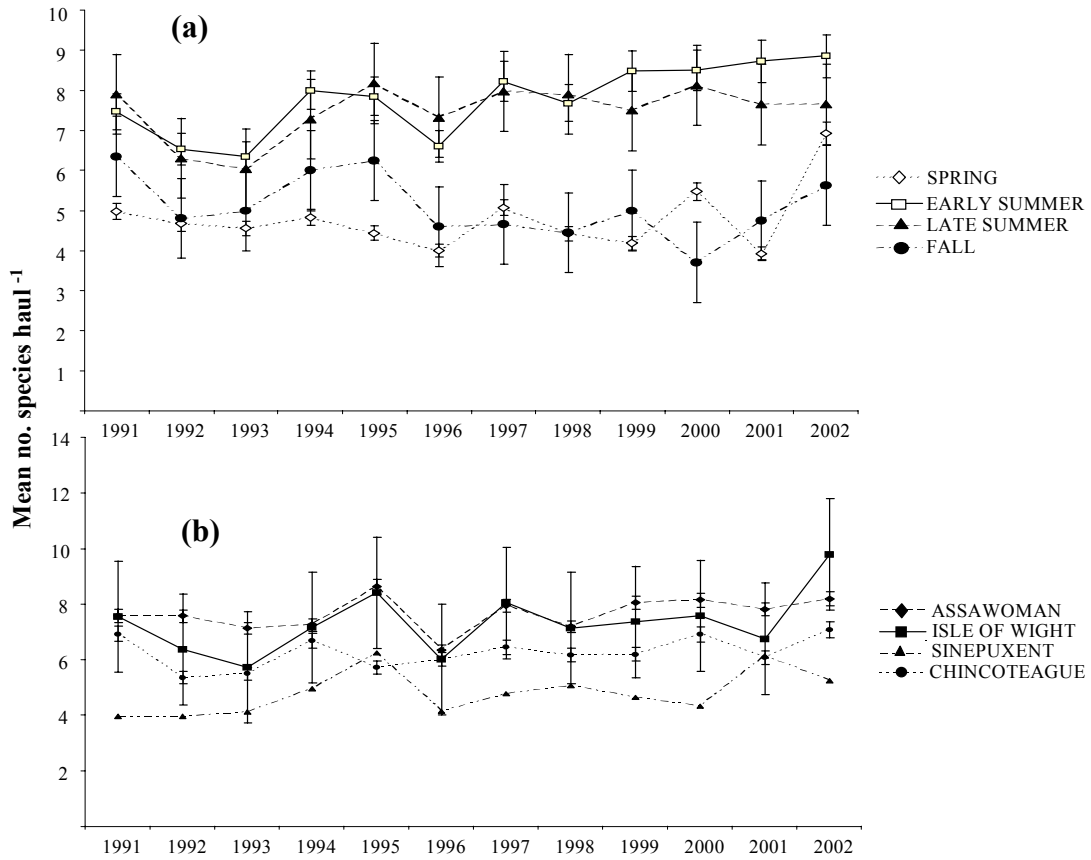
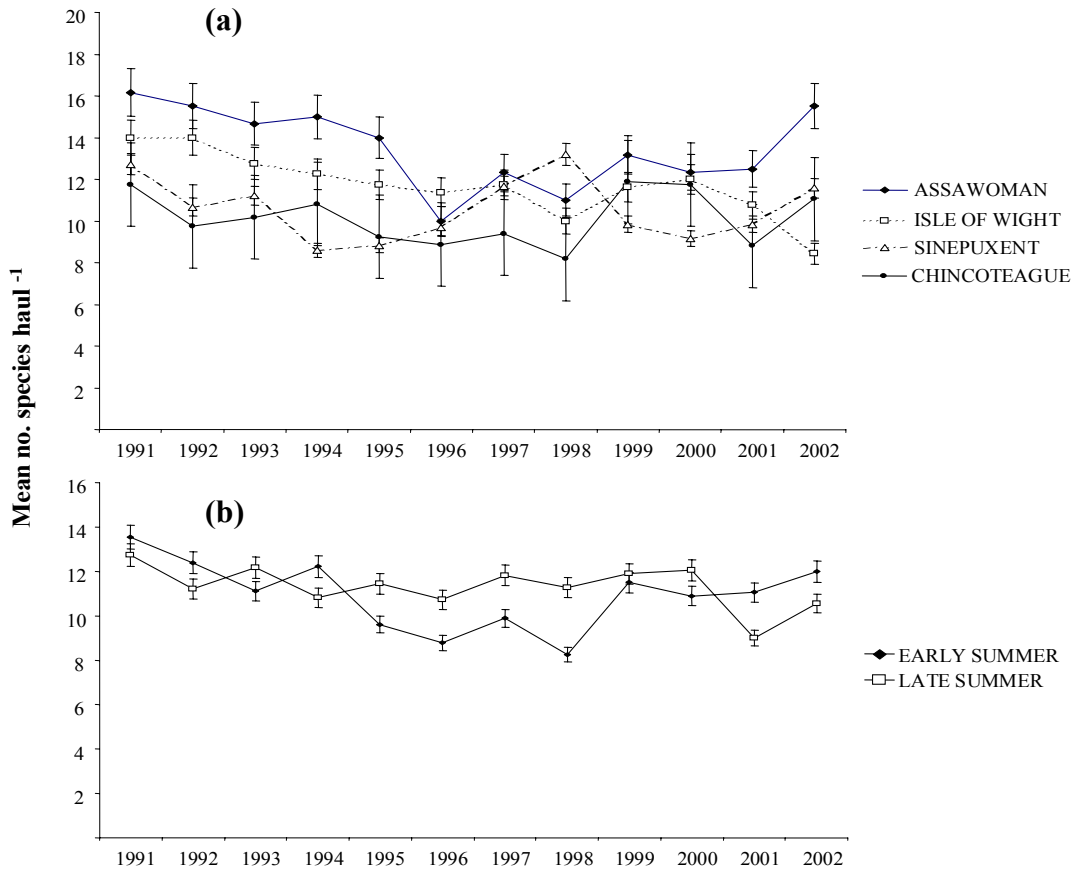


Figure 24: Mean number of species per haul for 1991-2002 MD DNR coastal finfish seine survey by bay (a) and season (b). Bars represent standard error.



(Table 11). The two summer seasons (early and late) were not statistically different from each other, but each was significantly higher than spring and fall samples based on Tukey's protected pairwise comparisons. The mean species richness per haul for the trawl data was without trend during the survey period, but showed a small peak in all bays except Chincoteague in 1995. A second peak was evident for 2002 for Isle of Wight bay.

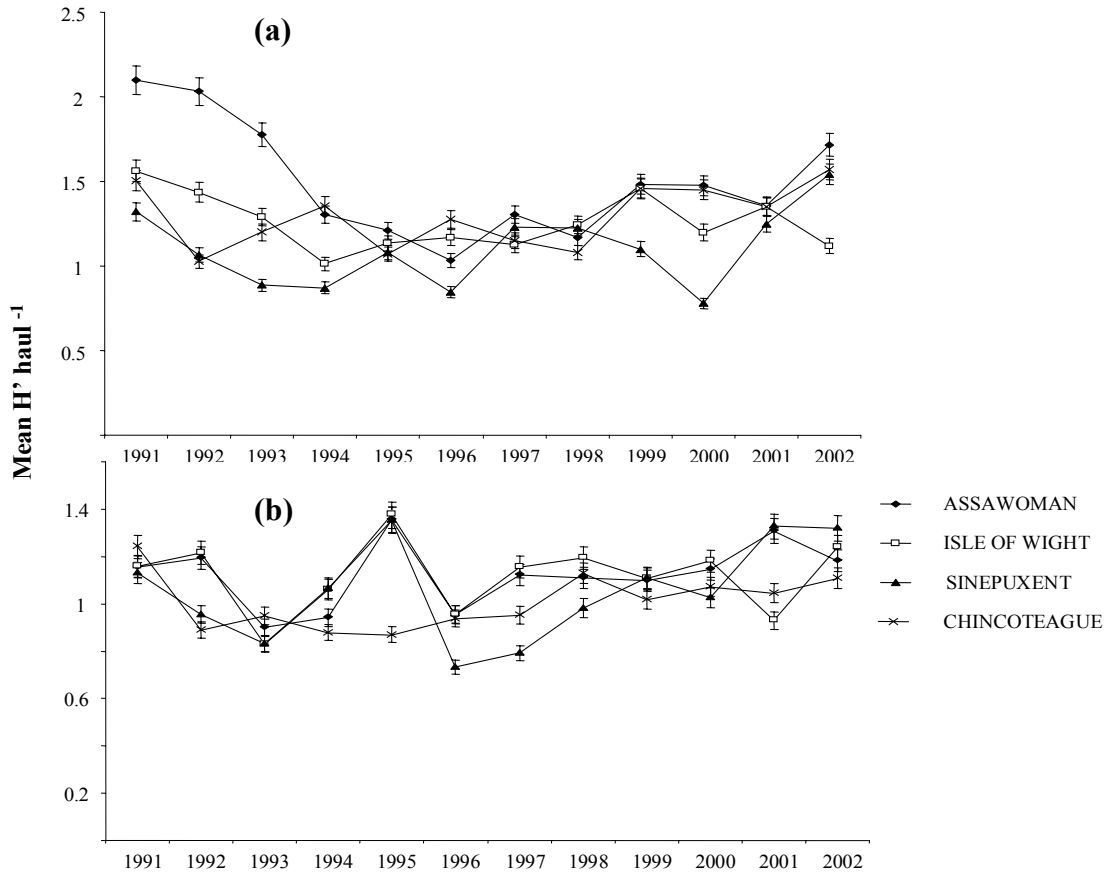
For the seine data, there was a significant difference between mean species richness by bay ($F = 13.8, p < 0.0001$) with Assawoman, Isle of Wight, and Chincoteague being statistically different from each other (Table 11) and Sinepuxent grouping with the neighboring bays of Isle of Wight and Chincoteague. The two northern bays again had the highest diversity values. There was no significant seasonal difference between early and late summer seasons ($F = 0.93, p > 0.335$). From 1991 through 2002, there was a general downward trend in richness through the 1990's followed by an upward trend in 1998 through 2002, particularly evident for Sinepuxent and Assawoman Bays. In Assawoman Bay, species richness dropped c. 40% from 1991 ($S=13.8 \text{ haul}^{-1}$) to 1996 ($S=7.6$), but subsequently regained the 1991 level in 2002.

The Shannon-Wiener diversity index H' varied across years for both gear types (Figure 25) and was statistically different between bays for seine ($F = 6.9, p < 0.0001$) and trawl surveys ($F = 5.5, p < 0.001$). For the seine survey, H' values declined 1991-1996 across all bays, but this trend was highly variable between bays (as noted by greater separation of curves in Fig 25a). After the 1996 low point, diversity increased and the difference between systems decreased as the curves are closer together. There was no significant difference between early and late summer for seine H' , but there was a

Table 11: Comparison of mean species richness (per haul) between bays and seasons by gear type, using Tukey’s protected test for pairwise comparisons. Bays or seasons with same group letter were not statistically different. N/S = non-significant; NA = not applicable

Bay	Group		Mean		N	
	Trawl	Seine	Trawl	Seine	Trawl	Seine
Assawoman	A	A	7.61	13.5	277	72
Isle of Wight	A	B	7.26	11.6	365	99
Sinepuxent	B	C	4.78	10.1	261	71
Chincoteague	C	BC	6.25	10.6	841	193
Season						
Spring	A	NA	4.8	NA	482	NA
Early summer	B	N/S	7.75	10.93	503	217
Late Summer	B	N/S	7.45	11.31	505	218
Fall	A	NA	5.12	NA	254	NA

Fig. 25: Shannon-Weiner diversity index (H') for 1991-2002 MD DNR coastal finfish survey across years and between embayments for both seine (a) and trawl (b). Bars represent standard error.



seasonal effect in the trawl data ($F = 67.9$, $p < 0.0001$) (Table 12). Similar to species richness, H' was highest in the northern bays and higher during summer months than during spring or fall.

The k-dominance curves (Figure 26) for the trawl data overlapped and therefore it is difficult to make definitive statements about dominance patterns across the bays. Nonetheless, greater dominance by just a few species in Chincoteague Bay was indicated by a higher y-intercept. The slopes were similar for the trawl curves, but Sinepuxent flattened out sooner indicating less dominance and perhaps a greater number of rare species. K-dominance curves for the seine data showed greater separation between embayments, but again because of crossing curves, one cannot make definitive interpretations regarding dominance patterns. Assawoman Bay exhibited a lower curve and reached an asymptote at a higher cumulative number of species, indicating greater diversity and reduced dominance. Sinepuxent, on the other hand, was dominated by a few species as indicated by a high y-intercept. The k-dominance curves for each season (Figure 27) in the trawl data showed good separation among seasons. The fall curve was consistently lower demonstrating greater diversity and less dominance in comparison to summer seasons. Seine k-dominance curves indicated higher diversity in the early summer in comparison to late summer.

Table 12: Pairwise comparisons of Shannon-Wiener diversity index (H') values for each gear type between season and embayment. Bays or seasons with same group letter were not statistically different. NA = not applicable, N/S = not significant

Bay	Group		Mean		N	
	Trawl	Seine	Trawl	Seine	Trawl	Seine
Assawoman	AC	A	1.10	1.50	277	72
Isle of Wight	C	B	1.1	1.26	365	99
Sinepuxent	ABC	B	1.05	1.11	261	71
Chincoteague	B	B	1.01	1.29	841	193
Season						
Spring	A	NA	0.84	NA	482	NA
Early summer	B	N/S	1.13	1.33	503	217
Late Summer	BC	N/S	1.20	1.25	505	218
Fall	D	NA	0.96	NA	254	NA

Fig. 26: K-dominance curves for each embayment by gear type.

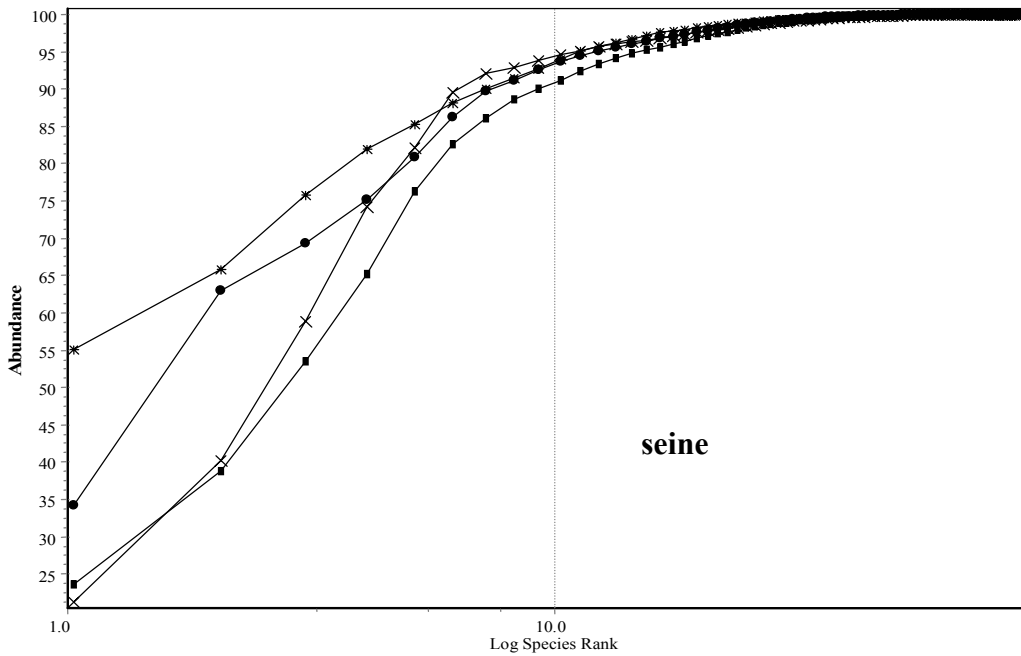
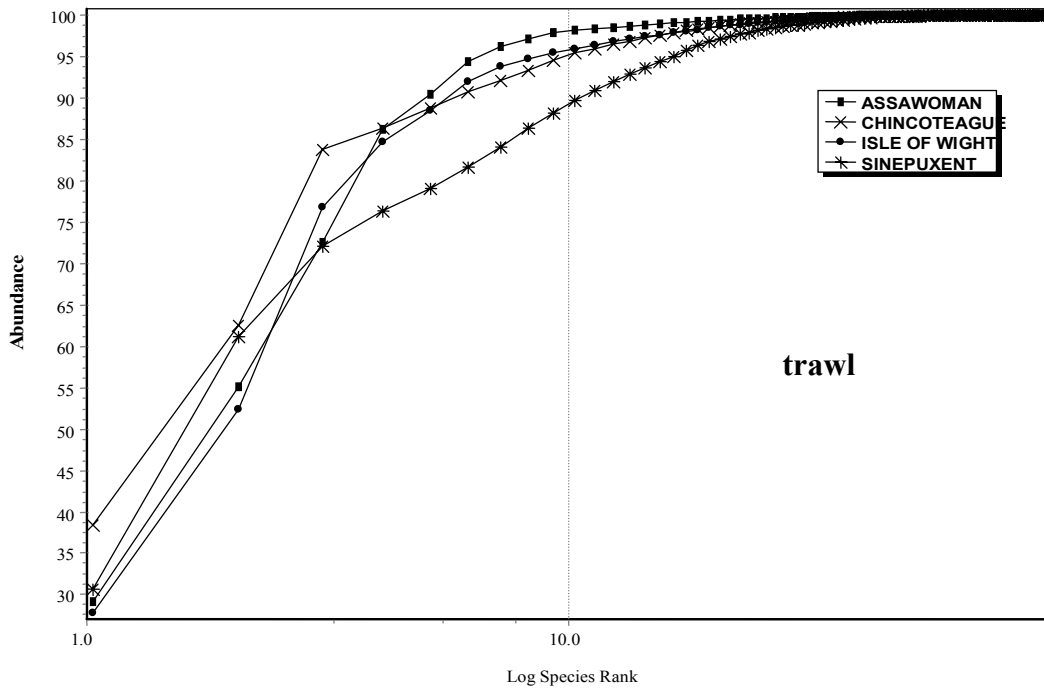
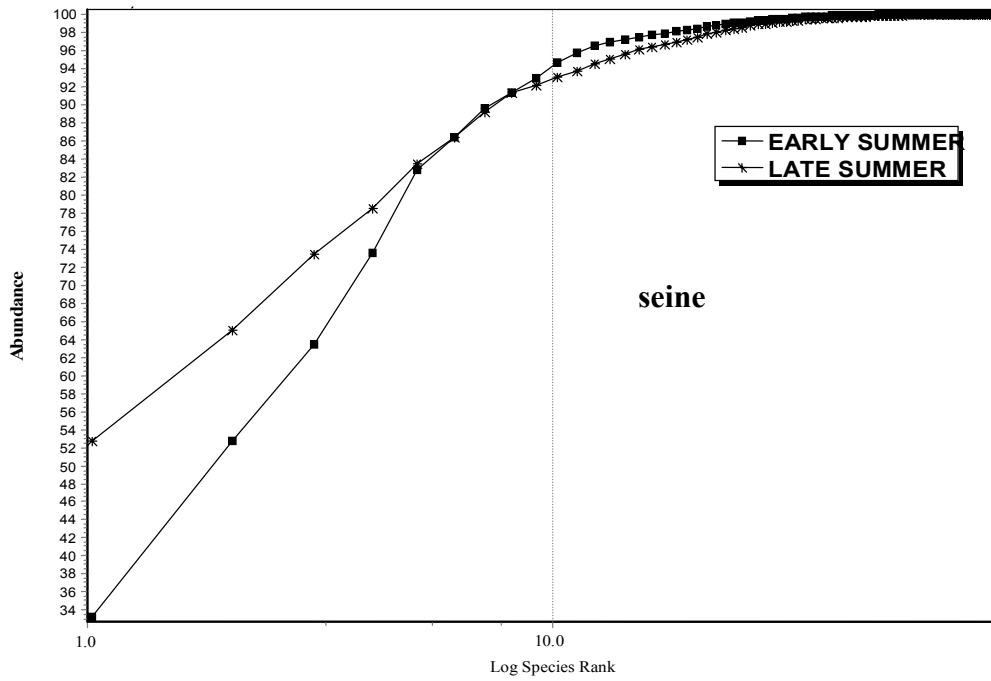
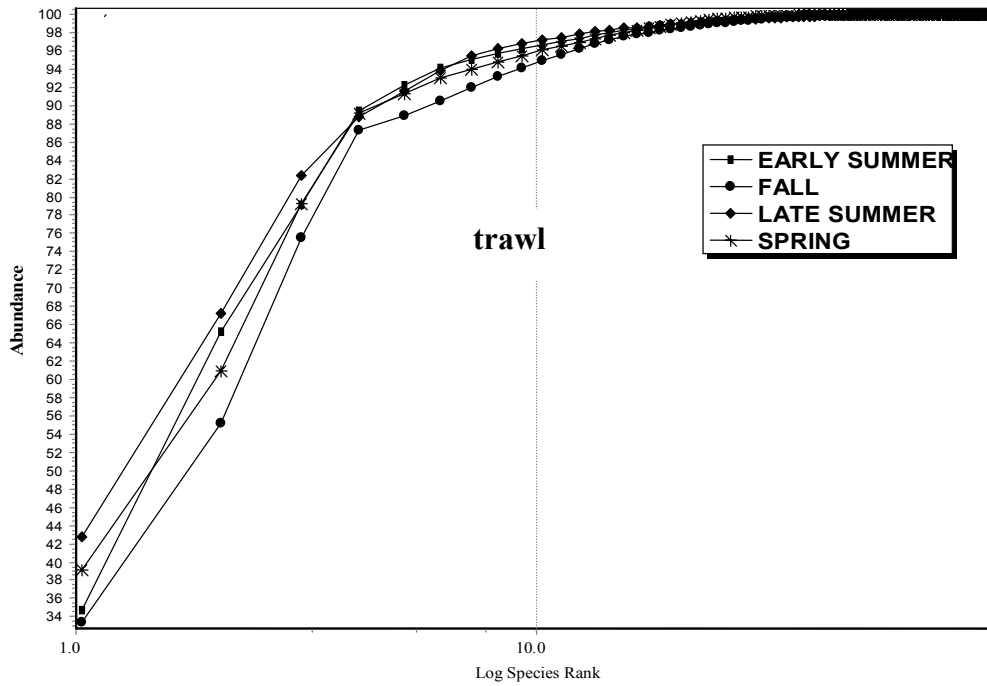


Fig. 27: K-dominance plot for each season by gear type



DISCUSSION

Fish Assemblage by Embayment

The coastal bays of Maryland support a diverse array of fish species comparable to other estuarine systems of the Mid-Atlantic Bight (Table 13) in terms of number of species and diversity. Ordination and cluster analyses demonstrated that each bay harbors a distinct grouping of finfish and blue crab, and that these differences between systems were due to the relative abundances of just a few species. Further, there was a general pattern of higher diversity and abundance of dominant species in the two northern bays in comparison to the two southern bays.

Inlets are the major, if not only source for marine transients into coastal lagoons (Mariani 2001). While each of the Maryland embayments supports distinct assemblages, it is apparent that there is a discrete effect produced by the inlets on those sites closest to them. Most of the sites align with other sites of the same embayment suggesting that the sites represented the assemblage of the embayment closely. Yet, some sites were distinctly ordinated away from their respective embayment and these may have been related to proximity to direct connections to coastal and freshwater inputs. For example, trawl sites 12 and 20 were anomalous from other sites in Chincoteague Bay (Fig.19). Site 12 lies within the northwestern portion of the bay in the sub-embayment of Newport Bay. This watershed (fed by Trappe Creek and Ayres Creek) drains the town of Berlin and the water quality is marginal to poor (Wazniak et al. 2004). This site is more aligned with those of the northern bays presumably due to similar water quality conditions supportive of similarly tolerant species (e.g., menhaden). On the other hand, site 20 is the southernmost site and while it was an outlier across all bay sites, it was loosely ordinated

Table 13: Comparison of species richness between adjacent coastal systems along the mid-Atlantic shore.

System	No. Species	Duration of Study (years)	Sampling Gear	Source
Chesapeake Bay	57	5	Mid-water trawl	Jung and Houde, 2003
Delaware River	63	14	Beach seine	Weisberg et al, 1996
DE Coastal Bays	46	3	Beach seine and otter trawl	Derickson and Price, Jr., 1973
MD Coastal Bays	94	12	Beach seine and otter trawl	This study
Great Bay, NJ	26	3	Otter trawl	Martino and Able, 2003
Atlantic Ocean surf zone, VA	12	3	Surf seine	Layman, 2000
Pamlico Sound, NC	78	2	Otter trawl	Ross and Eppersly, 1985

with Sinepuxent sites (Fig. 19). This site was most proximate to the southern inlet (Chincoteague, VA) and increased marine influence may have caused this site to ordinate close to Sinepuxent Bay, which is most proximate to the marine influence of Ocean City Inlet. Trawl site 7 was anomalous among Isle of Wight sites and appeared to be more associated with the Sinepuxent sites. Again, this may reflect the proximity of this site to the Ocean City Inlet and the marine influence on the assemblage there.

Among embayments, seine sites showed considerably more overlap than trawl sites (Fig. 12b), although the embayments maintained similar ordination. Anomalous sites included sites 4 (Isle of Wight), and sites 14 and 18 (Chincoteague). Site 4 is adjacent to the Ocean City Inlet and therefore corresponds more closely to the sites of Sinepuxent (e.g., site 9 that ordinated similarly). Interestingly, sites 14 and 18 ordinated closely and remained somewhat separated from the other Chincoteague sites. The initial expectation would be for site 18 (the southernmost seine site) to ordinate more closely to those sites in Sinepuxent, as observed for the trawl data. This would not explain its similarity to site 14, which is located mid-bay between the Ocean City Inlet and the Chincoteague Inlet. A more likely explanation is that these are the only sites occurring on the eastern side of Chincoteague Bay on Assateague Island. Dense and large seagrass beds uniquely occur in this region. Therefore, this could be a result of differing SAV habitat availability in this area.

Previous studies comparing assemblages in estuarine and coastal systems have focused on trends in species richness along gradients (Wagner 1999; Jung and Houde 2003; Martino and Able 2003), but rarely make direct comparisons between constituent systems using community metrics (Weinstein 1985). Maryland's coastal embayment

assemblages were characterized broadly by the species richness trends, Shannon-Weiner index and the k-dominance curves. In terms of trawl diversity measurements, Assawoman and Isle of Wight were not different, and Sinepuxent shared attributes of all the other embayments (Table 12). Chincoteague, on the other hand, was different than Isle of Wight and Assawoman Bays. The k-dominance curves supported these univariate interpretations of biodiversity between systems. The lower curves (Fig. 26) of Assawoman (primarily the seine survey) indicated greater diversity and less dominance in this bay. Conversely, Sinepuxent tended to show higher dominance indicated by a more arced curve, particularly evident in the seine data.

The seine data (i.e., the littoral zone) shows a more compelling dominance pattern as indicated by the dominance curves. Sinepuxent and Isle of Wight littoral zones were dominated by fewer species, perhaps indicative of either less habitat heterogeneity or less influence by transients. The first scenario is unlikely since Sinepuxent is well covered with eelgrass and has topographic heterogeneity due to the tidal action maintaining the channel. This dominance pattern may be alternatively the result of few species capable of prolonged inhabitation in these two embayments due to higher flow and other oceanic influences. The lower dominance seen in Assawoman and Chincoteague suggests that marine transients (which favor littoral habitats) are not spending time in Sinepuxent or Isle of Wight, but may be using these systems as corridors to reach areas in Chincoteague or Assawoman Bays. If this were the case, transient species would be using the channel for movement and the trawl survey would show more equitability in the dominance curves, which it does. In addition, the SIMPER analysis showed that Sinepuxent and Isle of Wight littoral zones were dominated most by *M. menidia* (Table 10) and that this

species is replaced as the dominant species in the other two bays by *B. tyrannus* (Assawoman) and *C. sapidus* (Chincoteague). *M. menidia* feeds primarily on zooplankton and it is likely that this preferred food source is in greater abundance nearer to the inlet region. In addition, because of this visual feeding strategy, *M. menidia* may face difficulties in feeding success in Assawoman where water clarity is decreased (Wazniak et al. 2004) and the dominant species switches to a filter feeder, *B. tyrannus*. Wazniak et al (2004) report low Secchi depth (a measure of water clarity) in Assawoman Bay suggesting that decreased visibility may play a role in structuring the fish assemblage there.

Species richness is the most widely used diversity measure and perhaps the easiest metric to use when assessing the biodiversity of a system (Hill 1973; Stirling and Wilsey 2001; Foggo et al. 2003). The number of species that a system can support indicates ecosystem stability, suitable habitat, niche availability, and trophic structure. There were significant differences in the number of species taken per sample between each of the bays, with the northern bays (Assawoman and Isle of Wight) supporting a higher richness than the southern bays. The trawl data indicated no difference between the two northern bays, while Chincoteague and Sinepuxent were different, with a particularly low number of species found in Sinepuxent ($\sim 4.8 \text{ haul}^{-1}$) (Table 11). The seine data exhibited a comparable pattern although there was greater overlap between Isle of Wight and Chincoteague, and a significant difference between the two northern bays. For both data sets, Assawoman consistently supported a higher number of species and individuals per unit effort even though the total number of species caught in the system (by trawl) was lowest (59 total). Total species richness for Assawoman seine data was second lowest

(after Sinepuxent-Table 6). This presents an interesting contrast from several viewpoints. On one hand, Assawoman Bay is supporting high diversity at the sample level as measured by species richness, diversity, and abundance. However, overall, not as many species are coming into the system over a larger temporal scale as it had the fewest number of species encountered. Assawoman Bay therefore is supporting a resident assemblage that is fairly diverse, but a fewer number of species exploits the bay over time. Causal factors of these differences could include eutrophication and associated water quality, habitat heterogeneity, or disturbance.

Eutrophication is an ongoing issue in coastal areas, and the lagoon system of Maryland is no exception (Boynton et al. 1996; Wazniak et al. 2004). The northern bays face increasing nutrient loads coupled with reduced habitat heterogeneity due to relatively low coverage of seagrasses (predominantly *Z. marina*) and increased development of the surrounding resort community. Thus, the carrying capacity of these systems may be altered. It has been argued that increased nutrient loads lead to increased fish productivity (Lee and Jones 1991; Nixon and Buckley 2002) as a result of escalated primary production. The higher abundances in the northern bays are consistent with this view because these bays are significantly more eutrophic than the southern two bays (Boynton et al. 1996). Still, eutrophication can also significantly affect the types of fish present in a waterbody (Caddy 2000). Generally, highly eutrophic waterbodies tend to have dominant populations of “rough” or pelagic fish that feed at lower trophic levels (Lee and Jones 1991; Price 1998). While the exact mechanism of the change from higher trophic level species (piscivores) to lower trophic level species with increasing fertility is poorly understood, some authors have suggested that it is related to such factors as

reduced foraging by piscivorous species brought about by increased turbidity from increased concentrations of phytoplankton (median values [2001-2004] for Chl a [$\mu\text{g L}^{-1}$]: Assawoman = 15, Isle of Wight = 11, Sinepuxent = 5, Chincoteague = 5, from Wazniak et al, 2004). This phenomenon may be taking place in Assawoman (and to an extent Isle of Wight) as the overall number organisms caught is consistent with higher eutrophication, but the higher mean species per haul in the northern systems is inconsistent with expectations. Price (1998) compared patterns of littoral species assemblages in the Maryland embayments with those of Delaware (heavily eutrophied and lacking seagrass habitat) and concluded that the eutrophied systems in Delaware shift to a *Fundulus*-dominated system in contrast to menhaden, spot, and silversides in more oligotrophic systems. However, CPUE (seine) for *F. heteroclitus* and *F. majalis* was generally higher in Sinepuxent Bay (Table 5) than the other bays contrary to an expectation of higher abundances in the two northern bays should this genus represent eutrophication patterns. Caddy (2000) has proposed that eutrophication may cause a shift in dominance from demersal species to pelagic species that are better equipped to exploit increases in primary production. Experimental work addressing issues of reduced foraging ability and a demersal-pelagic shift would prove quite valuable for resource managers as most piscivores are also commercially or recreationally valuable species.

Seasonal Variability of Dominant Species

Due to the differing temporal nature of the two surveys, the primary focus of seasonal analysis rested with the trawl data. The trawl statistics showed that peak catches occurred in the summer seasons, with abundances much reduced in spring and fall. The ordinations of sites by bay were maintained in large part across seasons (Fig.20).

Interestingly, sites 7, 12, and 20 consistently ordinated away from other sites of the same embayment, regardless of season. There was only moderate support for seasonal changes in the assemblages; early and late summer data ordinated in separate clusters. The pairwise comparisons between seasons, although all statistically significant, resulted in low R values, indicative of strong overlap in species composition between seasons. This suggests that while the seasons were discrete in terms of assemblage structure, many species were using these embayments for longer than one season, with the notable exception of herring (*C. harengus*) that is dominant within the overall survey dataset, but only appears in the spring (Fig. 11). This predictable springtime appearance of herring once supported a short-term anchor gill net fishery in upper Chincoteague Bay (Schwartz 1961) but its abundance or fishery has not been historically noted for Assawoman Bay (Schwartz 1964). The differential use of the embayments has apparently shifted as the CPUE for this species was highest in Assawoman and lowest for Sinepuxent over the course of this survey. Seasonality of this species was also reflected within the CART analysis which showed that temperature was the most important explanatory variable for their abundance. Herring caught during the course of this survey were generally small (< 70 mm) (R. Murphy, unpublished data), though, suggesting that only juvenile herring are entering the bays. Schwartz's work (1960, 1964) suggests that adult herring exploited the bays in large numbers that have not been recorded in recent times. This discrepancy indicates differential use of the habitat over large temporal scales where adults no longer enter the system and are therefore absent from the data. Another possible explanation for this apparent shift in life stage is the differing gears; gill nets select larger fish than the trawl and seine gear used in state surveys.

Not only did herring show considerable seasonal abundance patterns, but other important commercial species showed temporal variation based on migratory, foraging, or exploitation factors. Blue crab (*C. sapidus*) showed a consistent peak in abundance in early summer in both the trawl and seine data (Figs. 4 and 5). This pattern occurred across all embayments. Blue crabs exhibit migratory patterns in the coastal embayments (Cargo 1958) although little salinity gradient exists. Cargo (1958) showed through tagging experiments that crabs migrated to the southern inlet at Chincoteague although they were released much nearer to the inlet at Ocean City. Cargo also noted that there may be a resident population that completes its life cycle entirely within the embayments without migrating into coastal waters. The seasonal patterns in abundance in this study supported a possible late summer emigration to the ocean, although another possible explanation that deserves further investigation is that harvest pressure on a closed population from commercial and recreational crabbers during the summer months may cause a marked decline from early summer to fall months. Although seasonal catch data is not currently available for the coastal bays crab fishery and thus fishing mortality estimates are unattainable, the fishing pressure follows that of the Chesapeake Bay where peak effort occurs in mid-to late summer. Therefore, the seasonal decline in abundance was likely due to fishing pressure. Lastly, a recent infestation of a crustacean parasite (*Hematodinium* sp.) may also play a role in decreasing abundance over the course of a year (Wazniak et al. 2004). The impact of this parasite is currently under investigation.

Bay anchovy (*A. mitchilli*) was a dominant species in both trawl and seine surveys and showed modest fluctuations in abundance across years (Figs. 6 and 7). This species is abundant in mid-Atlantic estuaries (Weisberg et al. 1996; Jung and Houde 2003) and is

a critical forage species for commercially valuable species (Murdy et al. 1997; Scharf et al. 2002). Bay anchovy abundance followed a similar pattern to that of blue crabs in that there was a noticeable decline in abundance from early summer to late summer evident by both surveys, with consistently low abundances in spring and fall. Possible explanatory habitat variables were identified through the CART analysis including seagrass coverage (positive relationship) and dissolved oxygen (negative relationship). The negative response to DO levels is consistent with Jung and Houde's (2003) finding that bay anchovy may be aggregating in areas of increased phytoplankton abundance, which would create low DO patches. Low patches of DO result from increased community respiration in relation to primary productivity indicating increased feeding (York et al. 2001). Spot (*L. xanthurus*), showed the same pattern of peak abundance in summer months, and low abundances in spring and fall. The notable exception to this was in 1994 when spot abundance was at its peak (trawl and seine) during the 12-year survey period. Because of this large abundance of spot, it appeared that the population may have "lingered" based upon high abundances during the fall sampling season. Spot also exhibited high interannual variability ($CV_{\text{Trawl}}=1.68$, $CV_{\text{Seine}} = 1.49$ seine) with fluctuating catches peaking every two to three years that is reflected in both gear types (Figs.8 and 9). Spot apparently utilize the coastal bays as nursery areas, entering as juveniles in the early spring and spending the summer months feeding and growing before migrating offshore and to the more southerly locations (Murdy et al. 1997).

Weakfish (*C. regalis*) were completely absent in the spring, with abundance increasing early to late summer and declining in the fall (Figure 10). Murdy et al (1997) report a spring inshore migration in regions north of Cape Hatteras as weakfish begin

spawning in nearshore coastal waters. The abundance patterns observed in this survey are likely capturing young-of-year weakfish as they enter the embayments to feed. As a demersal species, weakfish tend to aggregate in deeper waters and are therefore better represented in the trawl survey and are only a minor component of the seine survey. Another demersal sciaenid, croaker (*M. undulatus*), also only appeared in the trawl survey. This species is uncommon in spring and is generally at its peak abundance in fall (Fig. 12). This coincides with Schwartz's (1964) observation that croaker first appear in July at the Ocean City Inlet and "fan out" into the embayments from there before emigrating out in the fall. There was no evidence for a spatial gradient in croaker abundance among sites as a function of distance to inlet. The greater numbers observed in fall in this survey was probably indicative of increasing size and vulnerability to capture. Silver perch (*B. chrysoura*) also make their first appearance in early summer for both gear types (Figs. 14 and 15). Unlike croaker they reached peak abundance in late summer (Murphy et al. 1997) with numbers declining in the fall. Both gear types reflect this pattern with the seine capturing larger numbers of individuals, likely due to the littoral habits of this species. Interestingly, no silver perch were recorded from the seine survey in early summer 1999, although they do appear in the trawl survey from the same period.

Summer flounder (*P. dentatus*) was a dominant species in the trawl survey (rank=7) but of only moderate abundance in the seine survey (rank=16). Peak abundance occurred across all bays in early summer (Fig. 13). This species exhibited moderate levels of interannual variability (CV=0.44). Summer flounder is one of the most important recreational and commercial species along Maryland's coast (Wazniak et al.

2004) and its population trends are closely monitored. Juvenile summer flounder utilize *Zostera* beds (Murdy et al. 1997) in nearshore regions and migrate to deeper waters as they age. After age 3, summer flounder migrate to and remain in coastal waters where spawning takes place. Therefore, the fact that both of the northern bays, which have lesser coverage of *Zostera*, generally supported higher numbers of summer flounders in all seasons is at odds with the known life history pattern. Furthermore, the CART analysis for summer flounder abundance indicated that although SAV coverage was important, this species' abundance was largely determined by an interaction of temperature and tidal state (Fig. 18), although the mechanism of tidal influence is non-directional.

Menhaden (*B. tyrannus*) is another valuable commercial species that also serve as an important forage species. Menhaden was a dominant species in both gear types, particularly in the littoral zone sampled by seine. Both surveys showed an increase in abundance in the early summer with the total catch tapering off in late summer and fall months. This was most apparent particularly in the seine data, while the trawl data showed a greater degree of variability between seasons (Figs. 16 and 17). Menhaden in the US mid-Atlantic exhibit protracted spawning with a peak during the winter offshore (~ 16-32 km) where the larvae are then transported into estuarine systems to feed and grow through the summer. Both gear types demonstrate high interannual variability ($CV_{\text{Trawl}} = 1.19$, $CV_{\text{Seine}} = 1.14$) for this species.

Community structure

Contemporary theories attribute local community structure to at least four parameters (or combinations thereof): predation, competition, disturbance, and spatial heterogeneity (Menge and Sutherland 1976; Menge 1995; Connell 2002). Most studies on the role of competition and disturbance in fish communities have come from coral reefs (Steele 1997; Almany 2004) or freshwater systems (Werner and Hall 1977), while few estuarine studies have considered these factors (Weinstein 1985). Although this study was not undertaken to identify those factors responsible for community structure per se, inferences may be possible based upon the emergent assemblage structure described. For example, the continued decline of the forage fish index (comprised of the annual log CPUE of spot, bay anchovy, Atlantic silverside, and Atlantic menhaden) developed by the state (Wazniak et al. 2004) may reflect the continued loss of littoral habitat in the more developed bays to the north, thus reducing overall habitat heterogeneity. Yet, the production levels for these embayments are much higher (Table 6) which may serve to dampen the influence of low structural complexity. The forage fish index is a useful measure of overall prey availability as distributed across all bays, but lacks the spatial resolution that may be underlying changes in the index. The CART analysis did not indicate relationships between the various species comprising the index, suggesting that these populations are responding independently of each other. Furthermore, because the embayments are supporting varying populations of these species, the overall downward trend of this index may only be driven by one species, or abundances in a given bay. If this is the case, there may be confounding underlying mechanisms determining the composition of the forage species, and lack of a

straightforward reason (e.g., degraded habitats throughout the lagoon complex) for negative or positive trends.

Habitat complexity and competition are likely influences on fish assemblages in coastal lagoon systems, but simple area or volume of habitat can explain substantial variability in species richness across ecosystems (Wootton 1999; Frank and Shackell 2001). Wootton (1999) asserts several hypotheses that may account for a positive correlation between species richness and area. Large areas by definition provide more space and thus are more likely to support rare species. Chincoteague Bay (area = $189 \times 10^6 \text{ m}^2$ from Boynton et al, 1996) is by far the largest embayment and did exhibit the highest overall species count (Table 6). Additionally, larger areas are more likely to support diverse habitats (e.g. oyster reefs, seagrass, and channel edges) thus supporting more species exploiting these niches. Chincoteague has a greater amount of seagrass coverage than the other three embayments, but all are devoid of reef structure (Wazniak et al. 2004). The third possible reason for the area-richness relationship suggested by Wootton (1999) is that assemblages in small areas are more likely to incur high rates of extinction and lower rates of immigration. The extinction scenario is unlikely in the coastal bays due to the proximity of alternative habitats; however, differential immigration may be a plausible explanation for the low mean species per haul seen in Sinepuxent Bay, if this bay receives less oceanic influence due to reduced advection (see Introduction). Still, species are arriving in high diversity to Chincoteague. It is possible that the southern route for immigration is a dominant determinant of diversity throughout the Chincoteague. Lastly, it is possible that overall richness values were higher in

Chincoteague Bay due to the increased sampling effort there compared to the other three embayments.

Habitat and Assemblage Metrics

Catch per unit effort was 150% higher for the trawl survey in comparison to the seine survey and could suggest differences in abundances between littoral and profundal habitats (except for the Sinepuxent Bay where seine CPUE was c. 5-fold greater than trawl CPUE) (Table 6). These differences should be interpreted with caution as gear efficiency varies significantly, and each species is differentially selective to the gear types. This finding also conflicts with earlier studies (Whitfield 1993) where standardized sampling gear used along a gradient found that CPUE was up to three times larger in the littoral habitat versus profundal. Future analyses should examine the relative amount of each habitat type within the individual systems, and account for amount of available habitat compared to the amount sampled by each gear type. Considerations of differential habitat use and potential habitat-specific production rates are important when devising land-use and other management plans that potentially impact the two zones differently.

The littoral zone is important as an area of increased seagrass coverage, predominantly *Ruppia maritima* and *Zostera marina*. These beds harbor large numbers of fishes (Orth and Heck Jr. 1980; Olney and Boehlert 1988; Heck Jr. et al. 2003) and have made a notable recovery over the course of this survey (Wazniak et al. 2004). Although seagrass coverage did not have a significant effect on either species richness ($p = 0.19$) or CPUE ($p = 0.1$), it can affect seine efficiency, and thereby lead to under-

represented estimates of diversity and abundance (Serafy et al. 1988). Still, as indicated previously, the CART analysis showed a positive relationship between seagrass coverage with abundances of *A. mitchilli* and *P. dentatus* (Fig.18). Although increased seagrass coverage might be expected to result in increased juvenile fish production across the embayments, the forage fish index has declined concomitantly with increasing SAV coverage. One possible explanation may be that the bays differ in suitable seagrass habitat. However, Wazniak et al (2004) standardized the seagrass coverage for each system by calculating percentage of embayment covered. Chincoteague and Sinepuxent Bays are well vegetated (32 and 36 %, respectively) compared to Assawoman and Isle of Wight (8 and 6 %, respectively). Seagrass coverage may be important in many coastal systems, but other factors appear to be reducing the effects.

Most species encountered in the mid-Atlantic coastal bays are spawned elsewhere and enter the coastal bays either passively (via oceanic transport) (Warlen et al. 2002) or actively as they search for foraging regions and/or refugia (Heck Jr. et al. 2003). Recruitment (e.g., menhaden and blue crab) is often driven externally by drivers such as oceanic circulation and wind patterns (Epifanio and Garvine 2001; Warlen et al. 2002) although it is likely that many of the Fundulids represent a population recruiting from within the coastal bays. Thus, the assemblages of these systems are structured by processes of differing degree with large-scale external processes likely affecting the four connected bays similarly. In this respect, the embayments respond as open systems with the chief external driver at the Ocean City Inlet (and to a lesser extent, the Chincoteague Inlet given its further distance from the northern bays) serving as the oceanic vector. On a smaller scale, the embayments may exhibit attributes of closed systems. The northern

bays are differentially affected by anthropogenic influences and therefore the assemblages here are shaped by local conditions in comparison to the southern bays, which are less impacted and maintain different populations.

This study used multiple assemblage metrics to evaluate faunal attributes across four interconnected coastal lagoons. The ability to discriminate differences between the associated coastal lagoons based solely on the composition of species suggests that management actions may need to be tailored to individual lagoon systems and their attributes rather than applied generically across all bays. Likewise, the possible influence of increased eutrophication is a likely explanation for the increased productivity levels observed in the northern bays given the lower habitat availability and reduced water quality. In addition, because the differences between each embayment were characterized by changes in relative abundances of a few species (as opposed to a complete turnover in species), there seems to be a suggestion of varying carrying capacities of each embayment. Seagrass coverage, although increasing over the course of this survey, did not have obvious effects on species diversity and abundance across or within the embayments, although it did have positive associations with two important species. Further experimental work is warranted to provide greater understanding of aspects driving species diversity, community structure, and trophic changes within these systems.

LITERATURE CITED

- Able, K. W. and M. P. Fahay (1998). The First Year in the Life of Estuarine Fishes in the Middle Atlantic Bight. New Brunswick, New Jersey, Rutgers University Press.
- Almany, G. R. (2004). "Does increased habitat complexity reduce predation and competition in coral reef fish assemblages?" Oikos **106**: 275-284.
- Boynton, W. R., L. Murray, J. D. Hagy, C. Stokes and W. M. Kemp (1996). "A comparative analysis of eutrophication patterns in a temperate coastal lagoon." Estuaries **19**(2B): 408-421.
- Breiman, L., J. H. Friedman, R. A. Olshen and C. J. Stone (1984). Classification and regression trees. Belmont, California, Wadsworth International Group.
- Caddy, J. F. (2000). "Marine catchment basin effects versus impacts of fisheries on semi-enclosed seas." ICES Journal of Marine Science **57**: 628-640.
- Cargo, D. G. (1958). "The migration of adult female blue crabs, *Callinectes sapidus* Rathbun, in Chincoteague Bay and adjacent waters." Journal of Marine Research **16**(3): 180-191.
- Cerco, C. F., C. S. Fang and A. Rosenbaum (1978). Intensive hydrographical and water quality survey of the Chincoteague/Sinepuxent/Assawoman Bay systems, Volume III. Non-point source pollution studies in the Chincoteague Bay system. Gloucester Point, VA, Virginia Institute of Marine Science.
- Clarke, K. R. (1993). "Non-parametric multivariate analyses of changes in community structure." Australian Journal of Ecology **18**: 117-143.
- Clarke, K. R. and M. Ainsworth (1993). "A method for linking multivariate community structure to environmental variables." Marine Ecology Progress Series **92**(3): 205-219.

- Connell, S. D. (2002). "Effects of a predator and prey on a foraging reef fish: implications for understanding density-dependent growth." Journal of Fish Biology **60**: 1551-1561.
- Cyrus, D. P. and S. J. M. Blaber (1992). "Turbidity and salinity in a tropical Northern Australian estuary and their influence on fish distribution." Estuarine, Coastal and Shelf Science **35**: 545-563.
- Day Jr., J. W. and A. Yanez-Arancibia (1985). Coastal lagoons and estuaries as an environment for nekton. Fish Community Ecology in Estuaries and Coastal Lagoons: Towards an Ecosystem Integration. A. Yanez-Arancibia (ed). Mexico, UNAM Press: 17-34.
- Day Jr., J. W., C. Hall, W. M. Kemp and A. Yanez-Arancibia (1989). Nekton, the free swimming consumers. Estuarine Ecology(ed). New York, Wiley-Interscience, John Wiley & Sons.
- Derickson, W. K. and K. S. Price (1973). "The fishes of the shore zone of Rehoboth and Indian River Bays, Delaware." Transactions of the American Fisheries Society **102**(3): 552-562.
- Desmond, J. S., D. H. Deutschman and J. B. Zedler (2002). "Spatial and temporal variation in estuarine fish and invertebrate assemblages: Analysis of an 11-year data set." Estuaries **25**(4A): 552-569.
- Eby, L. A. and L. B. Crowder (2004). "Effects of hypoxic disturbances on an estuarine nekton assemblage across multiple scales." Estuaries **27**(2): 342-351.
- Epifanio, C. E. and R. W. Garvine (2001). "Larval transport on the Atlantic continental shelf of North America: A review." Estuarine, Coastal and Shelf Science **52**: 51-77.
- Fairbridge, R. W. (1980). The estuary: its definition and geodynamics cycle. Chemistry and Biochemistry of Estuaries. E. Olausson and I. Cato (ed). New York, Wiley-Interscience, John Wiley & Sons, Inc.: 1-36.
- Fasham, M. J. R. (1977). "A comparison of nonmetric multidimensional scaling, principal components and reciprocal averaging for the ordination of simulated coenoclines and coenoplanes." Ecology **58**: 551-561.

- Field, J. G., K. R. Clarke and R. M. Warwick (1982). "A practical strategy for analyzing multispecies distribution patterns." Marine Ecology Progress Series **8**: 37-52.
- Foggo, A., M. J. Attrill, M. T. Frost and A. A. Rowden (2003). "Estimating marine species richness: an evaluation of six extrapolative techniques." Marine Ecology Progress Series **248**: 15-26.
- Frank, K. T. and N. L. Shackell (2001). "Area-dependent patterns of finfish diversity in a large marine ecosystem." Canadian Journal of Fisheries and Aquatic Sciences **58**: 1703-1707.
- Franzoi, P., R. Maccagnani, R. Rossi and V. U. Ceccherelli (1993). "Life cycles and feeding habits of *Syngnathus taenionotus* and *S. abaster* (Pisces, Syngnathidae) in a brackish bay of the Po River Delta (Adriatic Sea)." Marine Ecology Progress Series **97**: 71-81.
- Gillanders, B. M., K. W. Able, J. A. Brown, D. B. Eggleston and P. F. Sheridan (2003). "Evidence of connectivity between juvenile and adult habitats for mobile marine fauna: an important component of nurseries." Marine Ecology Progress Series **247**: 281-295.
- Hagan, S. M. and K. W. Able (2003). "Seasonal changes of the pelagic fish assemblage in a temperate estuary." Estuarine, Coastal and Shelf Science **56**: 15-29.
- Heck Jr., K. L., G. Hays and R. J. Orth (2003). "Critical evaluation of the nursery role hypothesis for seagrass meadows." Marine Ecology Progress Series **253**: 123-136.
- Henderson, P. A. and A. R. Margetts (1988). "Fish in estuaries." Journal of Fish Biology **33**: 254 pp.
- Hill, M. O. (1973). "Diversity and evenness: A unifying notation and its consequences." Ecology **54**(2): 427-432.
- Hurlbert, S. H. (1971). "The nonconcept of species diversity: a critique and alternative parameters." Ecology **52**(4): 577-586.

- Ishitobi, Y., J. Hiratsuka, H. Kuwabara and M. Yamamuro (2000). "Comparison of fish fauna in three areas of adjacent eutrophic estuarine lagoons with different salinities." Journal of Marine Systems **26**: 171-181.
- Jeppesen, E., J. P. Jensen, M. Sondergaard, T. Lauridsen and F. Landkildehus (2000). "Trophic structure, species richness and biodiversity in Danish lakes: changes along a phosphorous gradient." Freshwater Biology **45**: 201-218.
- Jung, S. and E. D. Houde (2003). "Spatial and temporal variabilities of pelagic fish community structure and distribution in Chesapeake Bay, USA." Estuarine, Coastal and Shelf Science **58**: 335-351.
- Kupschius, S. and D. Tremain (2001). "Associations between fish assemblages and environmental factors in nearshore habitats of a subtropical estuary." Journal of Fish Biology **58**: 1383-1403.
- Lalli, C. M. and T. R. Parsons (1997). Biological Oceanography. Oxford, U.K., Butterworth-Heinemann.
- Lambshead, P. J. D., H. M. Platt and K. M. Shaw (1983). "The detection of differences among assemblages of marine benthic species based on an assessment of dominance and diversity." Journal of Natural History **17**: 859-874.
- Lankford, R. R. (1977). Coastal lagoons of Mexico: Their origin and classification. Estuarine Processes. M. L. Wiley (ed). New York, Academic Press. **2**: 182-215.
- Lee, G. F. and R. A. Jones (1991). "Effects of eutrophication on fisheries." Reviews in Aquatic Sciences **5**(3-4): 287-305.
- MacArthur, R. H. and E. D. Wilson (1967). The theory of island biogeography. Princeton, New Jersey, Princeton University Press.
- Maes, J., S. van Damme, P. Meire and F. Ollevier (2004). "Statistical modeling of seasonal and environmental influences on the population dynamics of an estuarine fish community." Marine Biology **145**: 1033-1042.

- Magurran, A. E. (1991). Ecological Diversity and its Measurement. London, Chapman and Hall.
- Mariani, S. (2001). "Can spatial distribution of ichthyofauna describe marine influence on coastal lagoons? A central Mediterranean case study." Estuarine, Coastal and Shelf Science **52**: 261-267.
- Martino, E. J. and K. W. Able (2003). "Fish assemblages across the marine to low salinity transition zone of a temperate estuary." Estuarine, Coastal and Shelf Science **56**: 969-987.
- Maxted, J. R., S. B. Weisberg, J. C. Chaillou, R. A. Eskin and F. W. Kutz (1997). "The ecological condition of dead-end canals of the Delaware and Maryland coastal bays." Estuaries **20**(2): 319-327.
- May, R. M. (1975). Patterns of species abundance and diversity. Ecology of Evolution of Communities. M. L. Cody and J. M. Diamond (ed). Harvard, Belnap Press.
- Menge, B. A. and J. P. Sutherland (1976). "Species diversity gradients: Synthesis of the roles of predation, competition, and temporal heterogeneity." The American Naturalist **110**: 351-169.
- Menge, B. A. (1995). "Indirect effects in marine rocky intertidal interaction webs: patterns and importance." Ecological Monographs **65**(1): 21-74.
- Mouillot, D., S. Gaillard, C. Aliaume, M. Verlaque, T. Belsher, M. Troussellier and T. D. Chi (2005). "Ability of taxonomic diversity indices to discriminate coastal lagoon environments based on macrophyte communities." Ecological Indicators **5**: 1-17.
- Murdy, E. O., R. S. Birdsong and J. A. Musick (1997). Fishes of Chesapeake Bay. Washington, DC, Smithsonian Institution Press.
- Nixon, S. (1982). "Nutrient dynamics, primary production and fisheries yields of lagoons." Oceanologica Acta **4**: 357-371.

- Nixon, S., B. Buckley, S. Granger and J. Bintz (2001). "Responses of very shallow marine ecosystems to nutrient enrichment." Human and Ecological Risk Assessment **7**(5): 1457-1481.
- Nixon, S. and B. Buckley (2002). "'A strikingly rich zone'- Nutrient enrichment and secondary production in coastal marine ecosystems." Estuaries **25**(4b): 782-796.
- Olney, J. E. and G. W. Boehlert (1988). "Nearshore ichthyoplankton assemblages associated with seagrass beds in the lower Chesapeake Bay." Marine Ecology Progress Series **45**: 33-43.
- Orth, R. J. and K. L. Heck Jr. (1980). "Structural components of eelgrass (*Zostera marina*) meadows in the lower Chesapeake Bay--Fishes." Estuaries **3**(4): 278-288.
- Orth, R. J., K. L. Heck Jr. and J. van Montfrans (1984). "Faunal communities in seagrass beds: a review of the influence of plant structure and prey characteristics on predator-prey relationships." Estuaries **7**(4A): 339-350.
- Paperno, R., K. J. Mille and E. Kadison (2001). "Patterns in species composition of fish and selected invertebrate assemblages in estuarine subregions near Ponce de Leon Inlet, Florida." Estuarine and Coastal Marine Science **52**: 117-130.
- Pauly, D. and A. Yanez-Arancibia (1994). Fisheries in coastal lagoons. Coastal Lagoon Processes. B. Kjerfve (ed). Amsterdam, Elsevier Science Publishers B.V.: 377-399.
- Peet, R. K. (1974). "The measurement of species diversity." Annual Review in Ecology and Systematics **5**: 285-307.
- Peet, R. K. (1975). "Relative diversity indices." Ecology **56**: 496-498.
- Perez-Ruzafa, A., J. I. Quispe-Becerra, J. A. Garcia-Charton and C. Marcos (2004). "Composition, structure and distribution of the ichthyoplankton in a Mediterranean coastal lagoon." Journal of Fish Biology **64**: 202-218.

- Poizat, G., E. Rosecchi, P. Chauvelon, P. Contournet and A. J. Crivelli (2004). "Long-term fish and macro-crustacean community variation in a Mediterranean lagoon." Estuarine, Coastal and Shelf Science **59**: 615-624.
- Pollard, D. A. (1994). "A comparison of fish assemblages and fisheries in intermittently open and permanently open coastal lagoons on the south coast of New South Wales, South-Eastern Australia." Estuaries **17**(3): 631-646.
- Pombo, L. and J. E. Rebelo (2002). "Spatial and temporal organization of a coastal lagoon fish community - Ria de Aveiro, Portugal." Cybium **26**(3): 185-196.
- Price, K. S. (1998). "A framework for a Delaware Inland Bays environmental classification." Environmental Monitoring and Assessment **51**: 285-298.
- Pritchard, D. W. (1960). "Salt balance and exchange rate for Chincoteague Bay." Chesapeake Science: 48-57.
- Rice, J. C. (2000). "Evaluating fishery impacts using metrics of community structure." ICES Journal of Marine Science **57**: 682-688.
- Ross, S. W. and S. P. Epperly (1985). Utilization of shallow estuarine nursery areas by fishes in Pamlico Sound and adjacent tributaries. Fish Community Ecology in Estuaries and Coastal Lagoons: Towards an Ecosystem Integration. A. Yanez-Arincibia (ed). Mexico, UNAM Press: 207-232.
- Scharf, F. S., J. A. Buckel and F. Juanes (2002). "Size-dependent vulnerability of juvenile bay anchovy *Anchoa mitchilli* to bluefish predation: Does large body size always provide refuge?" Marine Ecology Progress Series **233**: 241-252.
- Schwartz, F. J. (1961). "Fishes of Chincoteague and Sinepuxent bays." The American Midland Naturalist **65**(2): 384-408.
- Schwartz, F. J. (1964). "Fishes of Isle of Wight and Assawoman bays near Ocean City, Maryland." Chesapeake Science **5**(4): 172-193.

- Serafy, J. E., R. M. Harrell and J. C. Stevenson (1988). "Quantitative sampling of small fishes in dense vegetation: Design and field testing of portable "pop-nets"." Journal of Applied Ichthyology **4**: 149-157.
- Sokal, R. R. and F. J. Rohlf (1995). Biometry. New York, W. H. Freeman and Co.
- Steele, M. A. (1997). "The relative importance of processes affecting recruitment of two temperate reef fishes." Ecology **78**(1): 129-145.
- Stirling, G. and B. Wilsey (2001). "Empirical relationships between species richness, evenness, and proportional diversity." The American Naturalist **158**(3): 286-298.
- Szedlmayer, S. T. and K. W. Able (1996). "Patterns of seasonal availability and habitat use by fishes and decapod crustaceans in a southern New Jersey estuary." Estuaries **19**(3): 697-709.
- Vega-Cendejas, M. E. and M. Hernandez de Santillana (2004). "Fish community structure and dynamics in a coastal hypersaline lagoon: Rio Lagartos, Yucatan, Mexico." Estuarine, Coastal and Shelf Science **60**: 285-299.
- Vernberg, F. J. (1982). "Environmental adaptation to lagoon systems." Oceanologica Acta **1982**: 407-415.
- Wagner, C. M. (1999). "Expression of the estuarine species minimum in littoral fish assemblages of the lower Chesapeake Bay tributaries." Estuaries **22**(2A): 304-312.
- Warburton, K. (1978). "Community structure, abundance and diversity of fish in a Mexican coastal lagoon system." Estuarine and Coastal Marine Science **7**: 497-519.
- Warlen, S. M., K. W. Able and L. E. H. (2002). "Recruitment of larval Atlantic menhaden (*Brevoortia tyrannus*) to North Carolina and New Jersey estuaries: evidence for larval transport northward along the east coast of the United States." Fishery Bulletin **100**(3): 609-623.

- Wazniak, C., M. O. Hall, C. Cain, D. Wilson, R. V. Jesien, J. Thomas, T. Carruthers and W. C. Dennison (2004). State of the Maryland Coastal Bays, Maryland Department of Natural Resources: 44 pp.
- Weinstein, M. P. (1985). Distributional ecology of fishes inhabiting warm-temperate and tropical estuaries: community relationships and implications. Fish community ecology in estuaries and coastal lagoons: Towards an ecosystem integration. A. Yanez-Arancibia (ed), UNAM Press: 285-310.
- Weisberg, S. B., P. Himchak, T. Baum, H. T. Wilson Jr. and R. Allen (1996). "Temporal trends in abundance of fish in the tidal Delaware River." Estuaries **19**(3): 723-729.
- Werner, E. E. and D. J. Hall (1977). "Competition and habitat shift in two sunfishes (Centrarchidae)." Ecology **58**: 869-876.
- Whitfield, A. K. (1993). "Fish biomass estimates from the littoral zone of an estuarine coastal lake." Estuaries **16**(2): 280-289.
- Whitfield, A. K. (1999). "Ichthyofaunal assemblages in estuaries: a South African study." Reviews in Fish Biology and Fisheries **9**(2): 151-186.
- Wootton, R. J. (1999). Ecology of Teleost Fishes. Dordrecht, Kluwer Academic Publishers.
- Yanez-Arancibia, A., Ed. (1985). Fish community ecology in estuaries and coastal lagoons: towards an ecosystem integration. Mexico, UNAM Press.
- Yanez-Arancibia, A., A. L. L. Dominguez and D. Pauly (1994). Coastal lagoons as fish habitats. Coastal Lagoon Processes. B. Kjerfve (ed). Amsterdam, Elsevier Science Publishers B.V.: 363-376.
- York, J. K., Z. Witek, S. Labudda and S. Ochocki (2001). "Comparison of primary production and pelagic community respiration rates in the coastal zone of the Gulf of Gdansk." Oceanologia **43**(3): 365-370.

