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LONG-TERM TRENDS IN WATER QUALITY AND LIVING RESOURCES

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LONG-TERM TRENDS IN THE LOWER CHESAPEAKE BAY (1985-1992): I. WATER QUALITY

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Abstract. A long-term water quality monitoring program has been established in the Virginian waters of the Chesapeake Bay. To date, over 8 years of data have been collected and analyzed to characterize long-term trends in water quality. A series of nonparametric trend analyses were used for determining overall, site-specific, and season-specific long-term trends for water quality variables in the tributaries and Mainstem of the Bay. Particular attention was focused on determining the effect of river flow on the trends, because flow rates in some of the tributaries have changed dramatically since the beginning of the monitoring program. Many of the water quality variables from the tributaries displayed strong relationships to flow. Trends for increasing nutrients could be attributed to flow effects in certain tributary segments, while in others, potential trends for improvement in water quality were obscured by flow. Nonetheless, nonpoint-source controls in the tidal fresh region of the James River apparently produced decreasing nutrient trends ,which persisted for the flow-corrected data set.

Water quality conditions in the Mainstem of the lower Bay were not as greatly affected by flow. The up-Bay segments displayed trends for decreasing inorganic nitrogen and ammonia, while the down-Bay regions were characterized by declining trends for phosphorus. Total organic carbon and suspended solids displayed increasing trends throughout the Virginian waters of the Bay. Dissolved oxygen in bottom waters displayed a downward trend in the regions near the Bay mouth.

The management implications of flow effects on water quality and the use of the statistical flow correction protocol for assessing trends are discussed.

INTRODUCTION

A major focus of the Chesapeake Bay Program has been on the detection of natural and/or anthropogenic changes in environmental conditions over time. The Chesapeake Bay Monitoring Program (CBMP) has created extensive data bases that enable federal and state authorities to identify long-term trends in the environmental quality of the Bay. The ability to detect long-term trends allows environmental managers to determine the effectiveness of management actions and/or to determine which regions of the Bay continue to deteriorate, therefore, warranting greater management attention. Thus, the trend detection capabilities of the CBMP allow scientists and managers to take the pulse of the ecological health of the Bay in order to determine long-term management strategies.

Water quality collections have been made throughout the Bay and its tributaries since June of 1984. Virginia's portion of the CBMP for water quality includes 27 stations in the lower Bay Main stem and 31 stations in the three major tributaries (James, York, and Rappahannock Rivers). Water quality collections have been made at these stations 18-20 times per year: twice per month from March through October and once per month from November through February from 1984 to 1988; and thereafter, twice per month from April through September and once per month October through March. Details of the analytical methods and the collection regime for Virginia's water quality monitoring program have been presented elsewhere (Alden et al., 1989, 1990, 1991, 1992a,1992b). The water quality data were analyzed for trends, power, and robustness following 5 years of collections (Alden et al., 1990b, 1991, 1992a,1992b). The results of the 5-year trend analyses indicated that fallline loadings affected the trends of nutrients throughout the tributaries and that mainstem water quality patterns were dominated by declining trends in phosphorus due to the phosphate ban (Alden et. al. 1991, 1992a,1992b).

River flow is an environmental factor that is not under direct management control (on tributaries without dams), but that could potentially influence water quality and living resources. The flushing time (or the reciprocal, the flushing rate) of a region of an estuary has the strong potential for directly influencing various spatiotemporal ecological patterns (Pilson 1985). Flushing times are affected by river flow measured at the fall-line, runoff patterns in the watershed, as well as the local hydrographic conditions (e.g. a narrow section of an river/estuary would tend to be flushed faster than a wide region, if other factors are the same). Regional hydrographic conditions could be responsible for natural spatial differences in environmental conditions. Likewise, the flow components could have temporal patterns (short and long-term), which could influence the trends in many environmental variables, thus confounding the interpretation of the results of trend analyses by environmental managers. In order to assess trends produced by (or, at least, capable of being affected by) management actions, one must be able to account for natural influences, such as flow/flushing, that are beyond direct human control. Once managers are capable of correcting for these natural factors, they can evaluate "what if" scenarios concerning the potential longterm effectiveness of management actions in the absence of these external influences. Conversely, the managers can begin to understand the direction and magnitude of trends that are associated with factors beyond their control.

The purpose of the present study was to not only update trend results for 8 years of water quality monitoring data from the lower Bay and its tributaries, but to assess the effects of flushing rates on these trends. Trend analyses were run both on raw water quality data (designated hereafter as "uncorrected" data), and on the same data that has been corrected for flow/flushing effects (designated as "corrected" data). By comparing the direction and magnitude of trends observed for each of these data sets, one can assess the relative importance of natural and anthropogenic factors on the long-term temporal patterns of water quality in the lower Bay.

METHODS

Water Quality Data Analyzed

The data sets analyzed were from 8 years of water quality collections made in the lower Chesapeake Bay and its three major tributaries (The James, York, and Rappahannock Rivers). The variables analyzed (and abbreviations used in tables and figures) for the Bay stations included, temperature (TEMP), salinity (SAL), dissolved oxygen (DO), Secchi depth (SECCHI), total suspended solids (TSS), total organic carbon (TOC), dissolved ammonia (NH4), total nitrogen (TN), dissolved inorganic nitrogen (DIN), total phosphorus (TP), dissolved inorganic phosphorus (DIP), silicates (SI), and *chlorophyll a* (CHLA). All variables were analyzed separately for surface (1 m below surface; abbreviations designated with a "S" prefix) and bottom (1 m. above bottom; abbreviations designated with a "B" prefix) depths except for Secchi depth, and for dissolved oxygen, which was analyzed only for the bottom depth. The same variables were analyzed for the data sets from the tributaries, except that only surface *chlorophyll a* measurements are collected in the tributary monitoring program. Details of the collection methods, station locations, and analytical methods are presented elsewhere (Alden et al. 1989, 1990a, 1991, 1992a,1992b)

The trend analyses were run for regions of the Bay or tributaries that have been shown previously to display similarities in water quality patterns. Regions of similar water quality characteristics were identified by a series of multivariate protocols that have been described in detail by Alden et al. (1989, 1990a, 1991, 1992a,b,c). These previous studies have shown that the lower Bay has seven regions (designated MSI-VII; see figure 1a), going from the southern end of the "deep trench" (segment MSI) to the Bay mouth (segments MSV and MSVI) and the mouth of the James River (Segment MSVII). The tributaries displayed regions of similar water quality that roughly correspond to the lower estuarine (mesohaline), riverine/estuarine transitional (oligohaline), and tidal fresh segments of the original Chesapeake Bay Segmentation Scheme (Alden et al. 1992c; see figure 1b): segments JI-JIII for the James River; Segments YI-YIII for the York River; and segments RI and RII for the Rappahannock River (segment RIII, the tidal fresh segment was represented by the single fallline station, which was sampled by another program and the data were not available for the present study).

Figure 1. Station locations and site groups used in trend analyses for (1) the mainstem and (2) tributaries of Chesapeake Bay.

Statistical Analyses

The trends in the raw (uncorrected) water quality data were identified and quantified by three parallel nonparametric trend analysis techniques: the seasonal intra-block sign test from the Kendall Tau test (Seasonal Kendall) described by Hirsch et al. (1982) and adopted for use throughout the Chesapeake Bay Program; the aligned rank test (Sen's Tau) by Sen (1968); and the Van Belle and Hughes (1984) test. The first two tests focus on the detection of overall long-term trends (i.e., across all seasons and stations), while the Van Belle and Hughes method is used to detect long-term trends unique to certain seasons (or collection periods), stations, or the interaction of seasons and stations. Experience has indicated that these three tests are quite powerful, and robust, and that their results almost always agree when overall trends are detected (Alden et al., 1990, 1991, 1992a1992,b). The magnitude of significant trends ($p < 0.01$) were determined by the Seasonal Kendall slope estimator or by season-specific or station-specific modifications of this slope estimator, which have been made to fit the two-way Van Belle and Hughes design (Gilbert 1987). These methods were also used to reanalyze the data after the following flow-correction process.

Flushing time calculations were based on methods presented by (Pilson 1985). A 2-month rolling average for fall line flow measurements was used, where a mean value was determined for flow data from the month of collection and the preceding month. Corrections for tidal input were made using bottom salinity measurements from the nearest downstream station from each station being evaluated for flushing times. Segment volumes were obtained from estimates provided by the summary of Boynton et al. (1990) of the data compiled by Cronin and Pritchard (1975). Data for each of the water quality variables were regressed against the reciprocal of the flushing times (i.e., the flushing rates) of each salinity regime of each tributary using regression models developed by Smith et al. (1982) for the U.S. Geological Survey NASQAN Program. The statistically significant ($p < 0.01$) regression model with the highest R^2 value was employed to correct for flow effects. Trend analyses were conducted on the grand mean-centered residuals from the selected regression models. Thus, trend results were assessed for two sets of water quality data, identified as either uncorrected, or corrected for flow effects.

RESULTS

Relationships to Flow

The results of the flow correction process are summarized in table 1. Nearly 80% (163 out of 207) of all variable-salinity regime combinations displayed significant ($p < 0.01$) relationships to flow. The amount of the total variance "explained" by flow effects (as measured by the R^2 expressed as a percentage) ranged from less than 1% (for many of the variables in the James River oligohaline region) to nearly 55% (SSAL in the mesohaline region of the Rappahannock River). The percentages of variables displaying significant relationships to flow for specific tributary salinity regimes were as follows: 96%, 100%, and 87% in the tidal fresh, oligohaline, and mesohaline regions of the James River; 48%, 78%, and 100% in the tidal fresh, oligohaline, and mesohaline segments of the York River; and 61%, 74%, and 65% of all variables tested were significantly correlated to flow in the tidal fresh, oligohaline, and mesohaline regions of the Rappahannock Rivers, respectively. The James River data displayed the greatest number of relationships to flow, although many of the regression models for the oligohaline data set explained less than 1% of the total variance.

Figure 2 presents representative relationships between water quality variables and flow. Some variables tended to increase with long residence times and decrease with high-flow events (figure 2a and 2b), while others displayed the opposite pattern (figures 2d and 2e). Table 1 presents the direction of the relationship for each variable-salinity regime combination. The following variables were generally observed to have a negative relationship with flow: salinity, temperature, and chlorophyll (except mesohaline regions). The following variables were generally observed to have a positive relationship with flow: dissolved oxygen, silicates, ammonia, and total phosphorus (and, to a lesser extent, dissolved inorganic phosphorus, which had many values below detection limits). Total nitrogen and dissolved inorganic nitrogen concentrations also generally displayed a positive relationship to flow, except in the mesohaline Rappahannock River and in the tidal fresh region of the James River. Total suspended solids and secchi had inverse relationships to each other, both suggesting that suspended solids loads/turbidity increased with flow, except in the mesohaline region of the Rappahannock River and tidal fresh segment of the James River. Regardless of the direction of the relationships, the flow-correction process produced corrected data sets that were centered around the grand mean of the raw data, but which displayed no relationship to flow (figures 2c and 2f).

Flow corrections were also made for data from the lower Bay mainstem using flow data from the U.S. Geological Survey (USGS) for segments described by Bue (1968). The USGS segment C generally corresponded to stations in segments MSI-V and segment D generally corresponded to MSVI-VII. Table 2 presents the results of the flow correction regression models for the lower Bay Mainstem. In general, fewer of the regression models for the variable-segment combinations were significant (70% for segment C and 35% for segment D) than were observed for the tributary segments and the R^2 values were lower (ranging from < 1% to 14%; with most explaining < 2% of the total variance). This pattern is not surprising since flushing times would be expected to be longer in the open Bay than in the tributaries; thus, flow effects would tend to be diminished.

The directions of the relationships between the water quality variables and flow for the Bay mainstem are presented in table 2. The variables which had R^2 values ž 10% for relationships to flow displayed patterns similar to the same variables in the tributaries. Surface and bottom temperature in segment C displayed a negative relationship to flow, while dissolved oxygen displayed a positive relationship to flow. Many of the other variables displayed the opposite relationships to flow in the Bay mainstem compared to those observed in the tributaries, although the low R^2 values for most variables suggest that this inverse pattern may have negligible ecological significance.

Trends in Water Quality

Tables 3 and 4 summarize the trends for the tributary and mainstem data sets, respectively. Figure 3 presents examples of three possible relationships observed for the uncorrected and corrected data when analyzed for trends: 1) some trends persisted following adjustment for flow (figures 3a and 2) some trends disappeared when the data sets were corrected for flow effects (figure 3b and 3) some new trends appeared in the corrected data sets (figure 3c). Figure 4 summarizes the occurrence of these three scenarios in the results of the trend analyses: scenario 1 is represented by filled arrows (displaying direction of

Figure 2. Results of regression analyses for selected parameter/salinity regime combinations: (a) plot of BSAL against oligohaline portion of the Rappahannock River, (b) relationship between BSAL and flushing rate for the oligohaline por River, (c) plot of "flow" corrected residuals of BSAL and flushing rate for the oligohaline portion of the Rappahannoc against residence time for the oligohaline portion of the York River, (e) relationship between SCHLA and flushing rate River, and (f) plot of "flow "corrected residuals of SCHLA against flushing rate for the oligohaline portion of the Yo

Table 1. Summary of the NASQAN regression analysis results for the Virginia tributaries by salinity regime. Direction: + indicates a positive relationship between flow and the variable listed, - indicates a negative relationship.

River	Regime	Parameter	R^2	Direction
James	Tidal Fresh	SCHLA	0.0123	-
James	Tidal Fresh	SDIN	0.0561	-
James	Tidal Fresh	BDIN	0.0724	-
James	Tidal Fresh	SDIP	0.0630	+
James	Tidal Fresh	BDIP	0.0850	+
James	Tidal Fresh	BDO	0.0738	+
James	Tidal Fresh	SNH4	0.0717	+
James	Tidal Fresh	BNH4	0.0861	+
James	Tidal Fresh	SSAL	0.2268	-
James	Tidal Fresh	BSAL	0.2145	-
James	Tidal Fresh	SSI	0.1817	+
James	Tidal Fresh	BSI	0.2148	+
James	Tidal Fresh	STN	0.0609	-
James	Tidal Fresh	BTN	0.0565	-
James	Tidal Fresh	STOC	0.0335	-
James	Tidal Fresh	BTOC	0.0314	-
James	Tidal Fresh	STP	0.0998	+
James	Tidal Fresh	BTP	0.0691	+
James	Tidal Fresh	BTSS	0.0143	-
James	Tidal Fresh	STEMP	0.0267	-
James	Tidal Fresh	BTEMP	0.0204	-
James	Tidal Fresh	SECCHI	0.1252	+
James	Oligohaline	SCHLA	0.0022	-
James	Oligohaline	SDIN	0.0159	+
James	Oligohaline	BDIN	0.0105	+
James	Oligohaline	SDIP	0.0093	+
James	Oligohaline	BDIP	0.0034	+
James	Oligohaline	BDO	0.0437	+
James	Oligohaline	SNH4	0.0090	+
James	Oligohaline	BNH4	0.0035	+
James	Oligohaline	SSAL	0.0123	-
James	Oligohaline	BSAL	0.0046	-
James	Oligohaline	SSI	0.0073	+
James	Oligohaline	BSI	0.0059	+
James	Oligohaline	STN	0.0057	+
James	Oligohaline	BTN	0.0033	+
James	Oligohaline	STOC	0.0060	-
James	Oligohaline	BTOC	0.0066	-
James	Oligohaline	STP	0.0055	+
James	Oligohaline	BTP	0.0048	+
James	Oligohaline	STSS	0.0046	
James	Oligohaline	BTSS	0.0016	
James	Oligohaline	STEMP	0.0043	-
James	Oligohaline	BTEMP	0.0457	-
James	Oligohaline	SECCHI	0.0161	-
James	Mesohaline	SCHLA	0.0207	+
James	Mesohaline	SDIN	0.1015	+
James	Mesohaline	BDIN	0.0928	+
James	Mesohaline	BDO	0.0275	+
James	Mesohaline	SNH4	0.0526	+
James	Mesohaline	BNH4	0.0381	+
James	Mesohaline	SSAL	0.1692	-

Table 1. (continued)

River	Regime	Parameter	R^2	Direction
James	Mesohaline	BSAL 0.2173	-	
James	Mesohaline	SSI 0.1322	+	
James	Mesohaline	BSI 0.1101	+	
James	Mesohaline	STN 0.1422	+	
James	Mesohaline	BTN 0.0680	+	
James	Mesohaline	STOC 0.0392	+	
James	Mesohaline	BTOC 0.0333	+	
James	Mesohaline	STP 0.3491	+	
James	Mesohaline	BTP 0.1879	+	
James	Mesohaline	STSS 0.0128	+	
James	Mesohaline	STEMP	0.0216	-
James	Mesohaline	BTEMP	0.0112	-
James	Mesohaline	SECCHI	0.1143	-
York	Tidal Fresh	SCHLA	0.1212	-
York	Tidal Fresh	SDIN 0.4262	+	
York	Tidal Fresh	BDIN 0.4424	+	
York	Tidal Fresh	SDIP 0.0206	-	
York	Tidal Fresh	BDO 0.0887	+	
York	Tidal Fresh	STN 0.0379	+	
York	Tidal Fresh	STSS 0.0586	+	
York	Tidal Fresh	BTSS 0.0615	+	
York	Tidal Fresh	STEMP	0.0968	-
York	Tidal Fresh	BTEMP	0.0892	-
York	Tidal Fresh	SECCHI	0.1623	+
York	Oligohaline	SCHLA	0.2287	-
York	Oligohaline	SDIN 0.2546	+	
York	Oligohaline	BDIN 0.2798	+	
York	Oligohaline	SDIP 0.1436	-	
York	Oligohaline	BDIP 0.1128	-	
York	Oligohaline	BDO 0.1498	+	
York	Oligohaline	SNH4 0.0280	+	
York	Oligohaline	BNH4 0.0335	+	
York	Oligohaline	SSAL 0.3644	-	
York	Oligohaline	BSAL 0.3489	-	
York	Oligohaline	STN 0.0700	+	
York	Oligohaline	BTN 0.0355	+	
York	Oligohaline	STP 0.0343	+	
York	Oligohaline	STSS 0.0799	+	
York	Oligohaline	BTSS 0.0520	+	
York	Oligohaline	STEMP	0.1482	-
York	Oligohaline	BTEMP	0.1464	-
York	Oligohaline	SECCHI	0.2283	-
York	Mesohaline	SCHLA	0.0411	+
York	Mesohaline	SDIN 0.1695	+	
York	Mesohaline	BDIN 0.0490	+	
York	Mesohaline	SDIP 0.0475	+	
York	Mesohaline	BDIP 0.0222	-	
York	Mesohaline	BDO 0.0451	+	
York	Mesohaline	SNH4 0.0561	+	
York	Mesohaline	BNH4 0.0109	+	
York	Mesohaline	SSAL 0.2352	-	
York	Mesohaline	BSAL 0.2126	-	
York	Mesohaline	SSI 0.1563	+	
York	Mesohaline	BSI 0.1162	+	
York	Mesohaline	STN 0.1067	+	

Table 1. (continued)

River	Regime	Parameter	R²	Direction
York	Mesohaline	BTN 0.0874	+	
York	Mesohaline	STOC 0.0144	+	
York	Mesohaline	BTOC 0.0488	+	
York	Mesohaline	STP 0.4213	+	
York	Mesohaline	BTP 0.3274	+	
York	Mesohaline	STSS 0.0330	+	
York	Mesohaline	BTSS 0.0412	+	
York	Mesohaline	STEMP	0.0383	-
York	Mesohaline	BTEMP	0.0374	-
York	Mesohaline	SECCHI	0.2412	-
Rappahannock	Tidal Fresh	SCHLA	0.2810	-
Rappahannock	Tidal Fresh	SDIN 0.4706	+	
Rappahannock	Tidal Fresh	BDIN 0.4709	+	
Rappahannock	Tidal Fresh	BDIP 0.0477	+	
Rappahannock	Tidal Fresh	SNH4 0.1236	+	
Rappahannock	Tidal Fresh	BNH4 0.1217	+	
Rappahannock	Tidal Fresh	SSAL 0.1593	-	
Rappahannock	Tidal Fresh	BSAL 0.1159	-	
Rappahannock	Tidal Fresh	SSI 0.4275	+	
Rappahannock	Tidal Fresh	STN 0.1643	+	
Rappahannock	Tidal Fresh	BSI 0.4150	+	
Rappahannock	Tidal Fresh	BTN 0.1248	+	
Rappahannock	Tidal Fresh	STEMP	0.1079	-
Rappahannock	Tidal Fresh	BTEMP	0.1037	-
Rappahannock	Oligohaline	SDIN 0.4188	+	
Rappahannock	Oligohaline	BDIN 0.4011	+	
Rappahannock	Oligohaline	BDO 0.1061	+	
Rappahannock	Oligohaline	SNH4 0.1325	+	
Rappahannock	Oligohaline	BNH4 0.0919	+	
Rappahannock	Oligohaline	SSAL 0.5453	-	
Rappahannock	Oligohaline	BSAL 0.3818	-	
Rappahannock	Oligohaline	SSI 0.2776	+	
Rappahannock	Oligohaline	BSI 0.2218	+	
Rappahannock	Oligohaline	STN 0.3665	+	
Rappahannock	Oligohaline	BTN 0.3893	+	
Rappahannock	Oligohaline	BTP 0.1541	+	
Rappahannock	Oligohaline	STSS 0.4091	+	
Rappahannock	Oligohaline	BTSS 0.3387	+	
Rappahannock	Oligohaline	STEMP	0.1415	-
Rappahannock	Oligohaline	BTEMP	0.1395	-
Rappahannock	Oligohaline	SECCHI	0.3116	-
Rappahannock	Mesohaline	SCHLA	0.0094	+
Rappahannock	Mesohaline	SDIN 0.0171	-	
Rappahannock	Mesohaline	BDIN 0.0193	-	
Rappahannock	Mesohaline	BDO 0.0430	+	
Rappahannock	Mesohaline	SSAL 0.0637	-	
Rappahannock	Mesohaline	BSAL 0.0480	-	
Rappahannock	Mesohaline	SSI 0.0721	+	
Rappahannock	Mesohaline	BSI 0.0819	+	
Rappahannock	Mesohaline	STN 0.0308	-	
Rappahannock	Mesohaline	BTN 0.0266	-	
Rappahannock	Mesohaline	STSS 0.0648	-	
Rappahannock	Mesohaline	BTSS 0.0403	-	
Rappahannock	Mesohaline	STEMP	0.0485	-
Rappahannock	Mesohaline	BTEMP	0.0514	-
Rappahannock	Mesohaline	SECCHI	0.1861	+

Table 2. Summary of the NASQAN regression analysis results for mainstem flow segments C and D. Direction: + indicates a positive relationship between flow and the variable listed; - indicates a negative relationship.

Flow segment	Parameter	R^2	Direction
mainstem C	BCHLA	0.0230	+
mainstem C	SDIN	0.0285	+
mainstem C	SDIP	0.0037	-
mainstem C	BDIP	0.0613	-
mainstem C	BDO	0.0951	+
mainstem C	BNH4	0.0427	+
mainstem C	SSAL	0.0097	-
mainstem C	BSAL	0.0116	-
mainstem C	SSI	0.0263	-
mainstem C	BSI	0.0612	-
mainstem C	BTOC	0.0111	+
mainstem C	STP	0.0253	-
mainstem C	BTP	0.0408	-
mainstem C	SECCHI	0.0067	+
mainstem C	STEMP	0.0977	-
mainstem C	BTEMP	0.1021	-
mainstem D	SDIP	0.0184	-
mainstem D	BDIP	0.0199	-
mainstem D	BDO	0.0201	+
mainstem D	BNH4	0.0204	-
mainstem D	SSAL	0.0306	+
mainstem D	BSI	0.0212	-
mainstem D	STEMP	0.0240	-
mainstem D	BTEMP	0.0224	-

trends) in open boxes; scenario 2 is represented by cross-hatched shaded boxes (or lightly shaded boxes if the trend was season-specific); and scenario 3 is represented by white arrows in black boxes.

Among the upward trends in the tributaries (figure 4a), phosphorus (TP, and, to a lesser extent, DIP) was observed to increase in the lower James River (JI; the lower estuarine/mesohaline region), in the transitional (YII) and tidal fresh (YIII) regions of the York River and in the transitional region (RII) of the Rappahannock River. Most of these trends could be attributed to flow patterns, that the TP trends for JI, YII, YIII, and RII (surface) disappeared in the corrected data set (figure 4a). An upward trend for BTN in JI also disappeared in the corrected data set. Surface and bottom concentrations of silicates appeared to be increasing in both the corrected and uncorrected data, except in RI, where the trend for BSI disappeared after flow corrections. Surface chlorophyll concentrations were observed to increase in both the uncorrected and corrected data from nearly all regions of the three tributaries, except in JIII, where the trend appeared only after flow corrections were made. Increasing trends for BDO concentrations from JIII appeared for both uncorrected and corrected data sets. Upward trends in TSS concentrations of bottom waters of the tributaries were observed to disappear in the corrected data set. Increasing water temperatures for the James (JI-JII) and Rappahannock (RI-II) Rivers appeared only in the corrected data set.

The most evident downward trends in the tributaries (figure 4b) were observed for all nitrogen and phosphorus nutrients in the tidal fresh James River (JIII). All of these trends persisted in both the uncorrected and corrected data sets, indicating that these long-term trends are not caused by flow effects. The transitional region of the James (JII) also displayed downward trends in some of the nitrogen nutrients (NH4 and STN), but the results indicated that flow effects may have obscured potential decreases in DIN and SDIP. Likewise, downward trends in certain nutrients in the Rappahannock were only observed in the flow corrected data set: NH4 in RI, and SDIN and NH4

Table 3. Summary of the results of the Seasonal Kendall trend analyses on raw and flow corrected water quality for the Virginia tributary site groups. All results shown were significant at $p <= 0.01$. Slope: - indicates no significant overall trend was present for the parameter/segment combination given.

Site Group	Parameter	1985 Median	Uncorrected Grand Mean Slope	Corrected Grand Mean Slope
JI	SCHLA	3.91	1.7221	1.6696
JI	SDIP	0.02	0.001	0.0008
JI	BDIP	0.02	0.001	-
JI	BDO	6.75	-	-0.075
JI	SSAL	17.93	-0.4871	-
JI	BSAL	20.95	-0.35	-
JI	SSI	1.16	0.1635	0.1461
JI	BSI	0.98	0.1168	0.1058
JI	BTEMP	20.04	-	0.1166
JI	BTN	0.65	0.015	-
JI	STOC	5.5	-0.21	-0.2684
JI	BTOC	6	-0.27	-0.3189
JI	STP	0.1	0.001	-
JI	BTP	0.1	0.001	-
JI	BTSS	82	0.177	-
JII	SCHLA	9.54	2.5319	4.0767
JII	SDIN	0.425	-	-0.0042
JII	BDIN	0.495	-	-0.0067
JII	SDIP	0.02	-	-0.0012
JII	SNH4	0.075	-0.0233	-0.0196
JII	BNH4	0.075	-0.0213	-0.0236
JII	SSI	2.9	0.2882	0.2201
JII	BSI	3.11	0.293	0.2237
JII	STEMP	20.5	-	0.2899
JII	BTEMP	21	-	0.2886
JII	STN	1.0625	-0.035	-0.0272
JII	STOC	6.5	-0.265	-0.2558
JII	BTOC	6.5	-0.2417	-0.2267
JII	BTP	0.115	0.005	0.003
JII	STSS	662	1	-
JII	BTSS	280	0.293	-
JIII	SCHLA	4.77	1.4204	-
JIII	SDIN	0.83	-0.064	-0.0657
JIII	BDIN	0.85	-0.0538	-0.0532
JIII	SDIP	0.155	-0.025	-0.0254
JIII	BDIP	0.15	-0.0225	-0.0226
JIII	BDO	7.75	0.1813	0.1826
JIII	SNH4	0.35	-0.0672	-0.0679
JIII	BNH4	0.35	-0.0613	-0.0603
JIII	SSI	4.02	0.4208	0.4136
JIII	BSI	4.11	0.4523	0.4598
JIII	STN	1.2	-0.075	-0.0742
JIII	BTN	1.11	-0.08	-0.0801
JIII	STOC	5	-0.2767	-0.2396
JIII	BTOC	5	-0.3033	-0.262
JIII	STP	0.2	-0.02	-0.0199
JIII	BTP	0.21	-0.0206	-0.0206
YI	SCHLA	6.74	1.4681	1.7526
YI	BSAL	22.21	-0.3282	-
YI	SSI	1.03	0.102	0.1064

Table 3. (continued)

Site Group	Parameter	1985 Median	Uncorrected Grand Mean Slope	Corrected Grand Mean Slope
YI	BSI	0.98	0.0597	0.0605
YII	SCHLA	8.09	3.2178	3.2675
YII	SDIP	0.015	0.0015	0.0017
YII	BDIP	0.02	-	0.0017
YII	SSAL	10.89	-	-0.461
YII	BSAL	14.04	-0.55	-0.6305
YII	SSI	2.96	0.3307	0.3382
YII	BSI	2.76	0.3271	0.335
YII	STP	0.1	0.001	-
YII	BTP	0.1	0.001	-
YIII	SCHLA	7.06	0.3421	0.7416
YIII	SSI	4.35	0.514	0.5061
YIII	BSI	4.86	0.5178	0.514
YIII	STOC	8	-0.17	-0.17
YIII	STP	0.1	0.001	-
YIII	BTP	0.1	0.001	-
YIII	BTSS	44	0.518	-
RI	SCHLA	8.93	1.0093	0.842
RI	BDO	6.35	-	-0.0999
RI	SNH4	0.05	-	-0.0015
RI	BNH4	0.05	-	-0.0016
RI	SSAL	15.62	-0.3492	-0.2488
RI	BSAL	16.7	-0.306	-0.2252
RI	SSI	1.47	0.1455	0.1314
RI	BSI	1.47	0.0935	-
RI	STEMP	20.22	-	0.1941
RI	STOC	5	-0.2667	-0.298
RI	BTOC	5.5	-0.29	-0.0003
RI	BTSS	-	0.093	-
RII	SCHLA	9.61	1.6356	3.7306
RII	SDIN	0.33	-	-0.0035
RII	SNH4	0.05	-	-0.0135
RII	SSAL	1.46	0.001	-
RII	BSAL	2.46	0.001	-
RII	SSI	3.29	0.3271	0.1903
RII	BSI	3.67	0.3875	0.2709
RII	STEMP	21.07	-	0.2698
RII	BTEMP	20.76	-	0.2692
RII	STOC	5	-0.1933	-0.206
RII	BTOC	5	-0.215	-
RII	STP	0.1	0.001	-
RII	BTP	0.1	0.0033	0.0043

Table 4. Summary of the results of the Seasonal Kendall trend analyses conducted on raw and flow corrected water quality for the Chesapeake Bay mainstem site groups. All results shown were significant at p < = 0.01. Slope: - indicates no significant overall or season specific trend were present for the parameter/site group combination given.

Site Group	Parameter	1985 Median	Uncorrected Grand Mean Slope	Corrected Grand Mean Slope
MSI	SCHLA	6.210	-	-0.367
MSI	SDIN	0.1001	-0.005	-0.005
MSI	BDIN	0.1243	-0.006	-0.005
MSI	SDIP	0.0100	<-0.001	<-0.001
MSI	BDIP	0.0100	<-0.001	-
MSI	SNH4	0.0210	-0.002	-0.002
MSI	BNH4	0.0845	-0.003	-0.003
MSI	BSAL	21.54	-	0.119
MSI	BTOC	3.302	0.157	0.16
MSI	STP	0.023	0.169	0.139
MSI	BTSS	13.625	-	0.877
MSII	SDIN	0.0790	-0.007	-0.008
MSII	BDIN	0.0895	-0.007	-0.007
MSII	SDIP	0.0100	<-0.001	<-0.001
MSII	BDIP	0.0100	<-0.001	<-0.001
MSII	SNH4	0.0230	-0.002	-0.003
MSII	BNH4	0.0350	-0.002	-0.002
MSII	SSAL	18.05	-0.162	-
MSII	BSAL	18.94	-0.111	-
MSII	STOC	3.889	0.216	0.196
MSII	BTOC	3.302	0.223	0.221
MSII	STP	0.023	<0.001	-
MSII	STSS	10.00	0.65	0.599
MSII	BTSS	16.50	0.514	-
MSII	BTEMP	17.75	0.092	-
MSIII	SDIN	0.0595	-0.005	-0.005
MSIII	BDIN	0.0883	-0.006	-0.005
MSIII	SDIP	0.0100	<-0.001	<-0.001
MSIII	BDO	7.58	-	-0.073
MSIII	SNH4	0.0210	-0.002	-0.002
MSIII	BNH4	0.0485	-0.002	-0.002
MSIII	SSAL	20.24	-0.12	-
MSIII	STOC	3.462	0.194	0.177
MSIII	BTOC	3.166	0.192	0.183
MSIII	STP	0.0230	<0.001	<0.001
MSIII	BTP	0.0425	<-0.001	<-0.001
MSIII	STSS	9.70	0.467	-
MSIII	STEMP	18.25	0.092	0.176
MSIII	BTEMP	17.30	-	0.157
MSIV	SCHLA	6.13	-	0.324
MSIV	BCHLA	6.36	-	0.362
MSIV	SDIN	0.0450	-0.004	-0.005
MSIV	BDIN	0.0603	-0.005	-0.006
MSIV	SDIP	0.0100	-	<-0.001
MSIV	BDIP	0.0100	<-0.001	<-0.001
MSIV	SECCHI	1.50	-	-0.04
MSIV	SNH4	0.0200	-0.002	-0.002
MSIV	BNH4	0.0220	-0.002	-0.004

Table 4. (continued)

Site Group	Parameter	1985 Median	Uncorrected Grand Mean Slope	Corrected Grand Mean Slope
MSIV	SSAL	22.26	-0.222	-0.417
MSIV	BSAL	22.37	-0.229	-0.42
MSIV	BTN	0.4860	-	0.008
MSIV	STOC	3.5265	0.232	0.266
MSIV	BTOC	3.4085	0.245	0.284
MSIV	STSS	12.00	0.814	1
MSIV	BTSS	16.00	0.621	-
MSIV	BTEMP	18.47	-	-0.04
MSV	SDIN	0.0283	<-0.001	<-0.001
MSV	BDIN	0.0458	<-0.001	<-0.001
MSV	SDIP	0.0140	<-0.001	<-0.001
MSV	BDIP	0.0200	<-0.001	<-0.001
MSV	BDO	7.48	-	-0.114
MSV	SECCHI	2.15	-0.05	-0.047
MSV	SNH4	0.0106	<-0.001	<-0.001
MSV	BNH4	0.0172	<-0.001	-
MSV	BTN	0.3085	-0.006	-0.007
MSV	STOC	2.7937	0.114	0.11
MSV	BTOC	2.3354	0.102	0.101
MSV	STP	0.0466	-0.002	-0.002
MSV	BTP	0.0590	-0.003	-0.003
MSV	STSS	5.565	0.275	0.275
MSV	BTSS	9.085	0.939	0.942
MSVI	BDIN	0.0268	<0.001	<0.001
MSVI	SDIP	0.0118	<-0.001	<-0.001
MSVI	BDIP	0.0185	<-0.001	<-0.001
MSVI	BDO	7.96	-	-0.132
MSVI	STOC	2.3334	0.071	0.072
MSVI	BTOC	2.1625	0.115	0.113
MSVI	STP	0.0498	-0.002	-0.002
MSVI	BTP	0.0573	-0.002	-0.002
MSVI	STSS	6.0375	0.248	0.277
MSVI	BTSS	9.7842	0.95	0.836
MSVII	SDIP	0.0300	-0.001	-0.001
MSVII	BDIP	0.0230	-0.001	-0.001
MSVII	BDO	7.00	-	-0.132
MSVII	SNH4	0.0382	-	-0.002
MSVII	STOC	3.3432	0.11	-
MSVII	BTOC	2.8309	0.169	-
MSVII	STP	0.0743	-0.003	<-0.001
MSVII	BTP	0.0898	-0.005	<-0.001

Figure 3. Plots of concentration versus date and grand mean slope trend lines for selected parameter/site group combinations: (a) downward tend for uncorrected STN in site group JIII, (b) downward trend for corrected STN in site group JIII, (c) upward trend for uncorrected BTN in site group JI, (d) corrected BTN in site group JI, (e) uncorrected SSAL in site group MSII and (f) downward trend for corrected SSAL in site group MSII. Vertical bars represent on standard error of the mean for each collection period.

Figure 4. Summary of the trend analyses on uncorrected and corrected water quality data: (a) mainstem increasing trends, (b) mainstem decreasing trends, (c) Tributary increasing trends, and (d) Tributary decreasing trends. A solid arrow indicates a significant trend occurred before and after flow correction. A blank arrow on a black box indicates a trend appeared only after flow correction. A cross-hatched box indicates a trend that disappeared after flow correction. Letters indicate significant trends for one or more seasons (W = winter, Sp = spring, Su = summer, F = fall). Letters on a shaded background indicate the trend appeared only after flow correction.

in RII. Total organic carbon concentrations decreased in the James and Rappahannock Rivers, persisting in both the uncorrected and corrected data sets for both STOC and BTOC. Decreasing salinities appeared to be related to flow in the JI and YI regions, in that these trends disappeared in the corrected data set. However, the declining trends in salinities for YII and RI persisted in the corrected data set. Decreasing trends for BDO from RI were only observed in the corrected data set, suggesting that a potential long-term pattern of decline had been obscured by flow effects.

The predominant upward trends in the Bay mainstem (figure 4c) were observed for TSS and, particularly TOC, although some of these trends disappeared in the corrected data set. Potential upward trends in TSS were observed for MSI only in the corrected data set. Likewise, a potential upward trend for bottom salinity from this segment was observed only in the corrected data set. Upward trends in water temperatures in region MSIII persisted in the flow-corrected data, but disappeared in the flow-corrected data from MSII.

The most obvious of all trends for the Bay mainstem were the long-term decreases in nutrients (figure 4d): DIN and NH4 decreased throughout most of the segments (MSI-V), and phosphorus (TP and DIP) decreased in many of the down-Bay segments (MSV-VII, and, to a lesser extent, MSIII-IV). Most of these trends persisted for the corrected data. However, some potential decreasing trends may have been obscured by flow effects; declining trends for Secchi depth for MSIV and decreasing SNH4 concentrations for MSVII were observed only in the corrected data set. Likewise, potential decreasing trends for chlorophyll concentrations from MSVII were only observed in the corrected data set. Bottom dissolved oxygen concentrations decreased in the segments near the Bay mouth (MSV, and MSVI). A potential downward trend in BDO was apparently obscured by flow effects in the segment at the mouth of the James River (MSVII). Downward trends in salinities observed in segments MSII, and MSIII were apparently associated with flow patterns, in that they disappeared following flow correction. The declining salinities observed for MSIV persisted for the corrected data.

DISCUSSION

Effects of Flow on Water Quality

The significance of flow effects on water quality conditions became obvious during the course of this study. Nearly 80% of the water quality variables collected in the tributaries and 65% of the variables from the Bay mainstem displayed significant relationships to flow. These relationships explain up to 50% of the total variation in the 8-year data set for some of the variables in the tributaries. On the other hand, the amounts of variation attributed to flow for variables in the Bay mainstem tended to be quite low. This contrast is not too surprising owing to the longer flushing times in the wide, deep waters of the lower Bay, a characteristic that may buffer against major flow effects. It is interesting to note that the James River, which has a flow greater than the York and Rappahannock Rivers combined, had the greatest percentage of variables displaying significant relationships to flow.

The direction of most of the flow effects in the tributary water quality variables are understandable. Variables displaying a negative relationship to flow include salinity, temperature, and chlorophyll (except in mesohaline regions). Salinity would be expected to display a negative relationship to flow owing to the dilution effects of greater freshwater runoff. Temperatures could have a negative relationship owing to the increased introduction of colder runoff water during high flow conditions, to the seasonality of high-flow events tending to take place in the colder months (late winter/early spring and, occasionally, fall) or both. The patterns of decreasing chlorophyll concentrations with increased flow could be associated with a “flushing out” effect; faster flow produces such short residence times that the resident phytoplankton may not be capable of producing substantial growth before the populations are washed downstream. On the other hand, when residence times are prolonged, the phytoplankton may be capable of producing dense blooms while in a river segment. In the mesohaline reaches, where the pattern reverses itself, the tributaries tend to widen and the relative flushing rates slow, so populations may be able to keep up with the lessened flushing effect, especially when stimulated by the greater nutrient loads that are often associated with high-flow events (see below). In some cases, the downstream displacement of oligohaline phytoplankton blooms during higher-flow events may also increase chlorophyll concentrations in the mesohaline region.

The variables tending to display a positive relationship with flow included BDO, SI, NH4, TN, and TP. This relationship for BDO could be attributable to the re-aeration effect of increased surface runoff and the associated turbulence, and/or to the seasonality of flow events that may be associated with cooler water, which can hold higher concentrations of dissolved oxygen. The increasing loads of most of the nutrients with increasing flow rates could be associated with greater inputs from stormwater runoff. On the other hand, the higher levels of chlorophyll concentrations associated with long residence times may suggest that nutrients may tend to be depleted by dense phytoplankton populations during low-flow events. While both of these processes could be responsible for the observed relationships, the latter cannot explain why the chlorophyll/flow relationships reverse themselves in the mesohaline regions, but those for most of the nutrients do not (i.e., TN, NH4, DIN, TP, and SI continue to display positive relationships with flow in the mesohaline regions of the York and James Rivers).

The effects of flow on water clarity were generally as expected: suspended solids loads tended to increase with flow, while Secchi depth tended to decrease. Increased erosion and runoff input of suspended materials would be expected to be associated with high-flow events. The major exception to this pattern was observed for the mesohaline region of the Rappahannock River. The hydrography of this segment is different from those of the corresponding regions of the York and James Rivers, in that there is a shallow sill across the mouth of the Rappahannock. This feature may restrict flow through the mesohaline segment until high-flow conditions flush it out. Most water quality variables collected from this region displayed relationships to flow that were opposite from those observed from the other mesohaline regions, perhaps owing to the effects of its unique hydrography.

Long-Term Trends in Water Quality

A discussion of the results of this study must focus on the integration of two topics: the water quality trends actually observed in the lower Bay and its tributaries, and the effects of flow on the trends. In the tributaries, the upward trends for nutrients and chlorophyll concentrations noted for the 5-year data set (Alden et al. 1991) continued for many of the segments of the three rivers, albeit at more moderate rates of increase than observed previously. At least half of the increasing trends for nutrients appear to be associated with flow effects, in that they disappeared in the corrected data set. Likewise, most of the river segments displayed increasing trends for suspended solids loads, that were attributable to flow effects. It should be noted that two of the three rivers increased substantially in median flow rates during the second half of the study period. Based upon monthly averages, median flow in the James River went from 5,185 cu.ft./sec. from 1985 through 1988 to a median flow of 8,133 cu.ft./sec from 1989 through 1992 (a 57% increase), while the median flow rates for the Rappahannock River increased from 1,140 cu.ft./sec to 1,690 cu.ft./sec for the same two periods (a 55% increase). Considering these directional temporal patterns, it is not too surprising that flow effects produce significant trends for some of the variable-segment combinations.

The distinct downward trends for nitrogen, and phosphorus-based nutrients in segment JIII were not associated with flow effects. Rather, these trends, which were observed in both uncorrected and corrected data sets, appear to be associated with local point source controls. Alden et al. (1992a,1992b) demonstrated statistically that point-source controls in this region would have produced decreasing trends in nutrients in the 5-year data set, if not for opposing trends in increasing nutrient loads coming from above the fall line. After 8 years of monitoring, the effectiveness of the point-source controls is evident, despite the influence of any flow effects.

Flow effects are also apparent in the form of potential long-term trends that have been obscured in the raw data. For example, downward trends in various nutrients may have been observed in segments JII, RI and RII, if flow effects had not overwhelmed them. Thus, potentially declining trends in nutrients, which may have otherwise been attributed to management actions, are hidden from the monitoring program.

It is not too surprising that the effects of flow on trends are much less apparent in the mainstem data sets. While some trends associated with TSS (upward), salinity (downward), and temperature (downward) can be associated with flow effects, most of the trends persist in both the uncorrected and corrected data sets. Ammonia and DIN tended to decrease in the up-Bay segments of the lower Bay, while DIP and TP tended to decrease in the segments nearer to the mouth of the Bay. Unfortunately, neither TN nor chlorophyll concentrations displayed overall downward trends for either set of segments.

Total organic carbon increased throughout all segments. For the most part, the TOC trends could not be attributed to flow effects. Alden et al. (1992c) speculate that these upward trends in TOC could be a real, widespread phenomenon or could reflect the effects of methodological changes in laboratory analyses. This issue should be explored further, in that such an overall trend could have significant ecological implications, as well as potentially affecting modeling efforts, if it is real. Also to be noted are downward trends in BDO in two of the lower Bay segments, possibly associated with the increased carbon loadings.

Management Implications

The effects of flow on long-term trends in water quality that have been observed during this study can have important management implications. Trends of degrading water quality conditions in some of the regions of the tributaries can be attributed to flow alone. Obviously, such trends would tend to frustrate managers examining monitoring data for indications of the success of management actions. The protocol for conducting trend analysis on uncorrected and corrected data would allow such situations to be identified. Likewise, the protocol would allow managers to observe potential trends that may have been realized, if flow had not been a confounding factor. On the other hand, the process also would allow managers to validate trends that persist in the corrected data in order to demonstrate that the trends in water quality are really attributable to factors such as management actions (or, alternatively, due to anthropogenic damage) and not due to flow patterns alone. Because flow is a factor beyond the direct control of most management actions (at least in the tributaries of the lower Bay), it is important to be able to account for and filter out these effects in order to focus on trends that are associated with either management actions, or with anthropogenic degradation of the Bay.

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LONG TERM TRENDS IN THE LOWER CHESAPEAKE BAY (1985-1992): II. PHYTOPLANKTON

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Abstract. Long-term trends in phytoplankton communities of the lower Chesapeake Bay were identified using flow-corrected and uncorrected data sets, with 22 of the 23 significant trends being found to be similar for both data sets. The trends observed for the 7 1/2 year study period starting in 1985 were for: (1) overall reduced phytoplankton concentrations throughout the water column; (2) decreasing numbers of taxa during spring, summer, and fall months; and (3) seasonally mixed trends for diatom abundance in waters below the pycnocline, with spring months displaying decreasing densities over the study period and exhibiting increasing trends in November and December. In addition, there were overall decreasing trends within the tidal James, York, and Rappahannock Rivers for autotrophic picoplankton concentrations in data corrected for flow, suggesting that this component of the phytoplankton community would have declined, if the effects of river flow patterns had not overwhelmed these trends.

INTRODUCTION

Phytoplankton populations have been monitored in the lower Chesapeake Bay since July 1985 as part of the Chesapeake Bay Plankton Monitoring Program. Composite water samples are taken above and below the pycnocline during monthly collections at seven stations in the lower Chesapeake Bay and in a companion study of six tidal stations in the James, York, and Rappahannock Rivers. The samples are analyzed for the composition, biovolume, and abundance of the phytoplankton, autotrophic picoplankton abundance, ^{14}C productivity rates, and a variety of water quality parameters. The methodology and earlier results of this program have been reported by Marshall and Alden (1990a, 1990b), Marshall and Nesius (1994), Alden et al. (1992), among others. These studies have indicated a diverse and productive flora, rich in diatoms and phytoflagellates, with specific spatial and temporal assemblages, and seasonal patterns of succession.

Phytoplankton trends in the lower Chesapeake Bay have been described for the period between 1985 and 1990 by Marshall and Alden (1991). They noted modest but significant seasonal trends of increased abundance above the pycnocline and a decrease below the pycnocline. When these data sets were combined, these trends did not appear for the entire water column. The abundance increase above the pycnocline was most developed in July and associated with higher concentrations of phytoflagellates and small centric diatoms. The reduced abundance below the pycnocline was greatest during mid-winter (January). In addition, the trend for phytoplankton concentrations in the tidal York, James, and Rappahannock Rivers (during 1986-89) indicated a significant increase above and below the pycnocline, being least developed in tidal fresh waters, and most developed downstream in March, April, July, and November. At the same time, there were trends for increased levels of total nitrogen and total phosphorus in these rivers.

Since the monitoring began in 1985, it has been apparent that a common fluctuating event associated with the Bay and its rivers has been the changing pattern of water flow in this system. Owing to differences in the width and depth of various segments of this estuary, fluctuations in flow have been common. In addition, the average amount of flow was influenced by the timing of the seasonal rains and the passage of the spring freshet, both of which varied in their occurrence year to year.

Similar differences of flow rate and residence time within regions of Narragansett Bay were reported by Pilson (1985). He indicated flushing time was an important ecological variable to any biota present. To separate the influence of water flow from other environmental factors in this study, a flow corrected data set was established for comparison to raw data that were not corrected for flow effects.

METHODS

Phytoplankton Indices Analyzed

Two individual station sets of phytoplankton data were examined. These were based on composite water samples taken from above and below the pycnocline at each station (Marshall and Alden 1990b). There were six phytoplankton indices analyzed in this study. These were (1) total phytoplankton abundance (excluding picoplankton concentrations), (2) autotrophic picoplankton abundance, (3) abundance of dinoflagellates, (4) abundance of diatoms, (5) total phytoplankton and picoplankton biovolume, and (6) the number of phytoplankton taxa per sample. The abundance data were recorded as numbers of cells per liter, with biovolume based on cubic microns per liter as determined for each species.

The distinction made between total phytoplankton and autotrophic picoplankton was by cell size, with the later composed of cells < 2.0 microns and identified with epifluorescence microscopy. These indices were selected as representing those components and variables in the phytoplankton community that would likely be influenced by changing trophic and water quality conditions over time.

Statistical Methods

The data set covers the 7 1/2 year period from July 1985 through December 1992. The initial trend analysis of the data was based on the seasonal intrablock sign test from the Kendall Tau statistic (Seasonal Kendall) described by Hirsch et al. (1982) and the aligned rank test (Sen's Tau) by Sen (1968). A chi square protocol by Van Belle and Hughes (1984) was used to analyze trends unique to certain seasons, stations, and the interaction of seasons and stations. The median slopes of significant trends ($p < 0.01$) were determined by the Seasonal Kendall slope estimator (Gilbert 1987). This analysis gave results uncorrected for flow. These data were then reanalyzed following the flow-correction process.

Flushing time calculations were based on methods presented by Pilson (1985). A 2-month rolling average for fall line flow measurements was used, where a mean value was determined for flow data from the month of collection and the preceding month. Each of the phytoplankton indices were regressed against the reciprocal of the flushing times (i.e., the flushing rates) of each salinity regime of each tributary using regression models developed by Smith et al. (1982) for the U.S. Geological Survey NASQAN Program. The statistically significant ($p < 0.01$) regression model with the highest R^2 value was employed to correct for flow effects. Trend analyses were conducted on the grand mean-centered residuals from the selected regression models. Thus, trend results were available for two sets of phytoplankton data, identified as either uncorrected or corrected for flow effects.

RESULTS

Trends Uncorrected for Flow

There were 23 significant ($p < 0.01$) long-term trends identified for the period of study. These trends were observed across all regions of the lower Bay (i.e., no regional or station-specific trends were observed). The trends were decreasing total phytoplankton abundance, both above and below the pycnocline; seasonally mixed trends in diatom abundance below the pycnocline (decreasing for the three spring months, and increasing for November and December); and declining diversity (numbers of taxa per sample) above and below the pycnocline during most of the period from early spring through fall (March-October).

No trends in phytoplankton were identified in the rivers sampled during this period.

Trends Corrected for Flow

There were 22 significant trends for the flow corrected data for the lower Chesapeake Bay. As with the uncorrected data, these trends were found across all stations. While the magnitude of some of the trend rates changed (see below), the direction for all trends for the corrected data was the same as observed for the uncorrected data.

Autotrophic picoplankton abundance declined above and below the pycnocline in the flow corrected data from the James, York, and Rappahannock Rivers.

Total Phytoplankton Trends

In the flow-corrected and flow-uncorrected data sets, there were two similar trends of decreasing phytoplankton abundance, both above and below the pycnocline (figure 1 and 2). The long-term annual rate of decrease (trend slope) above the pycnocline intensified slightly from -1.9 x 10^5 cells l^1 yr^1 to -2.2 x 10^5 l^1 yr^1 when flow effects were taken into account. Likewise, the rate of decrease for phytoplankton below the pycnocline intensified from -2.0 x 10^5 cells l^1 yr^1 for uncorrected data to -2.3 x 10^5 l^1 yr^1 for corrected data. Thus, flow patterns in the mainstem Bay appeared to have slightly influenced the decreasing trends for phytoplankton that would have otherwise occurred. Of course, both the uncorrected and corrected data sets displayed long-term changes that were within the same order of magnitude, so these differences may be negligible ecologically.

Figures 1 and 2 also show seasonal variations in abundance and annual differences in the occurrence and magnitude of phytoplankton populations. Various seasonal combinations for abundance maxima appear. These include the occurrence of spring-summer, spring-fall, spring-summer-fall, summer-fall, summer, and spring maxima during different years of the study. Corresponding phytoplankton assemblages often differed in composition, yet, specific species remained numerically dominant during many of these periods,(e.g., *Skeletonema costatum*). In the uncorrected data set above the pycnocline, the abundance ranged from winter lows to spring peaks of 2.0 to 15.8 x 10^6 cells l^1, respectively. In the flow-corrected data, these concentrations ranged from 1.9 to 16.6 x 10^6 cells l^1.

Below the pycnocline, there were large spring blooms from 1986 to 1988, and these exceeded concentrations above the pycnocline in 1986 and 1988. The spring bloom was much less developed the following four years (1989-92), with greater counts occurring above the pycnocline, and usually higher concentrations associated with the flow-corrected data. In general, there was close similarity in the seasonal patterns and abundance levels determined for total phytoplankton abundance in both the corrected and uncorrected data sets. Abundance levels ranged from winter lows to spring peaks of 1.0 to 26.2 x 10^6 cells l^1 in the uncorrected data, and 0.9 to 25.9 x 10^6 cells l^1 for corrected data.

Autotrophic Picoplankton Trends

The data for autotrophic picoplankton are based on a smaller number of collections than the other phytoplankton data, in that this component of the program was not added until September 1989. In the flow-corrected data, there were trends of declining picoplankton abundance, above (-8.3 x 10^6 cells l^1 yr^1) and below (-7.2 x 10^6 cells l^1 yr^1) the pycnocline, in the tidal James, York, and Rappahannock Rivers (figure 3). These trends did not appear in the raw, uncorrected data or in the Bay data sets. Larger concentrations of picoplankton were associated throughout the water column during the summer months, with greater abundance in waters above the pycnocline (figure 3).

Diatom Abundance Trends

The concentrations of diatoms showed five significant trends below the pycnocline in both the corrected and uncorrected data sets. No trends appeared above the pycnocline. There were decreasing long-term trends of diatom abundance during March, April, and May. An example of a typical pattern for these months is given for April in figure 4. The abundance range and degree of trend expression is similar for both data sets. The second trend within the diatoms is for increasing abundance in November and December, and it also has similar patterns for the corrected and uncorrected data (figure 5).

Trends for Number of Taxa

A characteristic pattern of decreasing diversity (numbers of taxa per sample) was evidenced by negative slopes for 15 of 16 significant trends observed for phytoplankton data from above and below the pycnocline. Both the uncorrected and corrected flow data from above the pycnocline had this negative trend from March through June, plus August and September, with an October downward trend also occurring in the uncorrected set. Both data sets had similar trends below the pycnocline. This included positive trends in February and negative trends from March through October. An example of these trends is given in figure 6 for the month of April. In all these trends, the flow-corrected data show less of a loss in taxa over time.

Figure 1. Results of trend analysis for phytoplankton abundance (excluding the picoplankton), uncorrected and corrected for flow, above the pycnocline in the lower Chesapeake Bay.

Figure 2. Results of trend analysis for phytoplankton abundance (excluding the picoplankton), uncorrected and corrected for flow, below the pycnocline in the lower Chesapeake Bay.

Figure 3. Results of trend analysis for autotrophic picoplankton abundance that is corrected for flow, above and below the pycnocline, for tidal stations in the James, York, and Rappahannock Rivers.

Figure 4. Results of trend analysis for diatom abundance in April, uncorrected and corrected for flow, below the pycnocline in the lower Chesapeake Bay.

Figure 5. Results of trend analysis for diatom abundance in December uncorrected and corrected for flow, below the pycnocline in the lower Chesapeake Bay.

Figure 6. Results of trend analysis for number of taxa per sample, uncorrected and corrected for flow, below the pycnocline in the lower Chesapeake Bay.

DISCUSSION

Of the 23 trends identified for the uncorrected data sets from the lower Chesapeake Bay, 22 similar trends were found in the flow-corrected data. The exception was an October trend of reduced taxa number above the pycnocline that appeared only in the original data set. Overall, both data sets identified long-term trends of decreasing phytoplankton abundance and diversity throughout the water column. This reduction in phytoplankton abundance establishes a more specific trend in the lower Bay than the rather static condition previously described by Marshall and Alden (1991), where mixed results were noted above and below the pycnocline. These results support the value of long term data sets for trend analysis evaluations over shorter study periods. Another example of the value of more extended studies to balance fluctuating values in the system are the results obtained from the tributary data. In the earlier trend report the total phytoplankton abundance was increasing in the tidal James, York and Rappahannock Rivers. In the present study, that trend has apparently been neutralized and no longer exists. While this original trend has ended, a new trend has appeared in the corrected data set that indicates there would have been an overall decline of the autotrophic picoplankters in these rivers if flow patterns had not been a factor. These current results indicate a general curtailment of long-term increases in phytoplankton and picoplankton growth in the system, and are considered responses to practices reducing nutrient accumulation, or to factors producing less-favorable growing conditions for specific flora in the system.

Associated with these reduced concentrations of plankton flora is a similar reduction of total phosphorus (TP), dissolved inorganic phosphorus (DIP), ammonia, and dissolved inorganic nitrogen (DIN) in data uncorrected for flow in the lower Bay (Alden, et al. 1994). The seven phytoplankton sampling sites correspond to five segments of the lower Bay where water quality trend measurements have also been made. In all five segments, there are declining trends for reduced TP, with DIN and DIP decreasing in four of the five segments, and ammonia concentrations declining in two of the segments. Variables showing positive trends at this time were total suspended solids (TSS) and total organic carbon, increasing in four and five of the five segments respectively. The decreasing nutrient levels may have produced limiting conditions that reduced the potential for long-term increases in phytoplankton and picoplankton communities in the system. This situation, plus increases in the total suspended solids loads, which could have reduced light entry into the water column, would be negative factors in phytoplankton growth.

Both the corrected and uncorrected data sets identify diatom abundance decreasing in mid to late spring, and an increasing trend in November and December. The declining spring trend occurs during a transitional time period during which the typical spring bloom, usually peaking in February or March, are dominated by *S. costatum*, and are replaced by larger, less numerous species (e.g., *Cerataulina pelagica*). The overall declining trend at this time of year suggests that concentrations of cells associated with prime conditions for growth during the spring maximum are being reduced earlier, and the successional patterns are being initiated earlier, and even influencing the magnitude of diatom populations during late fall and early winter. Coupled with the decreasing trend of TP, DIP, DIN, and ammonia, there may exist limitation by these nutrients associated with the declining spring blooms that were dominated by diatoms, in that there was no pattern of silica reduction in the system.

A declining number of taxa per sample characterized the water column assemblages from spring through fall. The magnitude of this decline was greater in the uncorrected data set compared to figures corrected for flow. In either case, the diversity of species decreased over this time period. It may be theorized that during the early stages of this study, water quality conditions favored a more diverse development of species, exposed to a wider range of growth conditions that favored greater diversity. Over the past 7 1/2 years, there has been a reduction in certain nutrients, with an increase in TSS. Conditions of declining nutrients and light entering the water owing to added TSS would be expected to reduce the presence or growth of certain species.

In the lower Chesapeake Bay, these results indicate a close similarity in the trends identified for phytoplankton abundance, diatom abundance, and number of taxa per sample using values that were corrected and uncorrected for flow. Data sets that were most influenced by flow were the picoplankton concentrations trends within the tidal rivers. These trends were not identified in the uncorrected data set. The future use of flow-corrected data would apparently provide a more accurate evaluation of this particular community in assessing the potential trends that would have been evident if flow had not been a factor. Because flow is a factor beyond the direct control of most management actions, it is important to be able to account for and filter out these effects in order to focus on trends that are associated with either management actions or anthropogenic degradation of the Bay.

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Toward a Sustainable Coastal Watershed:
The Chesapeake Experiment. Proceedings of a Conference
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LONG-TERM TRENDS IN THE LOWER CHESAPEAKE BAY (1985-1992):
III. THE HYDRAULIC EFFECTS OF FLOW ON ZOOPLANKTON POPULATIONS

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Abstract: Plankton populations at most tidal freshwater monitoring stations and some downstream stations in the Chesapeake Bay tributaries are periodically subjected to hydraulic impact, as high flow events cause the rapid downstream transport of planktonic organisms. Hydraulic impact events occur regularly in some tributaries, usually at the time of the spring freshet, and vary in magnitude with flow rate, cross-sectional configuration at the station location, and the degree of upstream salt-wedge intrusion. Both temporal and spatial comparisons of plankton community metrics require that hydraulic effects be taken into account. Using the James and York Rivers as a case study, we here report on an approach to removing the effect of flow (or, conversely, flushing time) on plankton monitoring data.

Toward a Sustainable Coastal Watershed:
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LONG-TERM TRENDS IN THE MACROBENTHOS OF THE LOWER CHESAPEAKE BAY (1985-1992)

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Abstract. Long-term trends in macrobenthic communities of the lower Chesapeake Bay, were examined using data collected quarterly (March, June, September, and December) from 1985 through 1992 at 16 stations along a salinity gradient from tidal freshwater regions of the major tributaries (the James, York and Rappahannock Rivers) to the polyhaline region of the mainstem of Chesapeake Bay. In March 1989, two stations were added to the program in the mesohaline region of the Southern Branch of the Elizabeth River. A nonparametric trend analysis procedure was applied to five parameters characterizing macrobenthic community structure: community biomass, species richness, abundance of individuals, proportion of biomass composed of opportunistic species (opportunistic biomass composition), and proportion of biomass composed of equilibrium species (equilibrium biomass composition). A total of 51 trends were detected. No trends were detected for any benthic parameters in the Elizabeth River for the 1989-92 period. For community biomass, 6 trends were significant; all had positive slopes and at least one biomass trend occurred in each of the major tributaries and the mainstem of the Bay. For species richness, 10 trends were significant; all had positive slopes with 3 trends in the James River, 2 trends in the York River, 2 trends in the Rappahannock River, and 3 trends in the mainstem of Chesapeake Bay. For abundance of individuals, 25 trends were detected; all trends were seasonally dependent, had positive slopes, and occurred at 13 of the 16 stations sampled since 1985. For opportunistic biomass composition, 6 trends were significant; 4 had positive slopes with 1 trend in the Rappahannock River and 3 trends in the mainstem of the Bay; 2 trends had negative slopes and occurred in the tidal freshwater stations of the James and York Rivers. For equilibrium biomass composition 4 trends were significant; 2 trends had positive slopes, one in the James River and one in the Rappahannock River and 2 trends had negative slopes, both in the mainstem of the Bay. Trends in the three major tributaries (James, York, and Rappahannock Rivers) were considered to indicate improving conditions for the benthos, while trends in the mainstem of Chesapeake Bay were considered to indicate deteriorating conditions. Deteriorating conditions for the benthos were associated with regions exposed to summer low dissolved oxygen events.

INTRODUCTION

Long-term monitoring of macrobenthic communities of lower Chesapeake Bay has been conducted quarterly (March, June, September, and December) since March 1985 at 16 stations stratified by salinity-sedimentary regions. In March 1989, two stations were added in the Southern Branch of the Elizabeth River. Parameters associated with macrobenthic community structure, water quality and sediment particle distribution are measured along the estuarine gradient from tidal freshwater to polyhaline regions within the major tributaries (the James, York, and Rappahannock Rivers), the Southern Branch of the Elizabeth River, and the mainstem of lower Chesapeake Bay. Data reported in this study were collected as part of the ongoing Benthic Biological Monitoring Program of the Chesapeake Bay Program for the Virginia portion of Chesapeake Bay. The primary purposes of the benthic monitoring program are (1) to characterize the present health of regional areas of the lower Chesapeake Bay as indicated by the structure of the benthic communities (Dauer 1993a), and (2) to conduct trend analyses on long-term data to relate spatial and temporal trends of the benthic communities to changes in water and sediment quality within lower Chesapeake Bay (Dauer 1991, 1993b).

In this study a nonparametric trend analysis procedure was applied to the benthic data set for the 8 year period from March 1985 to December 1992 for the original 16 stations and for the 4-year period from March 1989 to December 1992 for the 2 stations in the Southern Branch of the Elizabeth River. Five parameters characterizing macrobenthic community structure: community biomass, species richness, abundance of individuals, proportion of biomass composed of opportunistic species (opportunistic biomass composition), and proportion of biomass composed of equilibrium species (equilibrium biomass composition). The selection of these parameters was based upon the assumption that healthy benthic communities can be characterized by (1) high biomass estimates dominated by relatively long-lived and often deep-dwelling species and (2) high species richness (Dauer 1993a). Based upon this assumption, positive slopes in all parameters, except opportunistic biomass composition, would indicate improving conditions. A negative slope for opportunistic biomass would indicate improving conditions.

Environmental conditions in Chesapeake Bay and its tributaries have deteriorated significantly over the past 50 years, attributed primarily to increases in eutrophication and toxic substances. Eutrophication can promote low dissolved oxygen events (Taft et al. 1980, Officer et al. 1984, Kuo and Neilson 1987, Smith et al. 1992), which affect recruitment, growth, and survivorship of the benthos. Low dissolved oxygen effects on benthic communities of Chesapeake Bay have previously been reported by Holland et al. (1977, 1987), Pihl et al. (1991), Dauer et al. (1992), and Dauer (1993a). Toxic substances, primarily from industrial and municipal point sources, become particle-bound and eventually concentrated in fine-grained sediments (Swartz and Lee 1980). Low dissolved oxygen events and high concentrations of toxic materials in sediments in Chesapeake Bay result in reduced levels of benthic community parameters (Dauer 1993a, Dauer et al. 1993).

METHODS

Station Locations

Sixteen stations in lower Chesapeake Bay have been sampled since March 1985 as part of the Benthic Biological Monitoring Program of the Chesapeake Bay Program. Stations were located within the mainstem of the Bay and the major tributaries-the James, York, and Rappahannock Rivers (figure 1). In the tributaries, stations were located within the tidal freshwater zone (TF5.5, TF4.2, TF3.3), turbidity maximum (transitional) zone (RET5.2, RET4.3, RET3.1), lower estuarine mesohaline muds (LE5.2, LE4.1, LE3.2), and lower estuarine polyhaline siltysands (LE5.4, LE4.3). The tidal freshwater station within the York River estuary was located in the Pamunkey River. In the mainstem of the Bay, three stations were located off the mouths of the major tributaries (CB8.1, CB6.4, CB6.1) and two stations in the deeper channels near the Bay mouth (CB7.3E) and above the Rappahannock River near the Virginia-Maryland border (CB5.4). In March 1989, two stations were added in the Southern Branch of the Elizabeth River (SBE2, SBE5). Table 1 provides average salinity, water depth, and sedimentary parameters for each of the eighteen stations of this study.

Data Collection

On each collection date, three replicate box core samples were collected for benthic community analysis. Each replicate had a surface area of 184 cm^2, and a minimum depth of penetration to 25 cm, was sieved on a 0.5 mm screen, relaxed in dilute isopropyl alcohol, and preserved with a buffered formalin-rose bengal solution. In the laboratory, each replicate was sorted and all the individuals were identified to the lowest possible taxon and enumerated. Biomass was estimated for each taxon as ash-free dry weight (AFDW) by drying to constant weight at 60 ^{o}C and ashing at 550 ^{o}C for 4 hours. Biomass was expressed as the difference between the dry and ashed weight.

At each station on each collection date: (1) a 50 g subsample of the surface sediment was taken for sediment analysis; (2) bottom salinity, temperature and dissolved oxygen were measured; and (3) water depth was recorded. See Dauer et al. (1992) for a summary of the pattern of bottom oxygen values at each station and Dauer et al. (1993) for a summary of the distribution of contaminants in the sediments.

Trend Analysis

Long-term trends in the benthic data set were analyzed by a series of powerful nonparametric trend tests. Overall trends in the data were analyzed by the seasonal intra-block sign test based on the Kendall Tau statistic described by

Figure 1. Lower Chesapeake Bay showing station locations.

Table 1. Station parameters. Data are means for data collected from March 1985 through December 1989 (n=60). Values for SBE2 and SBE5 are for 1989 only (n=12). Salinity is in parts per thousand, water depth in meters, grain size in phi units and silt-clay is a percentage by weight.

			Grain	Silt-Clay	
Stations	Coordinates	Salinity	Depth	Size	Content
James River					
TF5.5	37½18’46" N, 77½13’59" W	0.35	9.70	4.61	61.24
RET5.2	37½12’38" N, 76½47’35" W	2.68	8.20	6.43	80.70
LE5.2	37½03’30" N, 76½35’36" W	13.40	7.70	5.32	57.91
LE5.4	36½57’05" N, 76½23’19" W	20.93	7.90	2.44	10.50
Elizabeth River					
SBE2	36½48’45" N, 76½17’27" W	19.85	12.50	7.29	90.71
SBE5	36½45’54" N, 76½18’00" W	17.38	8.50	7.37	96.55
York River					
TF4.2	37½32’49" N, 76½58’29" W	0.33	9.65	5.48	55.05
RET4.3	37½30’49" N, 76½47’20" W	12.03	6.70	6.69	55.05
LE4.1	37½25’06" N, 76½41’36" W	17.03	8.55	4.68	50.62
LE4.3	37½14’27" N, 76½29’06" W	21.35	7.75	3.99	37.99
Rappahannock River					
TF3.3	38½01’08" N, 76½54’37" W	2.48	6.15	6.53	72.03
RET3.1	37½55’12" N, 76½49’18" W	8.18	5.65	7.56	87.59
LE3.2	37½40’13" N, 76½33’16" W	17.20	12.30	7.39	90.94
Mainstem of Bay					
CB5.4	37½47’28" N, 76½10’33" W	20.59	33.00	6.85	89.17
CB6.1	37½35’18" N, 76½09’45" W	21.38	12.65	5.79	84.67
CB6.4	37½14’11" N, 76½12’18" W	23.69	11.90	4.51	53.45
CB7.3E	37½13’29" N, 76½03’19" W	27.69	21.35	3.17	19.73
CB8.1	36½59’07" N, 76½10’07" W	26.58	9.25	4.09	38.97

Hirsch et al. (1982) and the aligned rank test described by Sen (1968). Trends unique to certain seasons, to certain stations, or to the interaction of stations and seasons were analyzed by a chi-square protocol described by Van Belle and Hughes (1984). The median slopes of significant trends were determined by the Seasonal Kendall slope estimator (Gilbert 1987). A recent study on representative data sets from the Chesapeake Bay monitoring program has indicated that these tests are generally quite powerful and robust, even when data violate most of the assumptions of parametric statistics (Alden et al. 1990). Trends were considered significant if $p < 0.01$.

Benthic Community Parameters

Nonparametric trend analysis was performed on five benthic community parameters: community biomass, species richness, abundance of individuals, proportion of biomass composed of opportunistic species (opportunistic biomass composition), and proportion of biomass composed of equilibrium species (equilibrium biomass composition). All parameters were calculated on a per replicate basis. Community biomass included ash-free dry weights for all species. Species richness was calculated as the mean number of species per replicate. Abundance of individuals was the mean number of individuals per replicate. The percentage of community biomass accounted for by two indicator species groups was also used to characterize the benthic communities. The two indicator species groups were (1) an opportunistic species group, and (2) an equilibrium species group. The opportunistic species group consisted of relatively short-lived, eurytopic species often characterized as dominating disturbed or stressed habitats and the equilibrium species groups consisted of relatively long-lived species that dominate the community biomass in undisturbed or unstressed habitats. See Dauer (1991, 1993a) and Dauer et al. (1992) for a list of the species included in the two groups. For this study the opportunistic species group was expanded to include all insect larvae and oligochaetes.

RESULTS

Table 2 summarizes the distribution of the 51 significant trends detected in this study. All parameters were homogeneous over collection times, except for abundance of individuals, therefore, for abundance of individuals, all trends were examined by each of the four collection times (March, June, September, and December). There were 25 significant trends for abundance of individuals; all trends had positive slopes, and the trends were widespread in occurrence in each tributary and the mainstem of Chesapeake Bay, occurring at 13 of the 16 stations.

In the James River, 14 long-term trends were significant, and all trends had positive slopes, except for a single negative slope for opportunistic biomass at TF5.5. In the York River, 12 long-term trends were significant, and all trends had positive slopes, except for a single negative slope for oppor-tunistic biomass at TF4.2. In the Rappahannock River, 10 long-term trends were significant, and all trends had positive slopes. In the mainstem of Chesapeake Bay, 15 trends were significant: opportunistic biomass composition had a positive slope at three stations, equilibrium biomass composition had a negative slope at two stations, species richness had a positive slope at three stations, community biomass had a positive slope at one station, and abundance had a positive slope at four stations.

Figure 2 is presented as representative of trends at a station (LE4.3) consistent with the expectation of improving conditions for the benthos. At this station, there were trends of increasing community biomass (figure 2a), increasing species richness (figure 2b), and the abundance of individuals increased in three of the four collection seasons (March, June, and December). The increase in community biomass corresponded with increases in the bivalves *Mercenaria mercenaria*, *Mya arenaria*, *Anadara ovalis*, and *Anadara transversa*, and the polychaete *Clymenella torquata*. All of these species have been previously classified as equilibrium species (Dauer 1991, 1993a; Dauer et al. 1992).

Figure 3 is presented as representative of trends at a station (CB6.4) consistent with the expectation of deteriorating conditions for the benthos - increasing composition of opportunistic species (figure 3a) and decreasing composition of equilibrium species (figure 3b).

DISCUSSION

Interpretation of the ecological significance of the trends observed in this study is dependent upon the expected relationship between benthic community structure and levels of eutrophication and/or sediment contamination. Highly stressed marine and estuarine macrobenthic communities are characterized by (1) low levels of species diversity (or species richness), abundance (number of individuals), and biomass; (2) dominance by species that are short-lived (opportunistic, pioneering, *r*-selected, stress tolerant), shallow-dwelling, and primarily annelids; and (3) the absence or rarity of species that are long-lived (equilibrium, *K*-selected), often deep-dwelling within the sediment, and representative of a diversity of major taxa (Boesch 1977, McCall 1977, Pearson and Rosenberg 1978, Rhoads et al. 1978, Gray 1979, Rhoads and Boyer 1982, Warwick 1986, Dauer 1993a). However, the relationship between benthic community structure and intermediate levels of stress from eutrophication and/or contamination of sediments is often difficult to detect or interpret for eutrophication or not well understood for sediment contamination (Scott 1989).

The most widely accepted relationship between benthic community structure and level of eutrophication (or organic enrichment) is the SAB model of Pearson and Rosenberg (1978) that summarizes the relationship between patterns of species richness (S), abundance of individuals (A), and biomass (B) to organic enrichment. Benthic community parameters of the SAB model increase from low levels near the source of organic enrichment to higher levels at a distance, in space or time, from the source of organic enrichment. However, intermediate or moderate levels of organic enrichment may result in benthic communities with values for species richness, abundance of individuals, and biomass that are higher than values for benthic communities exposed to both high and low levels of organic enrichment (Pearson and Rosenberg 1978, Dauer and Conner 1980, Ferraro et al. 1991, Fallesen 1992). Therefore, trends in species richness, abundance of individuals, and biomass must be cautiously interpreted.

Table 2. Trend analysis summary (1985-1992). Slope trends: + and - indicate a significant positive slope or significant negative slope respectively, for the parameter indicated ($p < 0.01$). Abundance had significant station-season interaction. Letter in parentheses indicates the season for which a significant trend was found (M = March, J = June, S = September, D = December). Slopes for community biomass are in g/m^2, for Species Richness in species per replicate, for opportunistic and equilibrium biomass in percent, and for abundance in individuals per m^2.

Stations	Community Biomass	Species Richness	Opportunistic Biomass	Equilibrium Biomass	Abundance (individuals)
James River					
TF5.5	+ 0.171	+ 0.80	- 2.6		+ 989 (M) + 891 (J) + 1,373 (S) + 559 (D)
RET5.2		+ 0.25			+ 842 (M)
LE5.2	+ 1.030			+ 1.9	
LE5.4		+ 1.33			+ 1,467 (M) + 1,014 (D)
York River					
TF4.2	+ 0.057	+ 0.20	- 4.8		+ 193 (S)
RET4.3					+ 567 (D)
LE4.1					+ 898 (M) + 629 (D)
LE4.3	+ 2.003	+ 2.33			+ 1,553 (M) + 813 (J) + 1,032 (D)
Rappahannock River					
TF3.3		+ 0.50	+ 0.1		+ 719 (J) + 322 (S)
RET3.1	+ 1.545	+ 0.50		+ 2.0	+ 583 (M) + 429 (J) + 432 (S)
LE3.2					
Mainstem of Bay					
CB5.4			+ 3.6		+ 894 (M)
CB6.1	+ 0.400	+ 0.50			+ 400 (M)
CB6.4			+ 1.6	_5.6	
CB7.3E		+ 1.00			+ 1,709 (M) + 744 (S) + 930 (D)
CB8.1		+ 1.16	+ 0.4	- 3.3	+ 823 (J)

Figure 2. Trends representative of improving conditions for the benthos at station LE4.3 of the York River. (a) Biomass in g m^{-2}. (b) Species richness in species per replicate.

Figure 3. Trends representative of deteriorating conditions for the benthos at station CB6.4 of the mainstem of the Bay. (percentage of community biomass composed of opportunistic species): (a) equilibrium biomass as a percentage of community biomass; (b) opportunistic biomass as a percentage of community biomass.

Highly stressed benthic communities are dominated in biomass and abundance by opportunistic species, while unstressed benthic communities are dominated in biomass, but not abundance, by equilibrium species (Pearson and Rosenberg 1978, Warwick 1986). Consistent with these approaches, trends in opportunistic species and equilibrium species composition of the benthic community are considered to be the best indicators of the ecological significance of the trends observed in this study. Improving conditions for the benthos should be accompanied by either a positive slope in equilibrium biomass composition or a negative slope for opportunistic biomass composition. Deteriorating conditions for the benthos should be accompanied by either a negative slope in equilibrium biomass composition or a positive slope for opportunistic biomass composition.

Based upon the expectations discussed above, the data in table 2 indicate improving conditions for the benthos of the James, York, and Rappahannock Rivers, and a mixture of improving and deteriorating conditions for the benthos of the mainstem of Chesapeake Bay. There were no trends in the James and York Rivers indicative of deteriorating conditions, while there was a single trend in the Rappahannock River indicative of deteriorating conditions (the increase in opportunistic biomass composition at TF3.3). The increase in opportunistic species composition at three stations and the decrease in equilibrium species composition at two stations in the mainstem of Chesapeake Bay indicate that conditions for the benthos deteriorated, probably associated with low dissolved oxygen events previously reported for stations CB5.4 and CB6.1 (Dauer et al. 1992). Although no trends were detected for station LE3.2 in the lower Rappahannock River or for the two stations in the Southern Branch of the Elizabeth River (SBE2, SBE5), stations have benthic communities that are highly stressed owing to low dissolved oxygen stress (LE3.2, see Dauer et al. 1992, Dauer 1993b) or to sediment contaminants (SBE2,

Table 3. Comparison of trend analyses for the five year period from 1985-1989—designated as 1989, the seven year period from 1985-1991—designated as 1991, and the eight year period 1985-1992—designated as 1992. Numbers indicate the total number of trends in each tributary for each benthic parameter. +, _ : indicates respectively a significant positive or negative slope for the parameter indicated (p - 0.01). For the five year period there were significant station- season interactions for species richness, opportunistic biomass and abundance, for the seven and eight year periods only abundance had a significant stations-season interaction.

Tributary	Community Biomass	Species Richness	Opportunistic Biomass	Equilibrium Biomass	Abundance
James River					
1989	1 (+)	1 (+)	1(-)		1 (+)
1991	3 (+)	3 (+)	1(-)	1 (+)	5 (+)
1992	2 (+)	3 (+)	1(-)	1 (+)	7 (+)
York River					
1989	3 (+)	3 (+)	3(-)	1 (+)	3 (+)
1991	2 (+)	2 (+)	1(-)	1 (+)	5 (+)
1992	2 (+)	2 (+)	1(-)		7 (+)
Rappahannock River					
1989	1 (+)				
1991			1 (+)	1(-)	3 (+)
1992	1 (+)	2 (+)	1 (+)	1 (+)	5 (+)
Mainstem of Bay					
1989	1(-)	1(-)	2 (+)	2(-)	1 (+)
1991		1 (+)	3 (+)	1(-)	4 (+)
1992	2 (+)	3 (+)	3 (+)	1(-)	6 (+)

SBE5, see Dauer 1993a, Dauer et al. 1993).

Comparison with Previous Trend Analyses for the Lower Chesapeake Bay

Previously, Dauer (1991) reported trends in the macrobenthic communities of Chesapeake Bay for the 5-year period from 1985 to 1989, and Dauer (1993b) reported trends for the 7-year period from 1985 to 1991. Table 3 summarizes the comparisons between the previous two reports of trends and this study. The total number of significant trends increased from 25 (though 1989) to 36 (through 1991) to 51 in this study. The conclusions of this study of improving conditions for the benthos of the James and York rivers, and deteriorating conditions for the benthos of the mainstem of Chesapeake Bay, are consistent with the results of Dauer (1991) and Dauer (1993b). For the James and York Rivers, trends in community biomass, species richness, equilibrium biomass and abundance had positive slopes while opportunistic biomass, had negative slopes in each analysis. For the Rappahannock River and the mainstem of the Bay there was a mixture or trends indicative both improving and deteriorating conditions. Trends indicative of deteriorating conditions were associated with stations affected by summer low dissolved oxygen events.

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TRENDS IN PHOSPHORUS, NITROGEN, AND DISSOLVED OXYGEN IN THE CHESAPEAKE BAY (1984-92)

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Abstract: A trend analysis of the first 8 years of water quality data for the Chesapeake mainstem collected by the Chesapeake Bay Program's monitoring program has found a significant decline in phosphorus concentrations in Chesapeake waters, but no change in nitrogen concentrations or the amount of oxygen in the water. Water clarity, as measured by Secchi depth, improved in the upper portions of the mainstem Bay. The analysis did not include tidal or nontidal tributaries.

The findings seem to verify estimates of nutrient loads by the states and the Chesapeake Bay Program that indicate more progress has been made in controlling phosphorus than nitrogen. The report found that phosphorus concentrations in the mainstem Bay had declined 16% over the 8-year period. For nitrogen and dissolved oxygen, the 8 years of data show no trend.

The phosphorus numbers are substantiated by estimates of reduced phosphorus loads from a computer model. The model, which estimates load reductions based on implementation of improvements at sewage treatment plants, use of runoff control practices on farmland, and other factors, has estimated that total phosphorus loads to the Bay dropped from 27 million pounds a year in 1985 to 22.8 million pounds in 1992. Total nitrogen loads decreased slightly from 376 million pounds a year in 1985 to 364 million pounds in 1992.

The lack of significant nitrogen reductions stems largely from the fact that major efforts to control nitrogen throughout the Bay are behind phosphorus control efforts. The lack of improved dissolved oxygen conditions could be attributable to several factors, including the need for nitrogen control and climatic conditions that determine nutrient loads and stratification.

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THIRTY-YEAR ANALYSIS OF EUTROPHICATION IN CHESAPEAKE BAY

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Abstract: A decades-long assemblage of dissolved oxygen and chlorophyll observations exists for Chesapeake Bay. Widely varying interpretations of this data set have been published. The varying interpretations are not likely to be reconciled through exclusive examination of the data. We have taken a different approach to detection and analysis of long-term eutrophication trends. A mechanistic model of the Bay, incorporating hydrodynamics, water column eutrophication processes, and sediment diagenesis, was applied to the period 1959-1988. The model was validated with intensive data collected during 1984-86 and with the historic data. The 30-year simulation allowed examination of dissolved oxygen and chlorophyll without complications inherent in the assembled observations. The model also provided insight into trends in substances, (e.g., nutrients), for which no long-term record existed. We have determined that the volume of anoxic water in the Bay was far more extensive in the decade 1969-79 than in preceding or succeeding decades. The anoxic volume corresponded to hydrology. The wettest decade had the highest anoxic volume. The primary cause of the enhanced anoxia was runoff-induced density stratification, however, rather than nutrient loading. Results indicate total phosphorus concentration was maximum circa 1972 and has declined continuously since. The decline is attributable to control actions and hydrologic conditions. Total nitrogen concentration is slowly increasing, primarily caused by concentration trends at major Fall Lines.

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WATER QUALITY TRENDS OBSERVED IN FOUR MAJOR RIVERS DISCHARGING TO CHESAPEAKE BAY, 1978-1993

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Abstract. The Maryland Department of the Environment and the U.S. Geological Survey have been monitoring four major rivers discharging to Chesapeake Bay since 1984. River discharge, nitrogen, phosphorus, and sediment concentrations are monitored under baseflow and storm- flow conditions on the Susquehanna, Potomac, Patuxent, and Choptank Rivers, which together contribute over 80% of the river flow to the Maryland portion of the Bay. The monitoring is conducted to provide the data needed to document nutrient and sediment loading to the Bay from these rivers and to track changes in these parameters that may occur as a result of implementation of the Chesapeake Bay Nutrient Reduction Strategy. These data, in combination with comparable historical data collected at these sites since 1978, have been used to calibrate a seven parameter log-linear regression model to estimate nitrogen, phosphorus, and sediment concentrations and loads for each of the monitoring stations. Time trends in the nutrient and sediment data and the flow adjusted residuals of the regression model were evaluated for the period 1978 through 1993 using the Seasonal Kendall trend test, linear regression, and the LOWESS data smoothing algorithm. Results indicate that the rate of increase in nitrogen and phosphorus concentrations is decreasing on all four rivers, and in some cases decreasing trends have been observed since 1984.

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HISTORICAL WATER QUALITY AND HABITAT TRENDS IN THE TIDAL POTOMAC ESTUARY

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Abstract. The tidal portion of the Potomac River Estuary in the vicinity of Washington, D.C. was considered grossly polluted during the late 1960s and early 1970s. Low dissolved oxygen and severe algal blooms were characteristic of the area. As a result of these conditions, aggressive nutrient pollution abatement programs were implemented to reduce elevated point-source loadings from the Washington metropolitan area during the 1970s and the 1980s.

This paper updates previous water and habitat quality trend analyses through 1993 in tidal areas of the Potomac. The estuary is divided into three sections for analysis: tidal fresh, transition, and lower estuarine. Total phosphorus, dissolved inorganic phosphorus, total nitrogen, dissolved inorganic nitrogen, bottom dissolved oxygen, chlorophyll, and Secchi depth are examined using Seasonal Kendall trend analyses in each area.

Results indicate that concentrations of nitrogen have increased slightly in the tidal fresh area and exhibit no trend in the other areas since the early 1970's. Point source loads of nitrogen have remained constant over this time period, even though wastewater flows have increased significantly. Phosphorus concentrations have decreased in the tidal fresh and transition zones corresponding to a greater than 95% reduction in wastewater treatment plant phosphorus loads. The submerged aquatic vegetation (SAV) habitat goals have not been attained in any of the three areas, even though there has been a dramatic resurgence in SAV since 1982.

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WATER AND HABITAT QUALITY TRENDS IN THE PATUXENT RIVER ESTUARY

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Abstract: Since 1970, the population in the Patuxent watershed has increased approximately 87% and projected growth to the year 2000 is only a little lower (54%). This rapid population growth has led to large increases in nutrient loads to the estuary. In response to the severe eutrophication impacts, Maryland created the Patuxent Nutrient Control Strategy in 1982. The strategy established specific nutrient load reduction targets that were designed to improve the habitat quality of the estuary. The strategy also required the establishment of a comprehensive water quality monitoring program to track the progress of nutrient controls. In the 1980s, stringent phosphorus controls were implemented at wastewater treatment plants (WWTPs) in the upper estuary. In combination with a state-wide ban on phosphates in detergent, this action has resulted in a 75% reduction of point source phosphorus loads to the estuary. In the tidal fresh region, concentrations have declined approximately 50-75% while, in the oligohaline region concentrations have declined approximately 25% in response to nutrient load reductions. Significant nitrogen controls were not implemented in the Patuxent Estuary until the early 1990s. Despite increases in WWTP effluent flows, nitrogen controls have reduced point source nitrogen loads by 35% annually. Water column concentrations of nitrogen appear to have declined in the tidal fresh region of the estuary in response to reduced point-source loadings. However, it is too early to detect any significant changes in the more saline regions of the estuary. No clear trends in *chlorophyll a*, secchi depth, and dissolved oxygen have been found in our preliminary analysis.

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TRENDS IN MEASURES OF BENTHIC COMMUNITY HEALTH IN THE PATUXENT RIVER

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Abstract: Management actions during the 1980s to reduce nutrient loadings to the Patuxent River watershed proved effective in reducing phosphorus and nitrogen concentrations in the nontidal and upper tidal portions of the river. To ascertain whether the condition of the benthos changed concurrently with reductions in these nutrients, several measures of benthic community "health" were tested for temporal trends. Benthic data collected for the State of Maryland's Water Quality Monitoring Program since 1984 in the tidal portion of the river, and since 1976 In the nontidal portion of the river, were used in the analysis. Comparable historic data collected in the tidal portion of the river in the 1970s and early 1980s were also included. The Van Belle and Hughes nonparametric statistical technique was used to test for trends. In the nontidal portion of the Patuxent River, improving trends were detected in several measures of benthic condition. In the tidal portion of the river, most improving trends were detected in the low mesohaline portion of the river. In the lower portion of the river, where nutrient concentrations have not decreased, no improving trends were detected.

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TEMPORAL TRENDS OF NEAR-COASTAL WATER QUALITY IN THE MID-ATLANTIC BIGHT

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Abstract. Over the past 25 years the near-coastal watersheds have seen dramatic increases in population with commensurate increases in inputs of nutrients. By the year 2010, the projected number of people living in the coastal watersheds will increase by 60%. The 1976 anoxic catastrophe in the mid-Atlantic indicated the potential of nutrient overenrichment causing major environmental problems. Since that time, the National Oceanic and Atmospheric Administration, through its Northeast Monitoring Program (NOAA/NEMP), and U.S. Environmental Protection Agency, Region Three, through its Middle Atlantic Coastal Eutrophication Study (MACES), have conducted long-term monitoring of coastal water quality. We have recently completed analysis of these extensive data sets. The purpose of this study is to evaluate the trends in nutrient inputs into coastal waters along the mid-Atlantic coast over a 10-year period. At issue is the success that has been made in water quality in the rivers and estuaries and whether those successes were felt in the coastal waters. Part of our study focuses on the influences of Chesapeake and the Delaware Bays on the coastal waters. To that end, cluster analyses of a variety of parameters were used to distinguish different water bodies along the mid-Atlantic. In addition, a 10-year data set was analyzed using parametric and nonparametric statistical methods to detect and describe trends in surface water quality.

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HISTORICAL LAND USE CHANGES AND PALEOECOLOGICAL INDICATORS IN THE CHESAPEAKE BAY

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Abstract: Stratigraphic records preserved in the sediments of the mesohaline Chesapeake Bay were used to reconstruct a 2,000-year history of sedimentation, eutrophication, anoxia, and diatom community structure over time. Diatoms, pollen, total and organic carbon (TOC), nitrogen, sulfur, acid-soluble iron, an estimate of the degree of pyritization of iron (DOP), and biogenic silica (BSi) were used as paleoecological indicators in four cores collected from a transect across the Bay from the Choptank River to Plum Point, Maryland. The cores were collected in areas with different patterns of hypoxic and anoxic bottom waters. The sediments were dated using radiocarbon and pollen techniques. Sedimentation rates were determined, and a chronology for each core compiled. More than 400 diatom species, primarily marine and estuarine taxa, were identified in the sediments. The distributional patterns of diatom species in terms of relative abundance of species, community diversity, centric/pennate ratios, and cluster analysis of diatom communities, are related to changing land use patterns of the watershed over time. Analysis of the data indicates that sedimentation rates, eutrophication, and turbidity have increased in Chesapeake Bay since the time of European settlement of the watershed. There is also evidence of increasing anoxia, and more freshwater input to the Bay, especially in the last 50 years.

INTRODUCTION

The stratigraphic record embedded in the sediments of the mesohaline Chesapeake Bay was used in this study to compare water quality conditions prior to European settlement (when the ecology of the Bay was governed primarily by climate) with conditions following settlement (when human activity began to affect silt and nutrient levels). This research included the study of diatom fossils, pollen, biogenic silica (BSi), total and organic carbon (TOC), nitrogen, total sulfur, acid-soluble iron, and an estimate of the degree of pyritization of iron (DOP) found in the sediments of the Chesapeake Bay. Each indicator adds information about the past environment and its spatial heterogeneity, and how the environment has changed through time.

Diatoms constitute 80-90% of the algal abundance during the spring biomass maximum within Chesapeake Bay (Sellner and Brownlee 1988), and account for approximately 50% of annual phytoplankton production (D'Elia et al. 1983). Diatoms grow a silica shell (frustules), with species-specific morphology, that remains as fossil evidence in the stratigraphic record. Different species have distinct ecological tolerances, and a substantial amount of autecological information is available. The fossil record of diatoms in sediments has been used to reconstruct the history of water quality in many lakes and estuaries (Dixit et al. 1992).

In Chesapeake Bay, previous studies and taxonomic lists of diatoms are primarily of the phytoplankton (Wolfe et al. 1926, Morse 1947, Griffith 1961, Mulford 1962, Patten et al. 1963, Marshall 1984, 1986). Wilderman (1984) has presented one of the few ecological studies of both planktonic and benthic diatoms, for the Severn River estuary of the Chesapeake Bay system. In this current paleoecological study, diatoms were counted and identified for the purpose of describing the composition and distribution of communities, both spatially and temporally, within the mesohaline Chesapeake Bay in relation to historical changes in land use and water quality within the Chesapeake Bay watershed. References for this work include Cooper and Brush (1991), Cooper and Brush (1993), Cooper (1993), and Cooper (1995).

METHODS

The four sediment cores used for this study were collected in a transect across the mesohaline section of Chesapeake Bay from the Choptank River to Plum Point, Maryland (figure 1). The cores were collected using a gravity corer in water 9.1-15.2 m in depth, and ranged in length from 114 cm to 160 cm with a diameter of 5.7 cm. In general, the sediments consist of a uniform mix of gray to black fine silt and clay. Bottom sediments from each core were dated according to carbon-14 analysis by Beta Analytic Inc., (Miami, Florida), and other samples were dated on the basis of pollen horizons and pollen concentration techniques (Brush 1989). The agricultural horizon was identified by a large increase in the percentage of *Ambrosia* pollen between samples within each core and a change in the ratio of *Quercus* pollen to *Ambrosia* pollen (Brush et al. 1982). Based on historical agricultural data, the initiation of widespread agriculture (the level at which Ambrosia pollen increases), is estimated at 1760 A. D. for land near this transect. The ^{14}C-determined dates at the bottom of these cores range from 910-2650 years before present (BP).

Sedimentation rates for each sample (2 cm interval) of each core were obtained by adjusting average rates between the radiometrically and pollen-dated horizons according to the pollen concentration in the sediment (Brush 1989). The methods and results of the geochemical parameters studied can be found in Cooper and Brush (1991), Cooper and Brush (1993), and Cooper (1993). Diatoms were extracted from 40 subsampled 2 cm intervals of the sediment cores using a modification of the method published by Funkhauser and Evitt (1959). After the extraction process was completed, each sample was diluted separately (from 50 to 125,000 times) in distilled water before mounting, to produce slides with a density of approximately 500-2,500 diatoms per slide. At least 400 diatoms were identified to species for each slide on a Zeiss light microscope under oil immersion at 100 times magnification (Cooper 1993, 1995).

Diatom community diversity was calculated for each sample as Shannon's H' (Shannon and Weaver 1949). Centric/pennate diatom ratios were also determined for each sample in this study. Centric diatoms (a classification term) are generally planktonic forms, and pennate diatoms are generally benthic or littoral forms. The abundant centrics identified in the most recent sediments of these Chesapeake Bay cores, including diatoms of the genus *Cyclotella*, are planktonic diatoms, which grow under high nutrient conditions (Palmer 1969, Marshall and Alden 1990). Many of the pennates identified in the sediments are benthic or epiphytic species, which grow in more oligotrophic waters. The centric/pennate ratio therefore simplifies the diatom data.

An "ecological distance" measurement, D, was used as a measure of the difference between every two diatom communities (all the specimens counted in each sample). The "ecological distance" between two diatom communities is based on the Euclidean distance measure. Cluster analysis of the distance measurements between samples was done by the unweighted pair-group method using arithmetic averages (UPGMA). This method and the results can be found in more detail in Cooper (1995).

Figure 1. Loacation of cores collected in the mesohaline Chesapeake Bay. From Cooper and Brush 1993.

RESULTS

Sedimentation Rates

The results show that sedimentation rates across the Bay are variable in location and time, as shown previously by Goldberg et al. (1978), Officer et al. (1984), and Brush (1984). However, there have been significant overall changes in sedimentation rates through time that correlate with changing land use patterns in the Chesapeake Bay watershed. Sedimentation rates have increased significantly in this area of the Bay since the time of initial land clearance for agriculture by European settlers, as previously reported (Cooper and Brush 1993). Sediment accumulation increased from 4-fold to 7-fold in cores R4-45, R4-50, and 50-E, and 1.5 times in core R4-30. Core R4-30, collected on the western flank of the Bay closest to Calvert Cliffs and the shoreline, has the highest and most variable sedimentation rates of these cores.

Geochemistry Results

The changes seen in organic carbon and nitrogen in the sediment samples of the cores, as a function of time, cannot be readily modeled using accepted decay models for reduction in marine sediments (Berner 1980, Dr. James Hill, pers. comm.). This is attributable to large variations in organic carbon and nitrogen input, especially since European settlement. The higher preservation of organic carbon and nitrogen since European occupation of the Chesapeake Bay watershed is no doubt influenced by the increase in sedimentation rates, as well as eutrophication of the water column.

The changes seen in sulfur preservation and DOP in the sediment samples of cores R4-45 and R4-50 indicate that the geochemical environment has changed over time in mesohaline areas of the Chesapeake Bay. These changes may be attributable to a combination of factors, including increased sedimentation rates, eutrophication, available organic carbon in anoxic sediments, and anoxic or hypoxic bottom water conditions. The measurements of DOP > 0.46 in recent sediment samples suggest that there has been an increase in duration and extent of anoxic bottom waters in this area of the Bay in the past four decades, although the Bay has probably always experienced episodes of anoxic bottom water conditions (Cooper and Brush 1991, Cooper and Brush 1993).

Diatom Results

A total of 16,524 diatoms were identified in 40 subsampled intervals from the four Chesapeake Bay cores. Over 400 taxa of diatoms from 75 genera were recognized in the sediment samples. A list of the 21 most abundant species seen in the sediments can be found in table 1. Many lightly silicified and delicate planktonic diatoms known to occur in Chesapeake Bay waters were not represented in the sediment samples of the cores (most likely because of to breakage and dissolution), such as species of *Rhizosolenia, Chaetoceros*, *Leptocylindrus*, *Asterionella* and *Cerataulina*. Some resting spores and pieces of valves representing these genera were seen. Many of the species identified have not been previously reported for the Chesapeake Bay or its tributaries.

The results show a steady decline in diversity of diatom communities through time in all four cores. Diversity, measured as Shannon's H', ranges from a high of 4.04 in older sediments to a low of 2.28 in recent sediments. The average diversity in sediments dated before European settlement of the watershed is 3.8, and the average diversity in the four most recent samples is 2.6. The average number of species and genera found in the sediment samples dated before 1760 A. D. was 103 and 39, respectively, compared to 77 and 30 in samples dated after 1760 A. D. (Cooper 1993, 1995).

Centric/pennate diatom ratios increase significantly in all four cores after European settlement of the watershed, with dramatic changes in cores R4-45 and R4-50 over the past 40-50 years. This trend indicates a change from a more evenly matched planktonic and benthic community ($c/p \sim 1$) before European settlement, to a predominantly planktonic community ($c/p > 1$) as preserved in the more recent sediments. This change in the ratio suggests increasing eutrophica-tion of the Chesapeake Bay system through time after European settlement of the watershed. This change may also be related to loss of benthic habitat suitable for diatoms owing to turbidity, the decline in submerged aquatic vegetation and filter feeding organisms such as oysters, and anoxia.

The diatom results show trends in the percentage abundance of specific diatom genera and diatom families over time as preserved in the sediments of the Chesapeake Bay cores. For example, diatoms of the genus *Cyclotella* show the highest overall abundance, and the greatest increase in abundance since European settlement,

Table 1. List of 21 most abundant diatom species identified in sediment samples from the mesohaline Chesapeake Bay.

Species	Number of Valves
Total diatoms counted	16,524
Cyclotella choctawhatcheeana Prasad (formerly referred to as *C. cf. caspia* Grunow)	2,923
Thalassionema nitzschiodes Grunow	1,516
Paralia sulcata (Ehrenberg) Cleve	1,233
Cyclotella striata (Katzing) Grunow	771
Skeletonema costatum (Grev.) Cleve	463
Achnanthes delicatula ssp. hauckiana (Grun.) L.-B. and Ruppel	460
Detonula cf. confervaceae (Cleve) Gran (spores)	443
Thalassiosira oestrupii (Ostenfeld) Proschkina-Lavrenko	345
Cocconeis scutellum Ehrenberg & varieties	302
Thalassiosira proschkinae Makarova	242
Thalassiosira lineata Jouse Coscinodiscus lineatus Ehren	215
Catenula adhaerens Mereschkowsky	202
Cocconeis peltoides Hustedt	179
Opephora olsenii Miller	170
Cocconeis placentula Ehrenberg	166
Neodelphineis pelagica Takano	157
Rhaphoneis cf. tenuis Hustedt	157
Navicula abunda Hustedt	127
Cyclotella sp. 2	126
Cyclotella stylorum Brightwell	110
Amphora tenerrima Hustedt	110

of any diatom genus identified in this study (see figures 2 and 3). *Cyclotella* is a predominantly planktonic genus, usually found in freshwater habitats (Round et al. 1990). Many species in this genus show a wide tolerance to environmental conditions, including high nutrient loading (Palmer 1969, Wilderman 1984, Marshall and Alden 1990). A total of 14 species of *Cyclotella* were identified in the sediments, but the large increase observed in this genus in recent sediments is attributable primarily to an increase in *Cyclotella choctawhatcheeana* (formerly referred to as *C. cf. caspia*), which is a brackish water species that is very tolerant of high nutrient conditions and variable salinity (Wilderman 1984, Marshall and Alden 1990).

The computed differences between samples for this study are contained in a 40 x 40 matrix of distance measurements. The samples showing smaller "distances" are generally dated within similar time periods. This becomes clear with cluster analysis of the data. A Euclidean distance of D = 0.15 separates the diatom communities (from each sample) into four distinct clusters. The first group or cluster is comprised entirely of samples dated before European settlement and land clearance of the watershed (pre-1760 A. D.) from all four cores. The second group is comprised of samples that are dated between the average dates of 1723 and 1851 A. D. from all four cores, except for one sample from core R4-50 with an average date of 1541 A. D. The time period covered by the second cluster includes European settlement and initial land clearance for agriculture. The sample dated 1541 A. D. that clusters with this group may indicate the influence of Native American settlement on the watershed at that time. These first two groups are linked together at a Euclidean distance of D = 0.2 and separated from the following two groups by a distance of D = 0.35. The third group is comprised of samples dated between the average dates of 1857 to 1898 A. D. from all four cores, except for two samples, one dated 1830 A. D. from core R4-50

Figure 2. Precentage abundance of selected diatom genera and species from core 50-E taken in the mesohaline Chesapeake Bay. Land use zones are delineated at 1760 A.D., 1850 A.D., and 1940 A.D.

Figure 2. Precentage abundance of selected diatom genera and species from core R4-50 taken in the mesohaline Chesapeake Bay. Land use zones are delineated at 1760 A.D., 1850 A.D., and 1940 A.D.

and one dated 1943 from core R4-45. The time period of this third cluster generally covers the period of maximum land clearance of the watershed for agriculture. The fourth and final group is comprised of samples all dated between the average dates of 1897 and 1980 A. D. This final group is further subdivided at a distance of D = 0.12 into two subgroups. The first subgroup contains samples dated between 1897 and 1942 A. D. (except for one sample of 1971 A. D.), and the second subgroup contains samples all dated from 1960 to 1980 A. D. This time period includes industrialization and urbanization of the watershed (see Cooper 1995).

DISCUSSION

The four cores used in this study contain a sedimentary record spanning 2,000 years and areas of the Bay that have historically different patterns of oxygen depletion in the bottom waters. All four cores show similar diatom community diversity at comparable time periods and all four cores show the same trends of decreasing diatom community diversity, including number of species and genera, through time. These trends are in opposition to expected changes owing to natural breakage and dissolution of diatom valves over time, especially considering the lower sedimentation rates before European settlement.

The results of the cluster analysis of diatom data also show that large changes in the diatom communities occur at times of historically significant changes in land use patterns of the watershed land use patterns that also affect sedimentation rates and geochemical indicators within the cores. The first two cluster groups of diatom communities are separated at the time period of initial European settlement and land clearance for agriculture (around 1760 A. D.).

Significant changes in other paleoecological indicators measured appear in the sediments at the time of European settlement and land clearance for agriculture around 1760 A. D. This time period is marked by an increase in the percent abundance of *Ambrosia* pollen preserved in the sediments, significantly higher sedimentation rates than in the hundreds of previous years, and a large increase in preservation of organic carbon, nitrogen, and sulfur in the sediments.

After initial land clearance for agriculture, deforestation continued with up to 80% of the land cleared in the mid 1800s, along with new deep plowing methods employed for agriculture. The data from this time period show the highest sedimentation rates for this area of the Bay, with peaks in organic carbon, nitrogen, and sulfur preservation in the sediments. The diatom communities as preserved in these sediments exhibit a continued decline in diversity, an increase in centric/pennate ratios, and continued changes in relative abundance of species as compared to older samples. There is a noticeable decline or absence of many marine benthic diatom species and an increase in brackish water planktonic diatoms.

Changes in diatom community structure continue into the 20th century with land use changes associated with urbanization and industrialization. The diatom assemblages preserved in recent sediments show a remarkable decline in diversity from older samples, with much fewer species and genera observed, increases in centric/pennate ratios, and changes in the relative abundance of the species. The number and abundance of marine benthic species continue to decline, with synchronous increases in planktonic brackish and freshwater diatom species such as *Cyclotella choctawhatcheeana* (formerly referred to as *C. cf. caspia*) and *Achnanthes minutissima*. This change suggests increased freshwater input to this area of Chesapeake Bay during this time period. The diatom communities as seen from this time period are very similar between cores, and very different from those seen in older sediments of the four cores, as measured by Euclidean distance and cluster analysis (Cooper 1993, 1995).

This time period also shows the largest peaks in organic carbon and sulfur preservation in the sediments, despite somewhat lower sedimentation rates than during the mid to late 1800s. The DOP results indicate that hypoxia and anoxia may be more severe and of longer duration in the last 50 years than in the previous history of Chesapeake Bay. It may be that a threshold has been reached in eutrophication of the water column and organic carbon within the sediments, so that years of high spring runoff and stratification of the water column can produce high levels of sulfur and DOP in the sediments. These conditions produce hydrogen sulfides, which exacerbate anoxic conditions within the bottom waters of the Bay.

The results of this paleoecological study of the sediments of the mesohaline Chesapeake Bay indicate that sedimentation rates and eutrophication of the waters of the Bay have increased dramatically since the time of European settlement of the watershed. There is also evidence that freshwater input and hypoxic and anoxic bottom waters have increased in the past 50 years. It appears that conditions have been more conducive to loss of oxygen below the pycnocline during years of high freshwater flow since at least the mid 1800s, as compared to pre-European time periods. The results also show that there has been a decline in habitat suitable for benthic and marine diatoms within the mesohaline Bay, probably owing to a host of factors, including eutrophication, turbidity, and freshwater input, all of which also affect other organisms such as submerged aquatic plants, filter feeders such as oysters, and other algae and bacteria (which in turn may affect diatom species abundances).

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ASSOCIATIONS BETWEEN CHESAPEAKE BAY PROGRAM ZOOPLANKTON MONITORING DATA AND MARYLAND AND VIRGINIA JUVENILE FINFISH SURVEY DATA

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Abstract: Many of the dominant finfish species in Chesapeake Bay waters are obligate planktivores (e.g., bay anchovy, Atlantic silversides, Atlantic menhaden), suggesting strong links might be found between zooplankton and finfish communities. Correlations between the average summer abundances of mesozooplankton (> 202 µm) and finfish that are obligate planktivores are examined in this paper. Abundances were obtained from the Chesapeake Bay Program zooplankton monitoring data and the Virginia and Maryland juvenile finfish seine survey data. Planktivore - mesozooplankton relationships were found at five of ten Chesapeake Bay Program zooplankton monitoring stations in tidal fresh and oligohaline waters during the summer growing season. Abundances were inversely related in tidal fresh waters, suggesting top-down control of the mesozooplankton by finfish planktivores. Abundances were directly related in oligohaline stations, indicating bottom-up control of the finfish planktivores by mesozooplankton. Planktivore and/or mesozooplankton abundances were consistently low at tidal fresh and oligohaline stations where no relationships were evident. These stations have poor water quality, changing water quality, and/or rapid flushing rates. Straightforward relationships between mesozooplankton and finfish planktivores were not evident at the four mesohaline stations, where invertebrate predation (jellyfish, meroplankton) and hypoxia/anoxia become significant controllers of the mesozooplankton community during the summer. Further investigations will hopefully substantiate the plankton/fish linkages in all salinity regimes and quantify the factors that disrupt linkages. Predator/prey relationships can be the basis for developing zooplankton indicators of finfish community structure and trophic imbalances in Chesapeake Bay, as they are elsewhere.

INTRODUCTION

Zooplankton are the prey of many abundant fish species, several depleted fish species, and most fish larvae in Chesapeake Bay. Bay anchovy and Atlantic silversides, currently the dominant resident species (Carmichael et al. 1992), are obligate planktivores their entire lives, feeding only on mesozooplankton. Atlantic menhaden, the dominant species, consume zooplankton during early life stages in coastal waters, then develop specialized brachial structures after entering the estuary that allow them to filter phytoplankton and detritus as well as zooplankton. American shad and the river herring, which were historically abundant and heavily exploited and are now habitat-impaired, feed principally on zooplankton during their growing periods in the estuary. Zooplankton are the obligate prey of larval stages of most finfish species, regardless of what prey the larvae switch to as they metamorphose. For example, striped bass feed on zooplankton in spring and early summer as larvae, become facultative predators of invertebrates near the end of their first summer, and are strict piscivores by one year of age. Finally, facultative predators on zooplankton (e.g. sunfish, minnows, killifish) are presently abundant in bay fish communities (Carmichael et al. 1992) and will consume mesozooplankton along with other prey.

The dominance of obligate and facultative planktivores in Chesapeake Bay finfish communities suggests that strong trophic linkages exist between finfish and zooplankton. Such linkages are evident as close correlations between predator and prey. The discovery of similar relationships in freshwater systems has lead to the development of zooplankton indicators of finfish community structure, for the purpose of fisheries management. Galbraith (1975) used the abundance of *Daphnia* spp. to predict the survival and "fishing quality" of rainbow trout in Michigan lakes. Mills and Schiavone (1982) successfully correlated zooplankton size, growth of planktivorous fish, and the size structure of percid and centrarchid populations in New York lakes. Mills, Green, and Schiavone (1987) further observed that zooplankton size in the New York lakes was a good indicator of the relative abundances of piscivores and "panfish" (planktivores). Resource management strategies in the Great Lakes have for some time recognized the value of zooplankton as indicators of fish community structure and ecosystem balance (Evans and Jude 1986, Johannsson 1987, Hartig et al. 1991). Considering the Bay-wide coverage of the zooplankton monitoring program, zooplankton indicators could be useful to Chesapeake Bay management if finfish/zooplankton linkages are found.

METHODS

In this paper, correlations between the average summer abundances of obligate planktivores and their prey, the mesozooplankton, in the upper mainstem and six Chesapeake Bay tributaries are calculated. Trophic linkages were expected to be strongest between obligate planktivores and their prey. Trophic linkages are also most evident in July, August, and September, when planktivorous species are actively feeding, growth rates are at their annual maxima, and young of the year contribute substantially to the overall predation pressure on mesozooplankton. Tributary differences in water quality, hydrology, and salinity were expected to help sort out the influences of other controlling factors and clarify the environmental limits within which strong trophic linkages are possible. For example, regressions with slopes nearly parallel to the graph's axes or with weak correlation coefficients indicate other controlling factors (e.g., salinity, water quality, high flow, predation by another group) strongly influence the mesozooplankton or finfish planktivores.

Finfish planktivore data were obtained from the Maryland Estuarine Juvenile Finfish Survey and the Virginia Juvenile Striped Bass Survey. Both are long-term, shoreline seine surveys done in bay tributaries and the upper Bay. Sampling sites are located in the spawning and nursery grounds of commercially important anadromous fish. Seine hauls are done in July, August, and September, and all species are at least identified and counted. Details of the programs and maps of the seine station locations are given in the *Chesapeake Bay Basin Monitoring Program Atlas* (Heasly et al. 1989) and elsewhere. For this study, only seine stations located near a zooplankton monitoring station were used (table 1).

Maryland and Virginia seine survey protocols are different, so the data sets were normalized to make them comparable. Maryland collects three rounds of seine hauls at each shore site, with two hauls per round, for a total of six hauls per summer. Virginia collects five rounds of two seine hauls per round for a total of ten hauls during the same time period. Occasionally, sites in both states were not sampled. To prevent the gaps from biasing finfish estimates, site-year data were excluded under the following conditions: (1) in Maryland if at least two hauls out of six total hauls were missing for a site in a particular year, except if a zooplankton monitoring station was paired with only one seine station, in which case all of the data were kept whether or not there were missing hauls. (nine site-year combinations were deleted from the Maryland data), and (2) in Virginia if at least three hauls out of the ten total hauls were missing for a site in a particular year (only one site had at least three hauls missing; an additional twenty-one site-year combinations had two of the ten hauls missing, but they were not deleted.)

For each year, species counts from the seine sites adjacent to each zooplankton monitoring station were grouped and averaged to obtain the mean abundance of each species per round (two seine hauls) in both Virginia and Maryland. Means of species known to be obligate planktivores were then extracted and summed to obtain average planktivore abundance per

Table 1. Station matches for zooplankton and juvenile finfish seine surveys in Maryland and Virginia. Alphanumeric listings in parentheses indicate previous station designation.

System	State Juvenile Finfish Seine Station	CBP Zooplankton Monitoring Station
James	J56	TF5.5 (1J)
	J36, J29	RET5.2 (2J)
York	P51, P45 P42, P41	TF4.2 (1Y)
Rappahannock	R55, R50, R44	TF3.3 (1R)
	R37, R28	RET3.1 (2R)
Potomac	49, 50	TF2.3 (XEA6596)
	51, 62, 52	RET2.2(XDA1177)
	55, 64, 56	MLE2.2
Patuxent	85, 86	TF1.5 (PXT0402)
	92	TF1.7 (XED4892)
	106, 90	LE1.1 (XDE5339)
Upper Bay	68, 59, 3	CB1.1 (MCB1.1)
	10, 11, 88	CB2.2 (MCB2.2)
Choptank	002, 66	ET5.1 (MET5.1)
	67, 28, 29	ET5.2 (MET5.2)

Table 2. Obligate planktivore finfish species in Chesapeake Bay.

- Alewife
- American shad
- Atlantic menhaden
- Atlantic silverside
- Atlantic thread herring
- Banded killifish
- Bay anchovy
- Blueback herring
- Bridle shiner
- Comely shiner
- Gizzard shad
- Golden shiner
- Spottail shiner
- Striped anchovy
- Pipefish

round for each year. A list of these planktivore species is given in table 2.

Mesozooplankton include copepodites and adult copepods, cladocera, meroplankton, and mysids. Mesozooplankton data were obtained from the ongoing Maryland and Virginia zooplankton monitoring programs of the Chesapeake Bay Program (CBP) for the years 1984-91 (Choptank, Patuxent, and Potomac Rivers, upper Bay), 1986-91 (James and Rappahannock Rivers) and 1987-91 (York River). At each zooplankton monitoring station, samples are collected with towed nets (202 um mesh) from several depths and combined. Station locations are shown in figure 1. Several lower tributary zooplankton stations and all of the Bay mainstem stations south of CB2.2 could not be matched with seine sites and are therefore not included in this study.

Regressions were made between the average station abundance of mesozooplankton for July, August, and September of each year

Figure 1. Chesapeake Bay Program zooplankton monitoring stations.

and each year's average planktivore abundance for the matching seine station(s). Historical mesozooplankton data were available in the vicinity of the upper Potomac TF2.3 station for 1974 (unpublished data obtained from Versar, Inc. and described in Ecological Analysts, [1974] and for 1981 (Buchanan and Schloss 1983).

RESULTS

Only stations that experienced similar salinities were directly compared because of the recognized impact of salinity on zooplankton community structure. The term "planktivore" refers to finfish planktivores in the following discussion, except when noted otherwise. Analysis results are summarized in table 3.

Four zooplankton monitoring stations are entirely in tidal freshwater (0 - 0.5 ppt salinity): CB1.1 (upper Bay), TF2.3 (Potomac), TF5.5 (James), and TF4.2 (York). The average planktivore abundances at seine sites near these four stations were low relative to brackish water sites during the study period. In contrast, summer zooplankton abundances were relatively high at the Potomac station, low at the James and York stations, and exceptionally low at the upper bay station (table 3). Summer zooplankton community structure in the tidal fresh was diverse compared to oligohaline and mesohaline communities. The upper Bay and the Potomac have the largest tidal freshwater reaches in the Chesapeake Bay complex of waterways; the York has one of the smallest.

The upper bay and Potomac stations showed inverse relationships between mesozooplankton and planktivore abundance during the summer months (figures 2a, and 2b). Mesozooplankton abundance decreased when planktivore abundance, and presumably predation pressure, increased. The inverse relationship indicates mesozooplankton abundance is the dependent variable and varies in response to planktivore abundance. The upper Bay data span a small range of mesozooplankton abundances and a large range of planktivore abundances. The 1991 datum was excluded because flows from the Susquehanna River were exceptionally low that summer. Mean daily flows averaged 5847.5 cubic ft^3/sec for 92 days (July-September), or near the 10th percentile of all mean daily flows for 1967-92. Consequently, seine sites near CB1.1 were at times oligohaline. The ten Potomac data points span a wide range of both planktivore and mesozooplankton abundances, and they best fit a log-log curve (i.e., log [mesozooplankton] = 7.796 - 1.5321 log [planktivores]) when the 1985 datum is removed. In 1985, submerged aquatic vegetation (SAV) returned suddenly to the TF2.3 area between Marshall Hall and Quantico (Carter and Rybicki 1986) and dramatically affected the tidal fresh ecosystem (see below). The position of the 1985 datum as an outlier suggests there were short-term repercussions on the planktivore/mesozooplankton relationship.

Planktivore/mesozooplankton relationships were not found at the tidal fresh James (TF5.5) and York (TF4.2) stations (table 3, figures 2c and 2d). In the James, summer planktivore abundances were approximately one-half of those in the tidal fresh Potomac and upper Bay. Summer mesozooplankton abundances were low, approximately one-half to one-quarter of those found in the tidal fresh Potomac and the smaller, freshwater/oligohaline reaches of the Choptank and Patuxent. The tidal fresh York had relatively low mesozooplankton abundances and populations frequently crashed below 1,000

Figure 2. Average summer (July-September) mesozooplankton abundance (number per m^{-3}) at tidal fresh zooplankton monitoring stations versus average planktivore abundance (number per round) at adjacent juvenile seine survey sites: (a) CB1.1 (upper Bay), (b) TF2.3 (Potomac River), (c) TF5.5 (James River), d) TF4.2 (York River). Solid lines are significant regressions ($p \le 0.08$). In (a), the 1991 datum is excluded because seine sites near this freshwater station were oligohaline at times. In (b), the 1985 datum is excluded because of significant changes in the SAV population that year; dotted circles are 1986-90. See table 3 and text for discussion.

Table 3. Summary of regressions between summer (July-September) averages of finfish planktivore abundance and mesozooplankton abundance in Chesapeake Bay tributaries and the Upper Bay All the regressions were linear, with the exception of the Potomac (all data), whi
regression of the log of planktivore abundance and the log of mesozooplankton abundance. See Figure 1 for station loc
0.5 ppt), O = oligohaline (0.5 - 5 ppt), LM = low mesohaline (5 -10 ppt), and M = mesohaline (10 - 18 ppt). Mesozoopl
average number per cubic meter, planktivore abundance as average number per pound. Within each salinity regime, tribut
to smallest.

mesozooplankton per m^3 (R. Birdsong, pers.comm.). Planktivore abundances near the tidal fresh York station were often the lowest found in the Virginia and Maryland seine surveys combined.

The relatively diverse zooplankton community of the tidal fresh shifts quickly to an *Acartia* dominated, estuarine community as it enters the oligohaline (0.5 - 5.0 ppt salinity). However, there is no consistent pattern of change in summer mesozooplankton abundance moving downstream from tidal fresh to oligohaline stations in Chesapeake tributaries (table 3).

Summer salinities at none of these stations were strictly oligohaline. Stations ET5.1 and TF1.5 typically experienced both fresh and oligohaline conditions during the summer. Stations RET5.2 and TF1.7 experienced both oligohaline and low mesohaline [5 -10 ppt] conditions. And stations RET2.2 and CB2.2 experienced fresh, oligohaline, and low- mesohaline conditions. Data for summers with predominantly tidal fresh conditions have been removed from the RET2.2 and CB2.2 regressions. The Rappahannock station TF3.3 was not included in this paper because salinities there range from tidal fresh to mesohaline, with no clear dominance of one salinity regime.

Abundances dropped in the Potomac, rose in the upper Bay, and remained low in the James. Abundances declined somewhat between the fresh/oligohaline and the oligohaline/low-mesohaline stations in the Patuxent. Summer planktivore abundances increased moving downstream to the oligohaline reaches, except in the James and Patuxent.

Three of the six oligohaline stations exhibited positive correlations between summer planktivore and mesozooplankton abundances during the study period (figures 3a, 3e, and 3f). Specifically, planktivore abundance was high when mesozooplankton abundance was high. The positive correlations suggest that planktivores are the dependent variable and are responding to mesozooplankton abundance (food availability). Clear relationships were not found in the oligohaline/low mesohaline James (figure 3c) or the tidal fresh/oligohaline Patuxent (figure 3d). A weak inverse relationship was found in the oligohaline/low mesohaline Potomac (figure 3b).

The mesohaline covers extensive stretches in the middle and lower tributaries of the Chesapeake Bay, as well as approximately half the length of the Bay mainstem. The Maryland and Virginia juvenile finfish seine surveys, from which the planktivore estimates were derived for this study, extend only into the tributary mesohaline reaches because they focus on summer nursery areas of anadromous fish. Furthermore, the James and the York do not have zooplankton monitoring stations in true mesohaline waters. Therefore, mesozooplankton/planktivore linkages could be examined only at four tributary mesohaline stations: ET5.2 (Choptank), LE1.1 (Patuxent), LE2.2 (Potomac) and RET3.1 (Rappahannock). No correlations were evident between mesozooplankton and planktivore abundance at these four stations (table 3). Summer densities of mesozooplankton at the tributary mesohaline stations were variable, with a relatively high average at ET5.2 (Choptank), moderate averages at LE1.1 (Patuxent) and LE2.2 (Potomac), and a low average at RET3.1 (Rappahannock). Planktivore abundances near zooplankton monitoring stations were moderate, except in the Choptank where they were high (table 3).

DISCUSSION

Tidal Fresh (0 - 0.5 ppt) and Oligohaline (0.5 - 5 ppt)

A planktivore/mesozooplankton relationship occurs at the upper Bay station (CB1.1) despite the suspected impacts of high flow and eutrophication on mesozooplankton abundance (table 3). The station is located in the high-flow zone at the mouth of the Susquehanna River. Samples from this particular station underestimate average zooplankton densities for the upper Bay area known as the Susquehanna Flats (K. Sellner and F. Jacobs, pers. comm.). However, they track zooplankton trends found downstream at CB2.2 and probably reflect actual trends and trophic relationships in the flats. Zooplankton populations appear to be affected by the relatively high ambient nutrient concentrations in the Susquehanna Flats; that is biomass ratios of microzooplankton to mesozooplankton are exceptionally high (1984-90 average = 85%) when compared to ratios from other tidal freshwater reaches of the bay. Nitrogen concentrations did not change significantly during the study period whereas phosphorus declined somewhat (Magnien and Boward in preparation).

Significant habitat changes at, the tidal fresh Potomac station during the 1980s appeared to shift the mesozooplankton/planktivore relation-

Figure 3. Average summer (July-September) mesozooplankton abundance (number per m^{-3}) at oligohaline zooplankton monitoring stations versus average planktivore abundance (number per round) at adjacent juvenile seine survey sites: (a) CB2.2 (upper bay), (b) RET2.2 (Potomac River), (c) RET5.2 (James River), (d) TF1.5 (Patuxent River), (e) TF1.7 (Patuxent River), (f) ET5.1 (Choptank River). Solid lines are significant regressions ($p \leq 0.06$). Dashed line in (b) is a questionable regression ($p = 0.08$). Data for 1984 and 1990 in (a) and 1989 in (b) are excluded because salinities were below 0.5 ppt. See table 3 and text for details. The positive regression slopes indicate mesozooplankton, the prey, are the independent variable, hence their abundances are put on the X axis. Correlation coefficients are the same regardless of which variable is placed on the X axis. See next page for (e) and (f).

Figure 3. continued

ship from one end of the regression curve toward the other. In 1985, biomass estimates of submerged aquatic vegetation (SAV) in the TF2.3 area increased 16.5-fold (spring) to 1.7-fold (fall) over the previous year (Carter and Rybicki 1986), and acreage quadrupled (Chesapeake Bay aerial SAV surveys). The SAV improved water quality (Carter et al. 1988) and allowed a resurgence of largemouth bass (L. Fewlass, pers.comm.). Largemouth bass is a top predator that feeds primarily on smaller fish, including planktivores and crayfish and relies heavily on SAV for habitat. Largemouth bass surveys (1988, 1989, 1992, 1993) in various Maryland Chesapeake Bay tributaries show that the Potomac population is presently the largest in Maryland, has good-excellent recruitment, and provides "an outstanding fishery" with a relatively high catch per angler per hour rate (Fewlass et al. 1993). On the average, planktivores declined about 20% after 1985 and mesozooplankton increased more than 60%. Different regression slopes seem to be evident for pre-SAV and SAV-concurrent data. Data for the three pre-SAV years (1973,1981,1984) suggest a shallow negative regression slope whereas the 5 years after 1985, when SAV populations were fairly stable in the vicinity of TF2.3, reveal a steep negative slope (table 3, figure 2b). Changes in water quality did not correlate with changes in mesozooplankton abundance in the tidal fresh Potomac during the study period. Total phosphorous concentrations were relatively low, and loads declined. Total nitrogen concentrations were high, and they increased (Magnien and Boward in preparation, Maryland Department of the Environment 1993).

In the tidal fresh James and York, planktivore and mesozooplankton were low, and no relationship between the two variables was found. These findings suggest some factor, or combination of factors, has an impact large enough to depress both planktivore and mesozooplankton populations and disrupt linkages between the two. Ongoing analyses to determine the importance of flow and residence time to zooplankton populations in Chesapeake tributaries will help to clarify whether or not natural causes are responsible. Other possible causes could include ammonia and low dissolved oxygen. For 1986 - 1988, 22 violations of the U.S. Environmental Protection Agency (EPA) ammonia criteria were observed in the tidal fresh James, with most occurring in summer, and summer levels of dissolved oxygen went below 4.0 milligrams per liter in approximately 9% of the tidal fresh York measurements (Alden et al. 1992).

The positive correlations between summer planktivore and mesozooplankton abundances found at the oligohaline stations CB2.2, TF1.7, and ET5.2 can be interpreted either as evidence of a direct link between predator and prey or as evidence of another factor causing predator and prey to vary in a similar fashion. The first interpretation seems correct because there do not appear to be any factors that can similarly control mesozooplankton and planktivore abundances in all the oligohaline reaches. Piscivores crop juvenile and adult planktivores but not mesozooplankton. Jellyfish predators influence both mesozooplankton and fish larvae, but they are rare in the oligohaline (Lippson et al. 1979). Similarly, most meroplankton predators of mesozooplankton and fish larvae have small, pulsed populations in the oligohaline and cannot exert a large, sustained predation pressure. A substantial population of facultative finfish planktivores in the oligohaline (e.g., striped bass young of the year (Y-O-Y), mummichog, sticklebacks, sheepshead minnow, and the rainwater, striped, marsh, and spotfin killifishes) could conceivably regulate both planktivore larvae and mesozooplankton, and this possibility remains to be examined. A cursory look at the fish communities in Maryland (Carmichael et al. 1992) suggests this possibility is unlikely because these are not dominant species. Water quality at CB2.2, TF1.7, and ET5.2 were not similar. Overall conditions ranged from "fair" (CB2.2, TF1.7) to "generally good" (ET5.2). Total nitrogen concentrations were "severely impacted" at TF1.7, "stressed" at CB2.2, and "fair" at ET5.2. Chlorophyll and phosphorous concentrations varied between low (CB2.2) and high (TF1.7). Dissolved oxygen concentrations ranged from sufficient (CB2.2 and ET5.2) to fair-poor (TF1.7) (Magnien and Boward in preparation, Maryland Department of the Environment 1993). If one accepts the interpretation that predator and prey are directly linked and not similarly controlled by a third variable, then the regression coefficients imply a substantial degree of bottom-up control by the mesozooplankton on the planktivores in the oligohaline reaches of Chesapeake Bay.

In comparison to the positive regression slopes above, the weak ($p = 0.08$) inverse relationship between planktivore and mesozooplankton abundances at the Potomac oligohaline station (RET2.2) is odd. Changing water quality conditions and significant concentrations of several chemical pollutants are characteristic of this portion of the Potomac. Ambient concentrations of nitrogen at RET2.2 are high and have increased since 1984. Total phosphorous concentrations are low and declining (Magnien and Boward in preparation, Maryland Department of the Environment 1993). A pilot study done for the Chesapeake Bay Program (Hall et al. 1992) and earlier studies found water column and sediment toxicity in the general area RET2.3. Known stressors in the water column downstream at Morgantown and the Dahlgren Naval Weapons Laboratory include tributyltin (TBT), copper, and nickel, and possibly mercury and lead, in excess of EPA water quality criteria. *Acartia*, the dominant species during the summer, is known to be sensitive to trace metals, and population crashes in the Elizabeth River (which empties into the lower James River) have been associated with elevated metal concentrations (Sunda et al. 1990). Further years of data are needed to resolve whether the inverse planktivore - mesozooplankton relationship at RET2.2 is a valid one or an artifact of other factors.

The absence of a mesozooplankton/planktivore relationship and the low abundances of both groups in the James (RET5.2) repeats the pattern found at the James tidal fresh station (table 3). It reiterates the hypothesis that outside controlling factors may be decoupling trophic linkages at this station. Possible reasons for this pattern are still uncertain and information on toxic pollutants other than Kepone is scarce. Kepone, a chlorinated hydrocarbon, was found in high concentrations in the James in 1975 and resulted in a decade-long restriction on fishing in the system. It is still an important contaminant in the lower James (Kennish 1992).

The Patuxent fresh/oligohaline station (TF1.5) is more transitional in nature than the other oligohaline stations, which may be one reason a planktivore/mesozooplankton relationship was not found here. Zooplankton species composition at this station is most like those in tidal fresh stations, except for frequent incursions by *Acartia*. Similarly, summer planktivore abundances are more comparable to those in the tidal fresh. Changing nutrient concentrations at this station owing to improved wastewater treatment may be another factor modifying a planktivore/mesozooplankton relationship. For example, summer concentrations of ammonium, a compound whose ionic form (NH_3^+) is toxic to aquatic organisms when present in high concentrations, have declined over 80%, and summer dissolved oxygen has increase 1.5 mg/l during the study period (S. Bieber, pers.comm.). Phosphorous loadings are still high here, however approximately three times the loadings in the larger Potomac River. When the system reaches a new dynamic equilibrium and improved water quality conditions are established, a mesozooplankton/planktivore relationship may become evident and the influence of salinity on the regression can be clarified.

The slopes of the significant planktivore/ mesozooplankton regressions differ markedly in the tidal fresh and oligohaline reaches, that is, they are negative in tidal freshwater and positive (with one possible exception) in oligohaline water. The difference indicates a fundamental change takes place in the zooplankton/fish relationship at the leading edge of the salt wedge. It is believed that the relative dominance of predator and prey responses to each other and to their environment determines whether predator/prey correlations are positive, negative, or absent when predation pressure is strong (Williamson et al. 1989). Abundant planktivores occur in all reaches of the Bay except the York (TF4.2) and the James (TF5.5, RET5.2), indicating predation pressure is strong. A comparison of the tidal fresh and oligohaline habitats and communities highlights some factors potentially causing shifts in prey vulnerability, predator/prey overlap, and predator efficiency as salinity changes. Zooplankton diel vertical migration, a versatile method of reducing predator/prey overlap in most aquatic systems, is regularly disrupted by strong vertical mixing in the tidal fresh (Buchanan and Schloss 1983) and oligohaline (Heinle et al. 1979) reaches of partially mixed estuaries. The loss of this adaptive behavior is somewhat compensated for by higher turbidity in estuaries which shrinks the reactive zones of visual planktivores (although not of Atlantic menhaden, the dominant species in Chesapeake Bay). Prey vulnerability is further reduced in the oligohaline by a major, salinity-induced shift in zooplankton species composition from a diverse freshwater community frequently dominated by cladoceran species to an estuarine community dominated by one copepod species, *Acartia tonsa*, in the summer. *Acartia* tolerate a wide range of salinities. They are omnivores capable of selectively consuming detritus, net phytoplankton and even smaller zooplankton (Lonsdale 1981, White and Roman 1992) and are therefore well adapted to utilizing the enormous amounts of organic material generated as freshwater species die out. They are also, as copepods, better adapted to escaping fish predators than the slower moving *Cladocera*, which rely more on vertical migration and transparency to avoid predation. The shift toward an *Acartia* dominated community could be expected to reduce the influence of both the environment and predation as controlling factors on the overall mesozooplankton population and consequently change the zooplankton/fish relationship.

The different regression slopes in the tidal freshwater and oligohaline also suggest the following hypothesis: the dominant direction of trophic control is top-down in the tidal fresh and bottom-up in the oligohaline. As evidenced by inverse regression slopes, predators appear to have more control over the prey's abundance than the prey, as food, have on predator abundance in the tidal fresh. This echoes a pattern found repeatedly in freshwater lakes, where manipulations of planktivore abundance bring about opposite changes in the abundance of lake zooplankton. If further analysis of the monitoring data and experimental work provide more evidence that top-down controls predominate in Chesapeake Bay tidal freshwater food chains, management actions that maintain moderate rather than excessive concentrations of planktivores will encourage vigorous populations of freshwater mesozooplankton species. Most of these zooplankton species are herbivorous on algae. Conversely, in Chesapeake Bay oligohaline waters during summer, the prey appear to have more control over predator abundance than the predators, as consumers, have on prey abundance. Management actions that increase zooplankton abundance in oligohaline waters could be expected to enhance planktivore survival and abundances there.

Tributary Mesohaline (5 - 18 ppt salinity)

No relationships were evident between mesozooplankton and planktivore abundance at the four mesohaline stations. The implication here is that the trophic linkage between obligate planktivores and their principal prey, the mesozooplankton, is either masked or uncoupled by other factors. A diverse, abundant collection of zooplankton predators and chronic summer hypoxia/anoxia are two known factors in mesohaline waters that exert strong controls on zooplankton populations. Predators of zooplankton during the summer include a variety of meroplankton larvae and epibenthic crustacea, *Neomysis americana* (mysid shrimp), the ctenophore *Mnemiopsis leidyi*, and the larvae of serially spawning finfish in addition to juvenile and adult finfish planktivores. All the stations stratify to some extent during the summer, the Choptank and Rappahannock sporadically and weakly and the Potomac and the Patuxent strongly and for long periods. Hypoxic, and sometimes anoxic, layers became established in the Potomac and Patuxent each summer and periodically intrude into the Choptank station from the Bay mainstem.

The diverse array of zooplankton predators in the mesohaline, in contrast to the tidal fresh and oligohaline, suggests in itself that planktivory is strong there and derives from numerous competing predators rather than one large group of similar predators (i.e., finfish planktivores). Many of the mesohaline predator species are thought to be capable of individually affecting zooplankton populations when they are abundant. For example, *Mnemiopsis leidyi*, the sea walnut, can consume 470 copepods per hour (Bishop 1967), and population maxima in mid-summer have been negatively associated with East Coast estuarine copepod abundances (Mountford 1980). The impact of this invertebrate planktivore is reduced when *Chrysaora quinquecirrha*, a jellyfish predator of the sea walnut and zooplankton, reaches its annual maximum. Similarly, *Chrysaora* predation on ctenophores indirectly influences the predation potential of ctenophores on fish larvae in Chesapeake Bay by reducing ctenophore numbers (Cowan and Houde 1992). Regressions that account for the predation pressures of both invertebrate and finfish planktivores may show a clear relationship to mesozooplankton abundance in mesohaline reaches.

Analyses of historical monitoring data (1976 -80) from mesohaline waters of the Chesapeake Bay mainstem near Calvert Cliffs indicates the multiple regression method has promise. Olson (1987) used weekly and monthly data in stepwise regressions of mesozooplankton with water quality, food, and predator abundance parameters monitored from 1976 to 1980. For the monthly data from May to September, biological variables that were significantly associated with *Acartia tonsa* abundance in single-year models included chlorophyll (1978), *Neomysis*, an invertebrate predator of zooplankton (1978, 1979), and bay anchovy biomass (1976). The one year that chlorophyll was significantly, and negatively, correlated with mesozooplankton coincided with many red-tide blooms, which are unpalatable to zooplankton. The relationships with *Neomysis* were negative (inverse), whereas the relationship with bay anchovy was positive, suggesting top-down control of zooplankton by the mysid shrimp and bottom-up control of the bay anchovy by the zooplankton. When all the years were combined, Atlantic menhaden biomass was the second most significant variable after temperature. Again, the regression slope was positive. Olson used data from May through September, which perhaps allowed temperature to dominate the combined-year model and many of the single-year models as the most significant variable. Reanalysis of the Calvert Cliffs data for the narrower time period of July through September, when temperatures do not span a wide range, finfish planktivory is typically at its annual maximum, and community composition is relatively stable would be very helpful in documenting summer linkages between mesozooplankton and their predators, both invertebrate and finfish, in the Chesapeake Bay mainstem for the late 1970s, and whether these linkages have changed in the last 15 years of increasing eutrophication.

Food Web Management Strategies

The inverse mesozooplankton/planktivore relationship described for the tidal fresh reaches indicate a “trophic cascade effect” in action. This concept was derived from recurring patterns of trophic interactions observed in freshwater lakes over many decades and recently synthesized into an overarching concept called the trophic cascade effect. The concept states that substantial changes in the top predator population will have significant repercussions on all of the lower trophic levels in an otherwise balanced system (ecosystems that exhibit a dynamic equilibrium over the long-term, whose populations fluctuate seasonally or over longer cycles but maintain constant baseline abundances and whose production of organic material is in rough proportion to consumption (according to *Chesapeake Bay Strategy for the Restoration and Protection of Ecologically Valuable Species*, 1993 prepared by the Chesapeake Bay Program), (Carpenter et al. 1987, Hartig et al. 1991). Studies have documented fundamental changes in planktivore, zooplankton, and phytoplankton populations when piscivores have been reduced or overstocked (e.g., Lazzaro et al. 1992, Olrik et al. 1964, Gophen et al. 1990, Elser and Carpenter 1988, Mills et al. 1987, Hartig et al. 1991). An underlying assumption of the concept is that predators and prey at all trophic levels exert controls on each other in a balanced system, but when drastic changes are made to the top of the food chain (top piscivore), controls at lower trophic levels either become excessive or very weak. When abundance of the top piscivore is brought back to premanipulation densities, the lower trophic levels come into balance again. In classic lake examples, overstocking the piscivores quickly results in very clear waters whereas overfishing the piscivore stocks results in a lake turbid with algal blooms. Food web management strategies for freshwater lakes that incorporate principles of the trophic cascade effect can probably be applied directly to tidal freshwater regions in the Bay area because their planktivore/mesozooplankton relationships appears to be identical to those found in lakes, that is, an inverse relationship. Development and maintenance of a sizable piscivore population (e.g., largemouth bass) in tidal fresh reaches that are otherwise balanced (stable, relatively moderate nutrient loadings; acceptable dissolved oxygen levels; no toxicity) can be expected to bring planktivore abundances down, and thereby raise mesozooplankton and ichthyoplankton abundances, increase grazing pressure on the phytoplankton, and increase the transfer of organic material to higher trophic levels.

The positive regression slopes between planktivores and mesozooplankton in the oligohaline, and the apparently complex relationship between the mesozooplankton and a diverse array of vertebrate and invertebrate predators in the mesohaline, suggest that food web management strategies developed for freshwater lakes may not be directly transferable to oligohaline and mesohaline waters. Trophic relationships in these complex and much more dynamic salinity regimes need to be further explored and documented before legitimate food web management strategies can be proposed. These salinity regimes would probably benefit from increased mesozooplankton abundances in the tidal fresh, however. Larger zooplankton populations in the tidal fresh would generate a better food base in higher salinity regimes for larval and Y-O-Y fish, which use these areas as nursery grounds, as well as for planktivores.

Other avenues of investigation remain to be explored. First, only the juvenile summer seine surveys were used in this study. There are a number of trawl surveys, done throughout the Bay during different seasons, whose data would allow a better understanding of zooplankton linkages with more open water fish communities. Second, plankton/fish linkages during the summer are evident in ways other than straightforward regressions between planktivores and their prey, the mesozooplankton. For example, finfish planktivory elicits specific changes in zooplankton size frequency distributions, abundance of invertebrate planktivores, and prey vulnerability responses. Finfish and invertebrate planktivores have very different relationships with their prey in estuarine waters, and each can possibly obscure the effects of the other in simple regressions, such as was done for this paper. Further investigations will hopefully substantiate the plankton/fish linkages at some stations and identify environmental variables that are disrupting the linkages at other stations.

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POPULATION FLUCTUATIONS, DENSITY DEPENDENCE ,AND STOCHASTIC VARIATION IN THE BLUE CRAB

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Abstract: Population abundance of the blue crab, *Callinectes sapidus,* fluctuates widely in Chesapeake Bay. We describe a comprehensive approach to the analysis of population variation in the blue crab; the approach is based on a stage-specific analysis of regulatory and stochastic processes driving survival and abundance of the component stages in the life history. We postulate that stochastic factors are primarily important in the larval and postlarval stages, whereas density-dependent processes such as predation and canabalism regulate the juvenile phase. Analysis is based on the collective evidence from field and laboratory experiments, and long-term field sampling of postlarvae, juveniles, and adults in Chesapeake Bay. These patterns and processes are related to population dynamics, habitat utilization, and recruitment relationships in the blue crab.

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CHESAPEAKE BAY SUBMERGED AQUATIC VEGETATION DISTRIBUTION, ABUNDANCE, AND HABITAT QUALITY TRENDS: 1971-1991

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Abstract: Submerged aquatic vegetation (SAV) historically contributed to the high primary and secondary productivity of Chesapeake Bay but experienced a dramatic Baywide decline in the late 1960s and 1970s. Since the first Baywide SAV survey in 1978, total distribution of SAV in Chesapeake Bay and its tributaries has increased by more than 50%. Along with the increase in SAV distribution between 1984 and 1991 was a concomitant increase in overall density of many SAV beds. Building on the two decades of aerial and ground survey SAV distribution data, the development of SAV habitat requirements, the establishment of SAV restoration goals and targets, the compilation of historical water quality data, and the implementation of a Baywide monitoring program, this paper describes trends in SAV distribution and abundance in Chesapeake Bay and its tidal tributaries from 1971-1991; relates SAV distributions over time to tiered distribution restoration goals and targets; relates trends in SAV distribution to corresponding trends in water quality, and correlates patterns of SAV distribution with existing relevant meteorological data. Patterns of change in SAV populations throughout Chesapeake Bay were complex, varying both in space and time. This complexity is attributable to differing characteristics of the major watersheds of the Bay, meteorological differences, and differences in the biology of species present. To further describe Bay-wide trends, patterns of SAV distribution from 1984 to1991 (and from 1971 to1991 when data were available) in all Chesapeake Bay segments have been characterized and assigned into one of the following five categories: increasing trend, fluctuating at high levels, fluctuating at low levels, decreasing trend, and little or no SAV.

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HIGHLIGHTS OF THE CHESAPEAKE WATERSHED WETLAND TRENDS STUDY

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Abstract: In late 1991, the U.S. Fish and Wildlife Service (FWS), with support from its National Wetlands Inventory Project, initiated a wetland trends study for the Chesapeake Bay drainage basin. The study examined wetland status and trends between 1982 and 1989 for the 63,000-square mile Watershed. The study was designed to generate watershed-wide estimates that update a previous study that estimated wetland trends from the mid-1950s to the late 1970s/early 1980s. In addition, the study would examine specific changes in several geographical areas in three states (Virginia, Maryland, and Pennsylvania). The watershed estimates were derived by statistically sampling 4 sq mi plots randomly selected in the watershed. Through photointerpreting wetlands and recent trends in these plots, wetland acreage by major type and on corresponding changes were estimated for the entire watershed. An estimated 5.2 million acres of wetlands and deepwater habitats were present in the Chesapeake watershed in 1989. Of this, 1.7 million acres were wetlands, with palustrine wetlands being most abundant (1.46 million acres). Nearly 4% of the watershed was covered by wetlands. From 1982 to 1989, estuarine vegetated wetlands declined by about 1,000 acres, while palustrine vegetated wetlands fell by over 22,000 acres. In contrast to the decline of vegetated wetlands, pond acreage increased by almost 6,000 acres. The study also examined wetland trends for 25 U.S. Geological Survey topographic maps in Virginia, 30 in Maryland, and 31 in Pennsylvania. These areas were selected by the U.S. Environmental Protection Agency and FWS with state input. Study highlights are represented in this paper.

INTRODUCTION AND DISCUSSION

In 1992, the U.S. Fish and Wildlife Service (FWS) received funds internally from the FWS's Chesapeake Bay Estuary Program and from the U.S. Environmental Protection Agency's (EPA) Chesapeake Bay Program to conduct a study of wetland trends in the 63,000 sq mi Chesapeake watershed. The purpose of this study was to estimate the changes in wetlands that occurred during the 1980s. This information would be used to evaluate existing wetland policies and to develop new strategies to improve wetland protection, as necessary. A technical report (Tiner et al. 1994b) and an executive summary report (Tiner 1994) have been published to present study findings to natural resource agency managers, regulatory personnel, the private sector, and others. In addition, a public information booklet is being prepared to condense the study results into a medium that the general public can understand. The present paper summarizes some of the more pertinent findings and present study conclusions/recommendations.

Study Methods

The study was designed to accomplish two primary tasks: (1) to generate watershed-wide estimates of wetland status and trends in the 1980s, and (2) to produce detailed assessments of wetland trends in preselected geographic areas of special interest to federal and state agencies. Phase I of the study used statistical sampling procedures to estimate the wetland status and trends in the Chesapeake watershed and in the portion of each of the six states that encompass the watershed (Delaware, Maryland, New York, Pennsylvania, Virginia, and West Virginia). This work would update results of an earlier study of wetland trends in this region that covered the period: mid-1950s to the late 1970s/early 1980s (Tiner and Finn 1986). This phase involved analyzing changes in individual wetlands in 4 sq mi plot sampling areas. These areas were randomly selected within 22 sampling strata defined by state boundaries, physiographic regions, wetland density, and expected degree of wetland alteration. Wetlands were defined and classified (to the class level) according to the FWS's official wetland classification system. The following wetland types were evaluated: estuarine emergent wetlands (E2EM), estuarine scrub-shrub wetlands (E2SS), estuarine forested wetlands (E2FO), estuarine unconsolidated shores (E2US), palustrine forested wetlands (PFO), palustrine scrub-shrub wetlands (PSS), palustrine emergent wetlands (PEM), palustrine unconsolidated bottoms/unconsolidated shores (PUB/US), and palustrine farmed wetlands (Pf). Estuarine wetlands are salt and brackish tidal marshes and swamps, while palustrine wetlands are freshwater wetlands including both tidal and nontidal types. Nonvegetated wetlands include unconsolidated shores (e.g., tidal mud flats and exposed shores of reservoirs, lakes, and ponds) and unconsoldated bottoms (ponds). Wetland trends were identified as changes in wetland type (e.g., pond to marsh, or forested wetland to scrub-shrub wetland owing to timber harvest), and losses to or gains from lakes/reservoirs, agricultural lands, urban development, rural development and other development. The specific methods are detailed in Tiner et al. (1994b).

To accomplish the second task (phase II), quad-based wetland trend assessments were performed for 15 areas designated by EPA and FWS representatives based on input from state wetland regulatory agencies. This study identified wetlands to the fullest extent possible following the FWS system, with wetlands classified from system level through water regime and special modifiers. Wetland losses were also attributed to specific causes, such as housing development, ditching, roads/highways, and stormwater detention basins. Therefore, the trends results were much more specific than the results of the phase I effort. A series of reports on each geographic area was published (Tiner and Foulis 1993a, 1993b, 1993c, 1993d, 1993e, 1993f, 1993g, 1993h, 1993i, 1994a, 1994b, 1994c, 1994d, 1994e, Tiner et al. 1994a).

Study Highlights

The following two subsections summarize the more significant findings of the phase I and phase II studies of wetland trends in the Chesapeake Bay watershed. For additional information, refer to the specific study reports (Tiner and Foulis 1993a, 1993b, 1993c, 1993d, 1993e, 1993f, 1993g, 1993h, 1993i, 1994a, 1994b, 1994c, 1994d, 1994e, Tiner et al. 1994a, 1994b).

Phase I Highlights: Watershed-wide Results

An estimated 5.2 million acres of wetlands and deepwater habitats existed in the Chesapeake watershed in 1989. Wetlands accounted for roughly 1.7 million acres, covering about 4% of the watershed. This amounts to an area about 1.4 times the size of Delaware or about one-quarter the size of Maryland. Freshwater (palustrine) wetlands are the predominant type, occupying nearly 1.5 million acres, with forested wetlands alone representing 60 percent of the Watershed's wetlands (figure 1). The watershed's wetlands fell within six states: Delaware, Maryland, New York, Pennsylvania, Virginia, and West Virginia (table 1). Approximately two-

Figure 1. Estimated 1989 wetland acreages for the Chesapeake watershed.

Figure 2. Distribution of wetlands in the Chesapeake watershed by state.

Figure 3. Distribution of wetland types for the Chesapeake watershed.

thirds of the watershed's wetlands occur in two states: Virginia (40%) and Maryland (27%) (figure 2). The distribution of wetland types is shown in figure 3.

Between 1982 and 1989, palustrine vegetated wetlands (freshwater marshes, wet meadows, swamps, and bogs) declined by 2% About 36,000 acres were converted to drylands and waterbodies: (1) 14,700 acres of forested wetlands, (2) about 10,600 acres of emergent wetlands, and (3) about 10,700 acres of scrub-shrub wetlands. These collective losses equal an area about the size of the District of Columbia. In addition, about 18,000 acres of palustrine forests were harvested for timber. This, however, is not considered a loss, because these areas are still wetlands that will likely return to forested wetlands in time.

Virginia had the greatest palustrine vegetated wetland losses of any state, losing an estimated total of approximately 23,000 acres: about 4,000 acres of emergent wetlands, over 8,000 acres of scrub-shrub wetlands, and nearly 11,000 acres of forested wetlands during the study period. Maryland lost about 5,000 acres of the palustrine vegetated wetlands during this time: about 2,400 acres of emergent wetlands, about 500 acres of scrub-shrub wetlands, and over 2,500 acres of palustrine forests. Pennsylvania lost almost 4,000 acres of these wetlands: mostly emergent wetlands (over 2,000 acres) and scrub-shrub wetlands (almost 1,700 acres). Causes of palustrine vegetated wetland losses are presented in figures 4, 5, and 6. Reservoir/lake construction was the leading cause of forested and scrub-shrub wetland losses, while agriculture had the greatest adverse impact on palustrine emergent wetlands. Estuarine emergent wetlands were most adversely affected by activities and processes that converted them to open water. Coastal erosion, dredging, rising sea level, and coastal subsidence are the major factors responsible for this wetland loss (figure 7). Table 2 summarizes vegetated wetland trends for the watershed based on wetland type.

Figure 4. Causes of palustrine forest destruction in the Chesapeake watershed. Note: Excludes about 18,000 acres that were harvested between 1982 and 1989.

Table 1. Estimated 1989 wetland acreages in the Chesapeake Watershed by state.

Figure 5. Causes of palustine scrub-shrub wetland destruction in the Chesapeake watershed.

Figure 6. Causes of palustine emergent wetland destruction in the Chesapeake watershed.

Table 2. Changes in specific types of vegetated wetlands in the Chesapeake watershed (1982-89).

** Standard error is equal to less than 20% of the estimated acreage.
* Standard error is less than 50% but greater than 20% of the estimated acreage.
Note: Estimates without an asterisk have higher standard error than the other two categories.

Figure 7. Causes of estuarine emergent wetland destruction in the Chesapeake watershed.

Phase II Highlights: Results for Specific Geographic Areas

Several areas were chosen by federal and state agency representatives for detailed examination of recent wetland trends. These areas may reflect either highly vulnerable areas or other geographic areas of interest to the agencies. Fifteen areas were selected in three states (Virginia, Maryland, and Pennsylvania): (1) Norfolk-Hampton region (VA), (2) Chickahominy River area (VA), (3) northern Virginia, (4) Dorchester County area (MD), (5) lower Eastern Shore (MD), (6) Western Shore (MD), (7) Kent Island area (MD), (8) north east (MD), (9) fall zone (MD), (10) Piedmont region (MD), (11) northeast glaciated region (PA), (12) greater Harrisburg (PA), (13) Williamsport area (PA), (14) Hazelton (PA), and (15) DuBois and Falls Creek (PA). The more significant results of these studies are summarized in table 3. Some areas where little wetland alteration occurred are not included in this table. For additional information, see the individual study reports (Tiner and Foulis 1993a, 1993b, 1993c, 1993d, 1993e, 1993f, 1993g, 1993h, 1993i, 1994a, 1994b, 1994c, 1994d, 1994e, Tiner et al. 1994a).

Table 3. Recent wetland trend highlights for selected areas in the Chesapeake watershed.

Wetland Loss Hotspots

The study identified seven areas in the Chesapeake Watershed where wetlands experienced enormous losses between 1982 and 1989: (1) southeastern Virginia, (2) Piedmont region of Virginia, (3) Eastern Shore of Maryland, (4) western Delaware, (5) Upper Coastal Plain of Virginia, (6) western Virginia (Blue Ridge and Appalachians), and (7) northeastern Pennsylvania. table 4 summarizes the general scope of the problem. These areas are in need of increased wetland protection if wetland functions are to be maintained.

Table 4. Summary of wetland losses in hotspots in the Chesapeake Bay watershed (1982-1989).

Hotspot Area	Palustrine Types Affected	Acres Converted to Dryland and Waterbodies	Major Causes of Losses
Southeastern Virginia	Forested Wetland	Over 2,000	Housing and Agriculture
Piedmont Region of Virginia	Vegetated Wetlands Scrub-Shrub Forested Emergent	About 17,000 ³ 7,500 ³ 7,000 ³ 2,200	Reservoir-Lakes, Construction and Pond Construction
Eastern Shore of Maryland	Vegetated Wetlands (mostly Forested and Emergent)	Over 4,000	Agriculture, Pond/Reservoir-Lake Construction and Urban Development
Western Delaware	Emergent Wetland	Almost 2,000	Agriculture
Upper Coastal Plain of Virginia	Vegetated Wetlands (mostly Forested)	Almost 2,000	Pond Construction
Western Virginia	Emergent Wetland	Almost 1,500	Agriculture
Northeastern Pennsylvania	Emergent Wetlands	About 1,300	Pond Construction and Agriculture

CONCLUSION AND RECOMMENDATIONS

Overall, the status of estuarine wetlands (salt and brackish tidal marshes) has improved. Prior to the enactment of state coastal or tidal wetland laws and strengthened federal regulation under the Clean Water Act, these wetlands were dredged and/or filled at high rates. The current study suggests that increased state and federal wetland regulation has improved the condition of these wetlands. They are no longer being wantonly destroyed. There is still pressure to convert them to alternative uses, but landowners, developers, and the general public realize the values of these wet areas and are fully aware of government programs to regulate activities in and/or protect these wetlands.

The situation for palustrine vegetated wetlands was quite different. These wetlands continue to be destroyed at alarming rates. Despite the existence of Federal regulations, nontidal freshwater wetlands continued to experience heavy losses. There was a 12-fold increase in the net annual loss rate of forested wetlands. From 1982 to 1989, the annual loss rate was about 2,000 acres versus almost 200 acres during an earlier period (mid 1950s -1980s). Much of this forested wetland "loss" resulted from increased timber harvest during the study period. In managed forests, this "loss" of forested wetlands is usually not a loss of wetland, but simply a temporary change in the wetland type. The emergent and scrub-shrub wetlands resulting from timber harvest are successional types that eventually become forested wetlands. Other harvested forested wetlands, however, may be converted to other uses. Almost 15,000 acres of palustrine forests were destroyed through conversion to drylands and to open waterbodies (e.g., reservoirs and ponds). In addition, 21,000 acres of vegetated wetland losses involved emergent and scrub-shrub wetlands. It is evident that wetland regulations must be improved if we are to protect our remaining wetlands.

Based on the statistical analysis, seven areas were identified as wetland loss hotspots where tremendous losses of certain wetland types occurred between 1982 and 1989: (1) southeastern Virginia, (2) the Piedmont region of Virginia, (3) the Eastern Shore of Maryland, (4) western Delaware, (5) the upper coastal plain of Virginia, (6) western Virginia (Blue Ridge and Appalachians), and (7) northeastern Pennsylvania (Susquehanna, Bradford, and Tioga Counties). These areas accounted for about 85% of the palustrine vegetated wetlands that were converted to drylands and waterbodies during the 7-year study period. Wetland protection efforts should be strengthened in these areas.

The following recommendations are offered to help improve the status of wetlands in the Chesapeake watershed. Some of the suggestions are specific to the watershed, while most are of a general nature applicable to many areas in the eastern United States and elsewhere:

- Develop and adopt strategies to increase protetion of palustrine vegetated wetlands, especially for seasonally saturated and temporarily flooded wetlands and isolated wetlands on the coastal plain and for the States of Virginia andDelaware. Such strategies must address agricultural uses of wetlands, since such activities continue to be major causes of wetland loss in the Watershed. Other activities that need to be included in these strategies are aquaculture, regulated shooting areas, and forestry practices in wetlands.
- Interpret the regulatory definition of wetland in a scientifically sound manner and use science based techniques to identify these wetlands on the ground. Use policy to regulate uses of wetlands and not to define what a wetland is. It is more efficient and effective to change policy to meet current needs than to try to change established scientific principles and practices to satisfy a public policy need.
- In southeastern Virginia, where palustrine vegetated wetlands are disappearing at an alarming rate, it may be advisable to establish an intergovernmental committee (federal, state, and local) to develop a regional strategy for reducing wetland losses while pursuing realis tic economic growth. This is perhaps the greatest challenge for similar "wetland loss hotspots" in the Chesapeake watershed and elsewhere in the country. It may require developing innovative tax incentives and wetland acquisition initiatives and establishing realistic land use options and growth/develop ment limits that maintain and enhance existing environmental quality. The 1988 report entitled *Population Growth and Development in the Chesapeake Bay Watershed to the Year 2020(Year 2020 Panel 1988)* provides insight into the problems and the vision of how this may be accomplished. This report offers many specific recommendations that should be implemented to maintain a high quality environment in the watershed.
- Eliminate government-sponsored wetland channelization and ditching programs and seek other more environmentally acceptable means of reducing flood damages (e.g., natural valley storage approach).
- Locate stormwater detention basins and agricultural sediment ponds outside of wetlands and of streams. With increasing urban development, stream flows increase leading to accelerated erosion of streambanks and stream beds. Proper location of these basins should minimize wetland and stream impacts.
- Increase wetland acquisition to preserve unctions of existing wetland systems. Identify large tracts of remaining wetlands and strive to connect them together, thereby linking presently isolated tracts into an interconnected network of wetlands. This effort attempts to minimize wetland fragmentation for improved wildlife habitat and should enhance other wetland functions as well.
- Identify wetland landscapes in need of restoration and initiate large-scale restoration efforts to restore ecosystem functions.
- Develop measures and programs to maintain and establish vegetated buffers around wet-

lands and along waterbodies. This could produce significant water quality benefits and enhance wildlife habitat values.
- Instead of wetland trend studies, develop and initiate monitoring programs to provide more real-time assessment of wetlands for analyzing and modifying current policies before too much wetland destruction occurs.
- Conduct research to increase our knowledge of the hydrology and functions of seasonally saturated wetlands and isolated temporarily flooded wetlands on the coastal plain.
- Develop outreach programs to encourage private landowners to protect their wetlands and/or to minimize wetland alteration during activities such as timber harvest.
- Continue to increase public education efforts. A well-informed public will likely select environmentally sound approaches to land use in the future.

Wetlands are the vital link between land and water. As such, they help improve water quality, temporarily store water to prevent downstream flooding, stabilize shorelines, and provide numerous other functions that benefit society. If we are to continue to receive these benefits, action must be taken to reverse the trends observed in the 1980s and earlier. We must continue our efforts to conserve estuarine wetlands that significantly slowed the losses of these wetlands. Our attention must now focus on the nontidal palustrine wetlands which remain under heavy threat for development. The living resources of Chesapeake Bay also depend on the welfare of these wetlands, that help filter out excess nutrients, sediments, and other pollutants, thereby preventing these potentially deleterious materials from reaching the Bay. Our quality of life is largely dependent on the abundance and condition of natural resources. We must strengthen wetland protection and initiate wetland restoration efforts to improve the quality of the Bay for its living resources, for ourselves, and for future generations. The significance of our land and water resources should not be underestimated. Based on the past experience of other civilizations, how we manage our natural environment will largely determine the fate of our society.

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STATUS AND TRENDS OF FOUR KEY FINFISH SPECIES IN CHESAPEAKE BAY: ACHIEVING SUSTAINABILITY WITH FISHERIES MANAGEMENT

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Abstract: This paper presents recent stock assessment information for bluefish, weakfish, American shad, and summer flounder. These species were chosen because (1) they are representative of four different groups of species found in Chesapeake Bay (2) the abundance of each has declined from historic levels in recent years (3) interstate management plans for each are being rewritten and/or implemented under new federal law and (4) they support or have supported economically important commercial and recreational fisheries in Chesapeake Bay.

The current status of these stocks as well as recent trends are described based on landings data, recruitment indices, spawning stock biomass estimates, and fishing mortality rates. These data guide management decisions under the fishery management plans for each species developed by the Atlantic States Marine Fisheries Commission. States party to these plans will have to fully implement them between now and March 1995 under the requirements of the Atlantic Coastal Fisheries Cooperative Management Act of 1993 or face a federal moratorium on the species in question.

INTRODUCTION

The living resources of Chesapeake Bay are the most tangible object of the public interest and of the various Bay cleanup activities. When they talk about the Bay, citizens always ask about the rockfish, the oysters, or the Bay grasses. There is great concern about pollution, but it is primarily motivated by the desire to maintain habitat for living resources.The living resources of the greatest public interest are those species that support commercial and recreational fisheries. Fisheries provide a direct economic benefit and are central to the rich cultural traditions of the Bay area (Norton et al. 1983, Horton 1992).

The theme of this conference is sustainability. Conference organizers have posed the question, “How can we achieve it and how can we maintain a productive estuary in spite of continuing growth and development in the coastal areas?” The focus is on human activities in the watershed that affect the Bay and presumably the habitat for the Bay’s living resources. While man’s ability to pollute the Bay and destroy aquatic habitat is well documented (Macalaster et al. 1983) and the need to address this problem is paramount, recent trends in fisheries indicate that the maintenance of a productive estuary also depends on good fisheries management. The striped bass experience clearly demonstrated man’s ability to both deplete a valuable fishery resource through overfishing and restore a fishery through applied fishery science (Atlantic States Marine Fisheries Commission 1994). The condition of four other key fisheries indicates a need to apply this lesson on a broad scale.

This paper looks at the status of four finfish species, weakfish, bluefish, summer flounder, and American shad, that either support or have supported important fisheries in Chesapeake Bay. These species represent four groups of species that use the Bay in various ways and have different estuarine habitat needs. The paper examines habitat requirements, fishery trends, and recent stock assessment information to shows

1. That fisheries management is an essential complement to habitat management for maintaining sustainability in Chesapeake Bay.
2. That fishery science must play a central role for fisheries management to be effective in maintaining sustainability.
3. That fisheries management must be coordinated throughout the range of a fish stock to be effective.

Weakfish

Weakfish belong to the sciaenid family of bottom-feeding fish that includes spot, croaker, and drum (Mercer 1989). They are found from Cape Cod to Florida on the Atlantic coast and spawn near the mouths of estuaries, including Chesapeake Bay. Weakfish utilize estuaries as juvenile nursery grounds and adult feeding grounds. Primary estuarine habitats are shallow water grass beds for juveniles and grass beds and other benthic communities for adults.

Weakfish support commercial and recreational fisheries with the majority occurring in the mid-Atlantic region. Catches in both Maryland and Virginia have declined markedly in the last decade (figures 1 and 5). Atlantic coast weakfish are managed as one unit stock under a fishery management plan (FMP) of the Atlantic States Marine Fisheries Commission (ASMFC) (Atlantic States Marine Fisheries Commission1985, Mercer 1985). The plan calls for the maintenance of sufficient spawning stock and promotes increased yield-per-recruit, but it has left the determination of specific catch restrictions up to the states.

Figure 1. Commercial catch for weakfish in the mid-Atlantic region.

In response to a continued decline in weakfish landings the plan was amended in 1992, and a schedule was adopted for reducing exploitation over a 5-year period (Seagraves 1991). The schedule called for a 15% cutback in 1992, increased to 25% in 1993 and 1994, and 50% thereafter. The record of implementation of this amendment by member states, including Maryland and Virginia, led the ASMFC FMP Review Team to report in November 1993,". . . all effects by states have been inadequate to bring about any start of recovery of the stock" (Atlantic States Marine Fisheries Commission 1993). No state was deemed in compliance with the 25% reduction called for in the amendment.

In part owing to concerns about weakfish, Congress passed the Atlantic Coastal Fisheries Cooperative Management Act (ACFCMA) in late 1993. The ACFCMA required states to implement fishery plans of the ASMFC or face a federal moratorium. Prior to that, all ASMFC plans except that for striped bass had been advisory, and the states' record of compliance was poor. In accordance with the new law, the ASMFC has adopted a schedule for compliance with all existing plans. The 25% exploitation cutback called for in amendment 1 of the weakfish plan is scheduled to be implemented by March 1995, two years after originally planned.

The latest stock assessment for weakfish reported in February 1994 describes the stock as "severely overfished" (Rugolo 1994). A contraction of both the length frequency and the range of the population has been observed. The spawning potential is down to 2%-3% of the maximum for an unfished stock. The target for stock recovery is 20%, which is estimated to require a fishing mortality (F) of 0.22. The current fishing mortality is 1.23 meaning an 82% reduction will now be necessary. There is a high short-term probability of recruitment failure. Given the worsening conditions, the ASMFC is beginning to develop another amendment to the weakfish plan to lay out alternative management strategies for adoption in 1995.

Bluefish

Bluefish is a highly migratory pelagic species that ranges along the entire U.S. Atlantic coast. It spawns offshore and utilizes the open waters of estuaries as feeding grounds for adults (Pottern et al. 1989). Bluefish was the predominant species harvested by marine recreational fishermen from 1979 to 1987 (Mid-Atlantic Fishery Management

Council 1989). Approximately 10% of the annual historical catch has been commercial. Bluefish range into federal waters (3-200 miles offshore) as well as inshore state waters and are managed as a unit stock under a joint FMP of the ASMFC and the Mid-Atlantic Fisheries Management Council (MAFMC). The principal management measures in the plan are a ten-fish creel limit for recreational anglers and a provision that the commercial catch shall not exceed 20% of the total. The ten-fish creel limit has not been uniformly implemented by member states.

Coastwide recreational landings declined approximately 70% between 1983 and 1992 (Northeast Fisheries Science Center 1993) (see figure 5) Although socio-economic data for the recreational and charter fisheries are not readily available, widespread anecdotal information suggests substantial losses as a result of this decline (see figure 2). This trend is reflected in the spawning stock, which is estimated to have declined from 300 million pounds in 1982 to 81 million pounds in 1989 (ASMFC 1993). While commercial catches have remained stable, they now make up 20% or more of the total, and a commercial quota system is being implemented as called for in the plan.

Although current stock assessment data is incomplete, fishing mortality is believed to have increased sharply since 1989 to approximately double the target level of 0.3 and may be substantially higher owing to unreported commercial hook-and-line catch (Crecco pers. comm.). Although bluefish are officially listed as “fully exploited” (Northeast Fishery Service Center 1993), scientists increasingly believe this stock to be overfished and may soon recommend further catch restrictions (Crecco pers. comm.).

Summer Flounder

Summer flounder are found throughout Atlantic coastal waters but are most abundant in the mid-Atlantic. They spawn in fall and winter on the continental shelf, and larvae and juveniles are swept into estuaries in late winter and spring where they utilize seagrass beds as nursery areas (Grimes et al. 1989). Adults also use estuaries as feeding grounds concentrating in inlets and along channel edges. Summer flounder are managed as a single stock under a cooperative management plan between the ASMFC and the MAFMC adopted in 1988 and amended several times (Mid-Atlantic Fishery Management Council 1988).

Summer flounder are highly sought after by both commercial and recreational fishermen, with recreational anglers historically accounting for about 40% of total landings. Recent analyses have shown that fishing mortalities have been very high during the last decade, peaking at 1.8 in 1988-89, well above the overfishing level of 0.23 (Northeast Fisheries Service Center 1993). Spawning stock biomass set a record low in 1989. These developments and a sharp drop in commercial and recreational flounder landings in the late 1980s (figures 3 and 5) precipitated the adoption of Amendment 2 to the plan in 1991 (Mid-Atlantic Fishery Management Council 1991). Amendment 2 was designed to lower fishing mortality to 0.53 in 1993, 1994 and 1995 and to 0.23 thereafter. It prescribed a series of management measures, including quotas, seasons, gear restrictions, minimum sizes, and creel limits.

With fairly close adherence to quotas, the preliminary indication is that the fishing mortality dropped from 1.1 in 1992 to the target of 0.53 in 1993. Spawning stock biomass and recruitment have increased in recent years, and, although the age structure is still truncated, quotas were able to be increased in 1994. The outlook is good for rebuilding the summer flounder fishery, provided that fishing mortalities are contained at target levels.

Figure 2. Commercial catch for bluefish in the mid-Atlantic region.

Figure 3. Commercial catch for flounder in the mid-Atlantic region.

American Shad

American shad are a highly migratory anadromous species ranging from Florida to Canada but historically most abundant in the mid-Atlantic (MacKenzie et al. 1985). Shad undertake extended seasonal ocean migrations but return to their river of origin to spawn in the spring. Thus, shad form individual river populations, with every major Atlantic coast river at one time supporting a stock (National Fisheries Service Center 1993). American shad tend to spawn in relatively shallow flowing water over sandy bottom, but they have been observed in a variety of riverine habitats (MacKenzie et al. 1985). Juveniles are found in upper Chesapeake Bay waters during the summer before their fall migration to marine waters. Documented habitat problems for shad include blockage of spawning rivers by dams and other obstructions and water pollution in spawning areas (Chesapeake Bay Program 1989).

American shad have historically supported important commercial and recreational fisheries in Chesapeake Bay. Early in this century, shad were the most valuable commercial finfish landed in the Bay (Hildebrand and Schroeder 1972). It has been estimated that the current dockside value of that historical catch would be $8.5 million (Chesapeake Bay Foundation and the U.S. Fish and Wildlife Service 1994). Tales of the sport fishery for shad are legendary as exemplified by this quotation from a 1957 Baltimore newspaper: "Ever since the first fisherman sank a hook into the bony jaw of a jump-crazy shad, this sport of kings and commoners alike has grown by geometric progression." American shad have declined dramatically in Chesapeake Bay during the last 20 years (Chesapeake Bay Program 1989). There is virtually no sportfishery for shad now, and moratoria on commercial fishing are in place in both Maryland and Virginia waters of the Bay. Commercial landings listed for the states are from the Atlantic Ocean, where migrations of stocks from various rivers are intercepted (figure 4).

Shad are managed under an interstate plan (Atlantic States Marine Fisheries Commission 1985). The plan recognizes the individual nature of specific river populations and lists recommended exploitation rates for each. Adherence to these recommended rates has been inconsistent. The plan also recognizes the priority rights of the traditional fisheries in the rivers of origin and discourages ocean intercept fisheries. In spite of this recom-

Figure 4. Commercial landings of American shad.

mendation, most member states still allow ocean harvesting. Recent analysis has corroborated concerns that ocean fisheries target mixed stocks of shad including healthy and depleted stocks (Brown and Chapman 1991). Under these circumstances data has not been available to estimate fishing mortalities on Chesapeake shad, but the stocks are considered severely depleted.

The ASMFC has recently decided to draft a new FMP for American shad with support from the Chesapeake Bay Program and the Maryland Department of Natural Resources. The new plan will have precise management recommendations for implementation in early 1995.

Figure 5. Estimated total recreational catches for weakfish, bluefish, and flounder in the mid-Atlantic region.

DISCUSSION

The recent history of these four species illustrates the high degree to which the sustainability of Chesapeake Bay living resources depends on effective fisheries management. It describes a pattern of fishing pressure growing until it becomes excessive, a slow response from management agencies, dramatic declines in stocks and substantial losses to the fisheries. Several factors bring about this pattern, and several solutions are suggested. Chesapeake Bay fisheries management has historically been politically driven. Science has taken a back seat to the desires of constituent fishermen. The best example of this phenomenon are the turn-of-the-century disputes over oysters that bordered on open warfare (Wennersten 1981). Fisheries decisions typically have been stalled first by a reluctance to act until an obvious crisis and second by disagreement over whether the cause of a given fish stock decline was overfishing, pollution or natural cycles, all of which influence Chesapeake Bay stocks.

The four examples here demonstrate the ability of science to assess stock condition and document declines. Analyses of spawning stock biomass, recruitment, and fishing mortalities, as well as other approaches, can provide managers with reliable assessments as data become available. Scientists correctly documented excessive fishing mortalities and declining spawning stock for weakfish. Precise recommendations for harvest control were not widely implemented, and the stock declined further. In the case of summer flounder, a science-based prescription for management is being implemented, and there are encouraging signs of stock recovery. An even better example is the recovery of striped bass which,

Figure 6. Estimated Atlantic coast female striped Bass spawning stock biomass (SSB).

after several years of strict adherence to a science-based restoration plan, is expected to reach the recovery target for the spawning stock in 1995 (figure 6).

These examples also emphasize the need for management to be coordinated throughout the range of a stock. In the cases of bluefish and weakfish, partial implementation of management recommendations by states party to those plans did not lead to stock recovery. The experience with Chesapeake stocks of American shad clearly demonstrate this problem. Maryland has had a moratorium on the harvest of shad in place since 1980, yet harvest has continued on Maryland stocks in neighboring states and along the coast, and the stocks have not rebounded. States' inability to implement uniform management for striped bass led Congress to enact the Atlantic Striped Bass Conservation Act in 1984, requiring states to comply with the ASMFC striped bass plan. Range-wide implementation of that plan has brought about a dramatic recovery. Congress followed its own lead on this point when it enacted the ACFCMA in 1993, extending the requirement for range-wide compliance to all other species covered by ASMFC plans.

Separating the influences of fishing, habitat, and natural cycles is a challenge, especially for a temperate estuary like Chesapeake Bay. However, the occurrence of declines in all four of these species at the same time minimizes the likelihood of natural cycles as a causative factor. These species also have differing estuarine habitat needs and utilize these habitats at different times of the year (table 1). It is highly unlikely that degradation of all these habitats during the last decade would have occurred and caused the recent declines in these species. Furthermore, striped bass, which requires many of the same habitats, has dramatically increased in abundance during this period. The main difference between striped bass and these species is that striped bass has been the object of science-based, range-wide management.

The tolerance of overfishing and fish stock declines that has typified Chesapeake Bay fisheries management historically directly undermines the goal of sustainability. The unwritten policy of avoiding active management until crises occur, even if the ultimate action is strong, results at best in widely fluctuating fishery resources and unstable social and economic benefits. Effective fisheries management that acts to avoid crises and maintain healthy fish stocks is an essential complement to habitat management in the effort to achieve sustainability. Congress has put in place a proven framework for fisheries management (ACFCMA). All those with an interest in maintaining sustainability in Chesapeake Bay living resources should promote a leadership role for Bay jurisdictions in that interstate fisheries management process.

Table 1. Life cycle and important estuarine habitat for weakfish, bluefish, summer flounder, and American shad in Chesapeake Bay.

	Life Cycle Dependence	Important Habitats
Weakfish	juvenile nursey, adult feeding	Grass beds benthos
Bluefish	Adult feeding	open waters
Summer Flounder	juvenile nursey, adult feeding	Grass beds benthos
American Shad	spawning	riverine

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WATERFOWL POPULATION TRENDS IN THE CHESAPEAKE BAY AREA

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Abstract. The historic use of Chesapeake Bay as a wintering area for large numbers of swans, geese, and ducks has often been used by biologists and managers to demonstrate the importance of this estuary to migratory waterfowl.

Although wintering habitat is essential, breeding habitat in northern areas is also necessary to maintain abundant Bay waterfowl populations. Aerial Bay-wide surveys conducted in January of each year by the U.S. Fish and Wildlife Service in cooperation with wildlife agencies in Maryland and Virginia reflect changes in the numbers of each species and the habitats being used. Major declines in most species of Bay ducks since 1950 parallel population changes in the Atlantic Flyway and the United States. These changes may be caused by loss of nesting habitat and the degradation of wintering grounds from increasing human populations. Bufflehead and goose populations have increased and swans, mallards, scaup, and sea ducks have been stable. In recent years, however, Canada goose and sea duck populations in Chesapeake Bay have declined, which may be related to overharvesting. Mallard population increases since 1985 are most likely attributable to the large releases of captive-reared ducks in Maryland. Canada and snow goose population increases during the 1950-80s resulted from their increased use of agricultural fields for feeding. Waterfowl populations can be maintained at current levels through manipulation of hunting regulations, but major increases of waterfowl, especially ducks, should not be expected until improvements occur in the quality of waterfowl habitat in Chesapeake Bay. Major changes in the management of waterfowl may be necessary owing to increasing human populations in the Chesapeake Bay area.

INTRODUCTION

Chesapeake Bay has received more fame and recognition as a waterfowl use area than any other area in the world. Explorers, naturalists, artists, and hunters were attracted to this huge estuary and marvelled at the magnitude of waterfowl populations that existed there. For 300 years the populations seemed inexhaustible, but by 1900, the huge flocks of ducks, geese, and swans were essentially gone. One species of waterfowl (Labrador duck, *Camptorhynchus labradorius*) had become extinct and warnings were made of more extinctions if greater protection was not forthcoming (Grinnell 1901, Forbush 1913, Salyer 1934).

The killing of waterfowl for sport and market, combined with the steadily increasing human population in this country, had taken its toll. Numerous efforts to reverse the downward trend in waterfowl abundance were initiated around the turn of the century, culminating in the passage of the Migratory Bird Treaty Act of 1918. This act put all migratory birds under the control of the federal government and protected all species. Hunting regulations for certain species whose numbers could withstand some harvest were established by the federal government with guidelines to the states. This paper reviews the exciting and sometimes tragic history of waterfowl populations in Chesapeake Bay and examines waterfowl population data from aerial midwinter surveys conducted from 1950 to 1994.

HISTORICAL REVIEW

The expeditions of Captain John Smith to the New World in the early 17th century provide some of the earliest reports on the number and types of waterfowl that inhabited Chesapeake Bay (Arber 1910). Unfortunately, early explorers and naturalists during the first three centuries of Chesapeake Bay activities did not provide accurate quantified information. They were either unable or unwilling to estimate numbers of birds, although much time and effort were expended on describing the species they saw. The early reports, however, do provide anecdotal information on the vast numbers of waterfowl that obviously existed here. All reports seem to be in agreement that the flocks seemed inexhaustible, but then sadly were exhausted owing to exploitation by humans.

Captain John Smith reported, "In winter there are great plenty of swans, .geese, brants, ducke, wigeon. But in sommer, not any, or a very few to be seene." On his journey to the Pamunkee in 1608, he stated that 148 fowl were killed by two men with three shots. The success or failure of the early colonists seemed to be dependent on their abilities to take advantage of the vast natural resources, including waterfowl (Matthiessen 1959).

Waterfowl also were killed for sport by the colonists, especially the wealthy. The shooting of waterfowl for sport started most likely in the Chesapeake Bay region in the early 1600s (Elman 1980). Letters written by the sport-loving George and Leonard Calvert of Maryland in 1633 spoke rapturously of the infinite numbers of birds for profit and pleasure. The first American work of art to incorporate the theme of wildfowling as a sport was probably a portrait of Thomas Mifflin done by Benjamin West in the 1750s. The well-dressed 14-year-old boy, who later became governor of Pennsylvania, is holding amuzzleloadeding gun and has several dead ducks at his feet. Other art pieces can be used to date the use of dogs in hunting, the type of guns, and the technique employed.

The first concern of waterfowl conservation in the colonies was given in the mid-1600s by William Bradford, governor of the Plymouth Colony, who foresaw a waterfowl decline (Elman 1980). Governor Bradford was a strong advocate of the need to live off the land, but recognized that doing so required maintaining productive habitat. In 1710, Massachusetts banned the use of boats, sailing canoes, or camouflaged canoes in the taking of waterfowl (Elman 1980).

In 1730, Maryland prohibited the taking of waterfowl and other game at night with a gunning light (a practice called firelighting), and Virginia outlawed the practice in 1792. Firelighting was a commonly used hunting technique through the late 1800s. Maryland was one of the last states to outlaw wildfowling in the spring, sale and export of wild game, and unlicensed hunting. The use of sinkboxes (submersible blinds) was outlawed in Maryland in 1935.

The explorations and writings of John James Audubon in the early 1800s provide a fairly good description of waterfowl in the Bay during this period (Audubon 1840). He stated that "innumerable ducks fed in beds of thousands, or filled the air of Chesapeake Bay; and that great flocks of swans, looking like banks of snow, rested near the shores". Audubon also stated that 40-50 ducks could often be killed with one shot with a small gun in Chesapeake Bay. Dr. J. J. Sharpless, writing to Audubon in the early 1800s, stated that, "the quantity of fowl of late years has been decidedly less than in times gone by; and I have met persons who have assured me that the numbers have decreased one-half in the last 15 years."

Some authors (Sanford et al. 1903) reported that wildfowl up to 1860 had not been hunted much and were unmolested during the Civil War. But they reported that "from 1865-90 the greatest natural home in the world for wild ducks has been nearly devastated of its tenants." Phillips (1925) appears to disagree on when the decline began and stated that "the tremendous persecution, which went on there both by night and by day in waters that were easily accessible for shooting, resulted in a steady diminution, which began long before the time of Wilson and Audubon."

There are numerous examples that indicate that mortality of waterfowl before the Civil War was excessive. Lewis (1855) reported that one man (W.W. Levy) killed 187 ducks in one day, that 7,000 canvasbacks were killed in the 1846-47 season, and that gunners at Havre de Grace killed 3,000 ducks on the first day of the 1854 season. Slaves were fed so many canvasbacks that they rebelled and refused to work unless given pork (Leffingwell 1890), and some contracts of slaves that were hired out stipulated the frequency with which canvasbacks could be fed to them(Grinnell 1901).

Increasing human populations in the United States in the 1800s resulted in agricultural expansion in the central part of the country. The first Swamp Act of 1849 resulted in the drainage of 70 million acres of breeding habitat for ducks so these areas could be farmed. Timber cutting along rivers and lakes in the East decimated tree-nesting species such as the wood duck (*Aix sponsa*), bufflehead (*Bucephala albeola*), and goldeneye (*Bucephala changula*). Unlike the effects of the gun, habitat losses are usually not recoverable.

The market hunting period in Chesapeake Bay extended from 1865 to 1900 and was responsible for untold numbers of waterfowl that were killed and sent to the major cities for food. Murphy (1882) stated that during the season the Bay was like a battleground, and that over 10,000 people were accustomed to shoot there. Grinnell (1901) reported a record of 500 ducks taken in one day by one gunner.

Losses of Chesapeake Bay waterfowl to market hunting did not occur only on the Bay. Prebble (1908) reported that thousands of whistling (now tundra) swans (*Olor columbianus*) were killed on the breeding grounds for their down feathers. Eggs were removed from nests for food and for use in making chemicals for the developing photographic industry. The effect of egg removal on waterfowl numbers is not totally understood (Elman 1980).

The great decline in waterfowl populations from overharvesting and loss of habitat essentially resulted in the end of market hunting, which was outlawed in 1918 by the Migratory Bird Treaty Act. This did not end the plight of waterfowl. During the early years of the 1900s, an additional 100 million acres of wetlands in nesting areas were drained to make way for farming. Drainage of salt marshes along the coast for mosquito control also had devasting effects on wintering waterfowl habitat. The continental drought of the 1930s was the final assault on waterfowl populations. In the mid-1930s, new regulations, organizations, and surveys were established to protect, support, and appraise waterfowl populations. Aerial surveys, which had begun in the 1930s, were expanded to include all waterfowl species nationwide after World War II, when planes became more available.

METHODS

Data in this report are based on aerial waterfowl surveys conducted by the U.S. Fish and Wildlife Service, (FWS) and cooperating states in early January of each year from 1948 to 1994. Only data from 1950 to 1994 were graphed in this report, as survey boundaries and time expended were too variable before 1950. Chesapeake Bay populations are represented by the combined totals from Maryland and Virginia, and therefore may represent some counts from non-Bay areas. This overrepresentation of Bay numbers is believed to be minor. Counts of two species of scaup (*Aythya* spp.) and three species of scoter (*Melanitta* spp.) are combined by genera. Data from the Atlantic Flyway and United States for each species are presented as a comparison with Chesapeake Bay population trends. All survey data used in this report were obtained from unpublished data in files of the Office of Migratory Bird Management, FWS's in Laurel, Maryland. Details on how surveys are conducted are described by Perry et al. (1981).

Annual rates of change for major species and groups of waterfowl were determined using simple linear regression for three areas (Chesapeake Bay, the Atlantic Flyway, and the United States). Because of known problems with autocorrelation of population count data (Hatfield et al. 1994), only the probability level of 0.01 was chosen for determining statistical significance of these rates. Although some studies (Eggeman and Johnson 1989) have reported biases in the midwinter survey data, the data do have value when looking at long-term trends in local areas such as the Bay (Chesapeake Executive Council 1990). Survey data in graphs are presented as 5-year averages to minimize annual fluctuations and to emphasize long-term trends.

RESULTS

Midwinter Survey

The midwinter survey data during the 45 years from 1950 to 1994 for waterfowl of Chesapeake Bay clearly indicate downward trends for dabbling and diving ducks and upward trends for geese and bufflehead. Swan, mallard, scaup, and sea duck populations were stable in the Bay. In most cases, the trends are similar to those of the Atlantic Flyway and the United States. This indicates that population changes in the Bay in most cases probably represent continental influences on waterfowl populations. There are exceptions to the general trends that are discussed for each species. Average and range of populations are presented in table 1, and rates of change are presented in table 2 for each species and group of species.

Swans

The swans in Chesapeake Bay consist mainly of the tundra swan, although there is a relatively small population of the exotic mute swan (*Cygnus olor)*. Swan species are usually not separated by species when counted on the midwinter aerial surveys, although separate surveys for mute swans enable biologists to follow trends of this nonmigratory species (Reese 1969, 1975, 1980). Tundra swans have been stable in Chesapeake Bay, although populations increased ($P < 0.01$) in the Atlantic Flyway and the United States (figure 1).

Swans are fully protected in the Maryland part of the Bay and they have adapted to feeding in agricultural fields. Field feeding by swans especially on winter cover crops was commonly observed in the early 1970s (Munro 1980) at a time when declines

Table 1. Range and average populations of waterfowl in Chesapeake Bay from 1950 to 1994 as determined from aerial surveys.

Species	Area	High Count	(Year)	Low Count	(Year)	45 Year Average
Tundra Swan	Chesapeake Bay	75854	(1955)	18399	(1958)	35803
	Atlantic Flyway	109788	(1992)	27717	(1959)	65 821
	United States	176560	(1981)	48362	(1950)	116993
Canada Goose	Chesapeake Bay	701470	(1981)	87100	(1951)	390075
	Atlantic Flyway	955039	(1981)	272183	(1951)	625825
	United States	4116761	(1991)	755937	(1950)	2002532
Snow Goose	Chesapeake Bay	126000	(1985)	17	(1955)	35378
	Atlantic Flyway	276003	(1992)	29500	(1970)	91484
Total Dabblers	Chesapeake Bay	611515	(1955)	83300	(1971)	178156
	Atlantic Flyway	1873563	(1955)	510800	(1985)	876994
	United States	20341712	(1976)	1050770	(1985)	14583430
Black Duck	Chesapeake Bay	282029	(1955)	33046	(1992)	78748
	Atlantic Flyway	582453	(1955)	194100	(1987)	299256
	United States	761018	(1955)	268200	(1987)	429791
Mallard	Chesapeake Bay	182245	(1956)	14668	(1950)	55144
	Atlantic Flyway	375515	(1956)	85166	(1950)	208072
	United States	10630870	(1958)	3238051	(1993)	6556706
Wigeon	Chesapeake Bay	144350	(1955)	900	(1984)	24624
	Atlantic Flyway	230715	(1955)	29300	(1985)	90478
	United States	2063537	(1965)	561413	(1993)	1245778
Pintail	Chesapeake Bay	78211	(1956)	400	(1970)	14169
	Atlantic Flyway	456316	(1956)	34000	(1985)	138255
	United States	6165538	(1977)	1313134	(1993)	3624543
Total Divers	Chesapeake Bay	1002150	(1954)	100500	(1986)	254238
	Atlantic Flyway	2410253	(1953)	671600	(1988)	1124184
	United States	4799241	(1952)	1899345	(1983)	2910633
Bufflehead	Chesapeake Bay	36808	(1991)	2500	(1959)	16154
	Atlantic Flyway	73847	(1991)	20305	(1952)	45591
	United States	149594	(1993)	32810	(1957)	86693
Canvasback	Chesapeake Bay	399320	(1954)	34300	(1986)	94650
	Atlantic Flyway	451041	(1954)	79847	(1991)	148327
	United States	607294	(1954)	179089	(1972)	298200

Table 1. (Continued)

Species	Area	High Count	(Year)	Low Count	(Year)	45 Year Average
Goldeneye	Chesapeake Bay	40636	(1956)	2311	(1989)	16263
	Atlantic Flyway	96401	(1951)	17588	(1993)	56171
	United States	190729	(1956)	100670	(1991)	149052
Redhead	Chesapeake Bay	118800	(1956)	282	(1992)	28042
	Atlantic Flyway	296600	(1966)	30200	(1987)	121345
	United States	1283133	(1956)	238800	(1985)	546966
Ruddy Duck	Chesapeake Bay	124740	(1953)	4703	(1976)	31384
	Atlantic Flyway	163520	(1955)	25518	(1990)	65208
	United States	345109	(1984)	109823	(1992)	194882
Scaup	Chesapeake Bay	403658	(1954)	5500	(1959)	64789
	Atlantic Flyway	1309944	(1953)	258400	(1988)	591652
	United States	2806899	(1963)	679700	(1983)	1403668
Sea Ducks	Chesapeake Bay	140000	(1971)	2940	(1968)	20349
	Atlantic Flyway	225200	(1967)	25443	(1990)	181473
	United States	415416	(1967)	76100	(1968)	167482
Old squaw	Chesapeake Bay	21900	(1972)	0	(50/59)	5308
	Atlantic Flyway	26100	(1983)	1181	(1950)	11157
	United States	31624	(1961)	1181	(1950)	13750
Scoter	Chesapeake Bay	130900	(1971)	1551	(1981)	15560
	Atlantic Flyway	207900	(1967)	14059	(1990)	70316
	United States	395114	(1967)	68599	(1983)	153732

Table 2. Rate of population change (birds per year) for waterfowl between 1950 to 1994 in three areas.

Species	Chesapeake Bay	Atlantic Flyway	United States
Tundra swan	-153	+1315**	+2223**
Canada goose	+7942**	+11255**	+66740**
Snow goose	+1924**	+3796**	
Total Dabbling Ducks	-5527**	-18924**	-104028
Black duck	-2614**	-5807**	-9097**
Mallard	+5	-1225	-73129**
Wigeon	-1898**	-2873**	-13010**
Pintail	-966**	-6485**	-36620**

Table 2. (Continued)

Species	Chesapeake Bay	Atlantic Flyway	United States
Total Diving Ducks	-8578**	-20777**	-31152**
Bufflehead	+360**	+820**	+1944**
Canvasback	-3435**	-3113**	-2615**
Goldeneye	-507**	-1330**	-541
Redhead	-2056**	-1545**	-10897**
Ruddy duck	-1308**	-1433**	-1558**
Scaup	-1585	-11912**	-20113**
Total Sea Ducks	-257	-206	+1058
Oldsquaw	+29	+258**	+209
Scoter	-301	-464	+849

** Significant ($P < 0.01$).

in submerged aquatic vegetation were also being detected. Field feeding by swans also occurs in North Carolina, where fields are larger and there is less disturbance by humans. Swan populations in North Carolina have remained high in spite of limited swan hunting since 1989.

Mute swan populations have been closely monitored because of their propensity to cause problems with humans. These problems have been well documented in other areas, especially New England (Willey and Halla 1972). Populations of mute swans also have been blamed for decreasing submerged aquatic vegetation in the Bay, because of their presence in the Bay during the spring growing season. Mute swans are known to eat large amounts of submerged aquatic vegetation and to cause turbid water, but are relatively unimportant in regard to submerged aquatic vegetation when compared to other Bay-wide factors that are impacting the vegetation. Mute swans, however, have caused the decline nesting populations of black skimmers (*Rhynchops nigra*) and least terns (*Sterna albifrons*) on beach areas.

Geese

Canada geese (*Branta canadensis*) that winter in Chesapeake Bay breed in subarctic areas north of Hudsons Bay, especially the Ungava Peninsula. The trend in Canada goose populations has been upward during the 45-year period (table 2) and showed phenomenal increases in the 1970s to early 1980s (figure 2). These increases were most likely the result of the geese adapting to field feeding, which was commonly observed beginning in the late 1950s. Field feeding occurred mostly on harvested corn fields, but also was observed in fields of winter cover crops. The high caloric value of corn enabled geese to meet their energy needs quickly, resulting in less time spent searching for food when they would be vulnerable to hunting.

In the mid-1980s, Canada goose populations in Chesapeake Bay and the Atlantic Flyway began a decline, which was attributed to poor reproduction and higher rates of harvest, especially on the Eastern Shore of Maryland. Greatly reduced bag limits and reductions in the season length were implemented in an attempt to reverse the downward trend and appear to have been successful (Hindman et al. 1994). Canada goose populations in the United States did not decline in the 1980s (figure 2).

A nonmigratory (resident) population of Canada geese increased during the 1960-1970s and was considered a problem in the 1980s when special surveys were conducted to determine the size and distribution of this population. A disturbing trend noted from these surveys was that an increasing number of resident geese were on the Western Shore in areas protected from hunting. These populations were causing problems, especially on golf courses and at airports. A special September hunting season on resident Canada geese was initiated in 1993 to slow the growth of this population.

Although a small flock of lesser snow geese (*Chen caerulescens caerulescens*) (including the blue phase) had been associated with Blackwater National Wildlife Refuge for many years, it was not until the late 1970s and early 1980s that the greater snow goose (*Chen c. atlantica*) was seen in large numbers in the Bay area. This normally coastal salt marsh feeding species adapted to field feeding and occurred in large numbers on many farms near the Bay on the Eastern Shore. Populations of greater snow geese have increased ($p < 0.01$) in the Chesapeake Bay and the Atlantic Flyway during the last 45 years (figure 3). Greater snow goose populations occur only in the East, and therefore, the Atlantic Flyway population essentially is the U.S. population. Their numbers should not be confused with the huge population numbers of the lesser snow geese in the Mississippi and Central Flyways.

Dabbling Ducks

Dabbling duck populations showed a decline in the Chesapeake Bay and Atlantic Flyway, but were stable in the United States (table 2, figure 4). Historically, the American black duck (*Anas rubripes*) was the most commonly recorded dabbling duck in Chesapeake Bay, with populations as high as 280,000 in the early 1950s. Many of these birds were observed on the Eastern Shore, especially the Chester River area, where they were feeding on the lush beds of submerged aquatic vegetation (Stewart 1962). Aerial survey data, however, reflect a substantial decline ($p < 0.01$) in this species in the Bay, Atlantic Flyway, and the United States (figure 5), most likely related to the loss and degradation of habitat (Perry 1987).

Several authors, however, have reported that black duck declines are related to hunting (Martinson et al. 1968, Geis et al. 1971, Blandin 1982). Recent reviews of black duck hunting mortality (Conroy and Krementz 1990, Nichols 1991) question the role of hunting and suggest new ways to evaluate hunting as a mortality factor. The black duck is a species that is receiving special attention in the North American Waterfowl Management Plan (Canadian Wildlife Service and U.S. Fish and Wildlife Service 1986).

Mallard populations (*Anas platyrhynchos*) have shown no population changes in the Bay and the Atlantic Flyway during the 45-year period, which differs from the long-term declines noted in the U.S. population (figure 6). Increases since 1985 in mallard numbers have been recorded in the Bay and Flyway. These increases probably are related to the large number of captive-reared mallards that have been released during the last 20 years by the State of Maryland ($n = 226,700$) and private citizens ($n = 1,150,150$). Recent studies (Smith, D.B., unpub. report), however, indicate survival past the hunting season was low for mallards released by the state. Although some data indicate that releases of mallards result in a small percentage of total ducks harvested (Hindman et al. 1992), others feel that released mallards are a major percentage of harvested waterfowl (L. Johnson, pers. comm.).

Other dabbling duck species that historically have been recorded in large numbers in Chesapeake Bay include the American wigeon (*Anas americana*) and the northern pintail (*Anas acuta*). Both of these species have shown dramatic declines ($p < 0.01$) in the Chesapeake Bay, Atlantic Flyway, and the United States (figures 7 and 8). These species were once closely associated with the abundant beds of submerged aquatic vegetation, especially in the upper part of the Bay (Stewart 1962). The continental nature of the declines in the numbers of these species indicate the impacts are much broader and probably are related to increased hunting and loss of breeding and wintering habitat.

Diving Ducks

As a group, diving ducks in Chesapeake Bay declined ($p < 0.01$) (table 2) during the last 45 years (figure 9). Four of the six diving duck species declined ($p < 0.01$) in numbers (table 2). Bufflehead numbers (figure 10) increased ($p < 0.01$) in the Bay. These trends were similar to population trends of these species in the Atlantic Flyway and the United States.

In a previous review of diving duck populations in Chesapeake Bay, Perry et al. (1981) suggested that canvasback (*Aythya valisineria*) and redhead (*Aythya americana*) populations in Chesapeake Bay did not show increases during the 1970s that were shown in the U.S. populations for these species. They attributed this to a more rapid degradation of the Bay wintering habitat. The present analysis does not seem to fully support this notion. Canvasback populations have been stable in recent years (figure 11) and seem to have adapted to an invertebrate diet mainly of Baltic

Figure 1. Population trends of tundra swans in the United States, Atlantic Flyway, and Chesapeake Bay in 5-year averages, 1950-1994.

Figure 2. Population trends of Canada geese in the United States, Atlantic Flyway, and Chesapeake Bay in 5-year averages, 1950-1994.

Figure 3. Population trends of snow geese in the United States, Atlantic Flyway, and Chesapeake Bay in 5-year averages, 1950-1994.

Figure 4. Population trends of dabbling ducks in the United States, Atlantic Flyway, and Chesapeake Bay in 5-year averages, 1950-1994.

Figure 5. Population trends of black ducks in the United States, Atlantic Flyway, and Chesapeake Bay in 5-year averages, 1950-1994.

Figure 6. Population trends of mallards in the United States, Atlantic Flyway, and Chesapeake Bay in 5-year averages, 1950-1994.

Figure 7. Population trends of wigeon in the United States, Atlantic Flyway, and Chesapeake Bay in 5-year averages, 1950-1994.

Figure 8. Population trends of pintail in the United States, Atlantic Flyway, and Chesapeake Bay in 5-year averages, 1950-1994.

Figure 9. Population trends of diving ducks in the United States, Atlantic Flyway, and Chesapeake Bay in 5-year averages, 1950-1994.

Figure 10. Population trends of buffleheads in the United States, Atlantic Flyway, and Chesapeake Bay in 5-year averages, 1950-1994.

Figure 11. Population trends of canvasbacks in the United States, Atlantic Flyway, and Chesapeake Bay in 5-year averages, 1950-1994.

Figure 12. Population trends of redheads in the United States, Atlantic Flyway, and Chesapeake Bay in 5-year averages, 1950-1994.

Figure 13. Population trends of scaup in the United States, Atlantic Flyway, and Chesapeake Bay in 5-year averages, 1950-1994.

Figure 14. Population trends of goldeneyes in the United States, Atlantic Flyway, and Chesapeake Bay in 5-year averages, 1950-1994.

clams (*Macoma balthica*) (Perry and Uhler 1988). Recent Bay-wide surveys done by Jorde and Haramis (unpub report, Migr. Birds Res., Patuxent Environmental Science Center) show high numbers of this clam in the Bay. Redhead populations, however, still are being affected by the loss of submerged aquatic vegetation and reached an all-time low count of 282 ducks for the Bay in 1992 (45-year average = 28,042). United States and Atlantic Flyway populations also show downward trends (figure 12).

Historically, scaup have been the second most numerous duck in the Bay after canvasbacks. Scaup represent two species in the Bay, but hunter surveys (unpub. data, FWS) indicate lesser scaup (*Aythya affinis*) are more abundant than greater scaup (*Aythya marila*). Harvest surveys during 1961-90 indicated lesser scaup comprised 65% and greater scaup 35% of the total scaup kill. Scaup populations have been stable in the Bay, although declines have been more significant ($p < 0.01$) in the Atlantic Flyway and United States.

Populations of goldeneye and ruddy ducks (*Oxyura jamaicensis*) declined (table 2) in the Chesapeake Bay, the Atlantic Flyway, and the United States between 1950 and 1994. Goldeneye declines (figure 13) during the 45-year period are surprising when compared to bufflehead trends. These species nest and winter in similar habitats and have similar feeding habits. Ruddy ducks have shown increases (figure 14) in recent years, which may reflect their ability to winter in areas near cities where they find adequate benthic animal food and are safe from hunting.

Sea Ducks

Populations of oldsquaw (*Clangula hyemalis*) did not significantly change in Chesapeake Bay and the United States during the 45 year period 1950-94. Oldsquaw populations did increase ($p < 0.01$) in the Atlantic Flyway during this period. Populations of scoters and sea ducks (oldsquaw and scoters combined) did not show significant changes between 1950 and 1994 in the Chesapeake Bay, the Atlantic Flyway, or the United States. Both oldsquaw and scoter populations were fairly low in all three areas when surveys began in 1950. Populations of both oldsquaw and scoter increased during the early 1970s and then decreased in the 1980-1990s.

Because of the dispersed distribution of sea ducks in Chesapeake Bay, traditionally they have been difficult to survey (Eggeman and Johnson 1989). Surveys have been made using aerial transects in the 1960s (V. D. Stotts, pers. comm.), and in the 1990s they have been surveyed by D. Forsell (USFWS, poster paper Ches. Bay Res. Conf. 1994). These surveys and others from the ground are encouraged so that more can be learned about the status of sea ducks. It also is important to obtain population data by individual species, because studies by Stott and Olsen (1972) showed differential vulnerability to hunting among species.

Declines in scoter and oldsquaw in Chesapeake Bay in the 1980-1990s should be of concern to waterfowl managers as they may reflect increased harvest. There appears to be increased guide service for sea duck hunting in the Bay in recent years, which might be related to decreased guiding for Canada goose hunting following restrictive regulations on geese. Sea duck populations also have been more affected by disease than other Bay species, with three major outbreaks (1970, 1978, 1994) of avian cholera in Chesapeake Bay.

Waterfowl In the Future

When considering the future of waterfowl in Chesapeake Bay, it is important that managers, hunters, and all Bay enthusiasts be conscious of the tremendous abundance of waterfowl that once existed here. It also is important to recognize the role of hunting in regard to these populations and consider the economic and recreational benefits that have been realized over the centuries. Current and future numbers should be judged on what potential populations can be sustained in the Bay, without causing negative effects. It is possible that the Bay could winter many more ducks than at present, and if Bay grasses returned to the upper Bay, the populations could be even higher.

It is also possible, however, that present management techniques are inadequate to obtain goals for healthy sustaining populations established by the Chesapeake Executive Council (1990). In addition, there also is concern that these goals have not been set high enough. There are large areas of the Bay that seem to have adequate habitat but few ducks are using them. Hochbaum (1955) warned that broken traditions caused by overhunting can account for the disappearance of ducks from suitable habitat. Recent surveys in the northern duck breeding areas indicate that water conditions have improved, but many of the wetlands do not have breeding pairs of ducks. We cannot expect high future duck populations if available habitat for breeding is not being used.

Although many of the trends of numbers of waterfowl have been continental in nature, there is reason to believe that local regulations can affect local populations. Mallard population stability in the Bay, at a time when the United States mallard population is declining, indicates the effect of locally released ducks. Although many managers scoff at the artificiality of such techniques, there have been many hours of recreation obtained and probably less pressure on species of lower numbers. Managers and researchers should evaluate mallard shooting in regulated shooting areas (private and public) as a management technique, and try to lessen the amount of hunting on public areas of the Bay. Such an evaluation would lend itself to an adaptive management strategy for harvesting waterfowl (Johnson et al. 1993). Regulations governing the release and harvest of captive-reared mallards on shooting preserves are presently being reviewed by the FWS (*Federal Register*, Vol. 58, No. 103).

Chesapeake Bay desperately needs more sanctuary areas where waterfowl are not harassed by hunters or other Bay users and where they can rest and feed. Without these areas, waterfowl will be forced to expend energy on moving owing to the extensive human uses of the Bay, and their condition for winter survival and breeding will be degraded. Open water sanctuaries may be especially beneficial to diving ducks (Haramis 1991) and geese. Freshwater impoundments near the Bay in upland sites also should be constructed in greater numbers, which will be especially beneficial to dabbling ducks as feeding and resting areas.

The Chesapeake Bay Waterfowl Policy and Management Plan (Chesapeake Executive Council 1990) mentions several of the above suggestions, but little progress has been made in implementation because of lack of funding.

Although actions (e.g., sanctuaries and impoundments) directed toward waterfowl in Chesapeake Bay may not affect continental populations, it might make major differences in the Bay and serve as an example to the rest of the country. It is important that if changes are to be expected, managers must initiate these changes, as it is obvious that past actions have not obtained the desired results. The role of private hunting areas, which now are providing thousands of acres of land for wildlife and probably reducing the pressure on public areas of the Bay, should be considered seriously by wildlife managers. Waterfowl population goals should be set high, and then managers should be prepared to make the difficult decisions to reach these goals. With proper management, trends of waterfowl could be upward for all species in the future.

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