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ICPRB Report 93-2

Zooplankton Indicators Project Team

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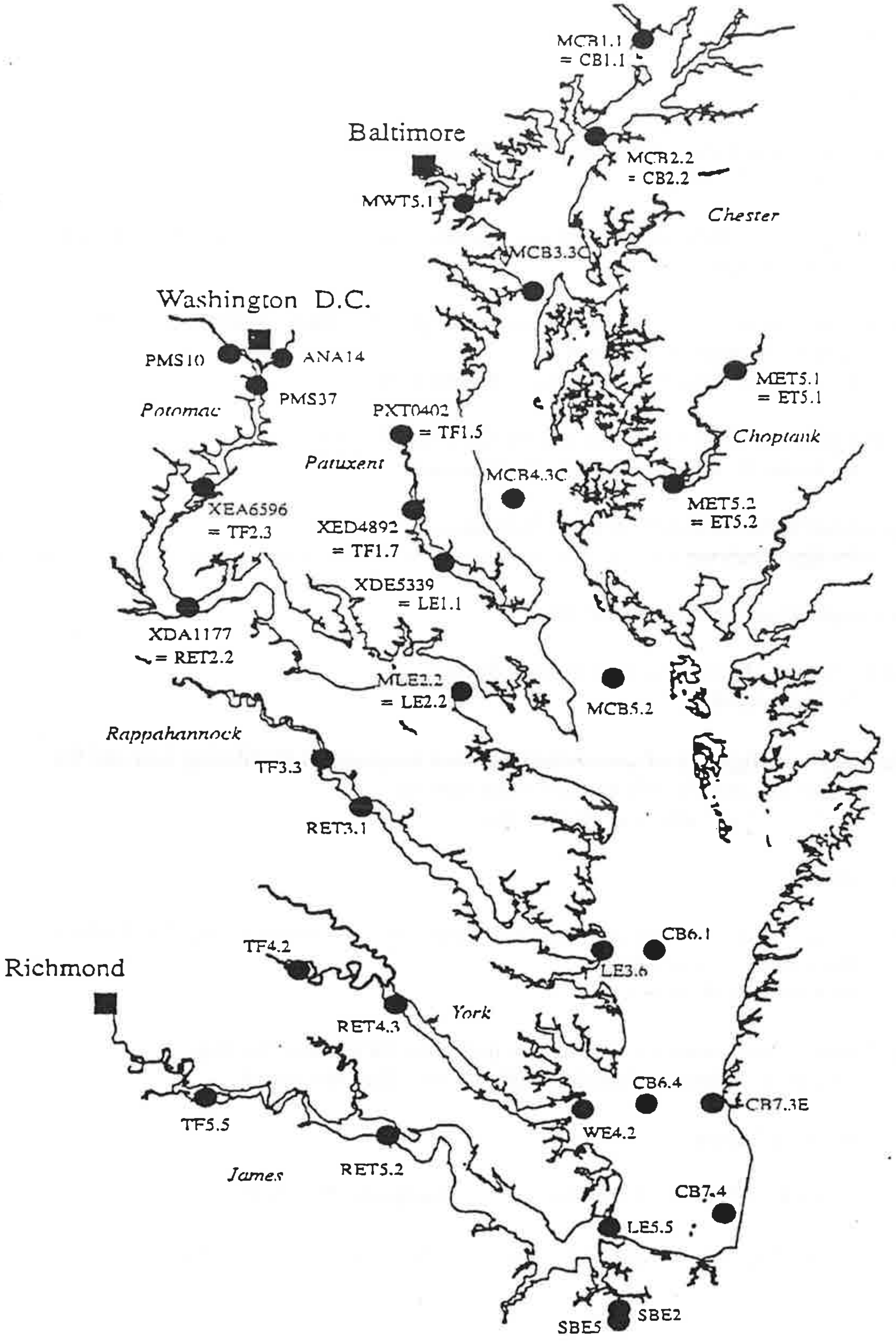
CHESAPEAKE BAY
ZOOPLANKTON MONITORING STATIONS



Findings and Recommendations

of the project through June 1993

The primary **goal** of this project is to produce a set of zooplankton-based environmental indicators applicable to the Chesapeake Bay. A second goal is to begin to promote the use of zooplankton data as environmental indicators in assessments of estuarine ecosystem health in the rivers and mainstem of Chesapeake Bay. The immediate objective of the project this year was to further develop and test zooplankton metrics for use as indicators bay-wide.

Zooplankton are a critical link between primary producers (phytoplankton) and higher consumers, especially forage fish and larval stages of predatory fish. Rigorous tests were begun on a selected group of "candidate" zooplankton indicators to confirm their scientific validity as indicators of environmental "health". Some of the candidates are proving to be genuine, strong indicators of nutrient and toxic impacts on zooplankton, of phytoplankton - zooplankton linkages, and of zooplankton - fish linkages.

Individual Principal Investigators (PI) working to interpret the analyses for groups of related indicators have not yet had a chance to integrate their results because of the time limits on the project. Each of the chapters in the body of this report would benefit enormously upon integration. This report should be viewed as a DRAFT. Project results will be integrated and a final list of indicators published in the next year of this project.

Highlights for management

- Microzooplankton, small-bodied zooplankton which are an integral part of the "microbial loop", are excellent indicators of eutrophication in Chesapeake Bay waters. One species of the microzooplankton, a hypotrich ciliate, seems to be a good indicator of recent low DO events.

- Mesozooplankton, larger-bodied zooplankton which are better food items for most larval fish, are at present *weakly* linked to the phytoplankton. Reductions in the existing nutrient concentrations in the bay generally paralleled an increase in mesozooplankton.

- Of the eight Chesapeake Bay striped bass nursery areas presently being monitored for mesozooplankton, none had "optimal" food levels required for normal striped bass larval growth during the spring (April - June). Only one (Choptank) has consistently met "minimum" food level requirements of 15-25 mesozooplankton liter^{-1}.

- Mesozooplankton and finfish planktivores are strongly linked in many tidal fresh and oligohaline reaches of the bay and its tributaries.

- Food web management strategies already developed for freshwater lakes can probably be directly applied to tidal fresh reaches in the bay in order to bolster mesozooplankton populations and reduce summer algal blooms. Trophic linkages in brackish waters of the bay indicate food web management strategies for freshwater lakes are not *directly* applicable to oligohaline and mesohaline waters. Further analyses of fish-zooplankton linkages will suggest food web management strategies for these waters.

- Toxic pollutants are the confirmed cause of low zooplankton abundance in the Elizabeth River, and are probably responsible for depressed abundances in the oligohaline portion of the Potomac

River. Zooplankton monitoring stations with depressed abundances for which no other alternative causes are obvious should be investigated for toxic pollutants.

- Preliminary efforts to develop an "index of biological integrity" for estuarine zooplankton have yielded promising results.
- Zooplankton indicators will be integrated into the CBP's existing continuum of environmental indicators, and will be used in the CBP time variable water quality model, in bay ecosystem processes models, and in assessments of ecological risk from toxic substances.

Spatial and temporal limits to the use of the indicators

The dynamic and heterogenous nature of temperate zone estuaries forces certain spatial and temporal constraints on the use of zooplankton metrics as environmental indicators. In Chesapeake Bay, salinity, temperature and flow are the primary *natural* forces governing the spatio-temporal distributions of zooplankton. These environmental controllers are essentially unaffected by man. If not accounted for, they can confound our understanding of how nutrient reductions and fish restorations change zooplankton populations.

Plankton (algae, zooplankton, fish larvae) abundances at riverine stations that experience frequent periods of high flow may be severely depressed during these events. The flow rate at which zooplankton populations are impacted appears to be roughly twice the median flow rate, but is probably modified somewhat by the cross-sectional area of the river. High flow events almost always occurred in the spring during the study period.

Salinity very dramatically affects species composition, species dominance, and trophic relationships at the tidal fresh/oligohaline interface. Salinity gradients may influence species composition in polyhaline waters near the mouth of the bay, but an influx of marine species is a more likely explanation for the increased diversity there.

To date, no detailed analyses of temperature effects on the zooplankton indicators have been done but analyses done for this project have focused on indicator values averaged for "spring" and "summer" periods which correspond very well to the April - June and July - September periods of the annual temperature cycle, respectively.

Zooplankton Linkages to Water Quality, Toxics and Phytoplankton

Nutrients indirectly affect zooplankton through primary producers (phytoplankton). The strong relationships commonly found between zooplankton (micro- and meso-) and nutrients or nutrient loads in freshwater environments are presently not strong in Chesapeake Bay and its tributaries. TP and mesozooplankton were coincident, and TP was not controlling, for all saline areas. Nutrient loads, the product of nutrient concentration and river flow, affected zooplankton through their flow component rather than concentration component.

Microzooplankton abundances and biomass were directly related ($r^2 > 0.60$) to phytoplankton biomass (expressed as ambient chlorophyll biomass) with strongest relationships evident in the tidal fresh regions. Phytoplankton biomass exerted bottom-up control on the microzooplankton, and appears to be a good predictor of microzooplankton biomass.

Several hypotrich ciliates (microzooplankton) were strongly related to hypoxic-anoxic conditions at stations experiencing low dissolved oxygen at least a few times each year. The presence/absence of the hypotrich ciliate in the water column looks promising as an effective indicator of recent low dissolved oxygen (DO) events.

Variations in mesozooplankton abundances and biomass were somewhat explained by variations in phytoplankton biomass in the tidal fresh. In more saline regions of the system, mesozooplankton were inversely related to secchi depth, primarily a function of suspended phytoplankton particles. Water quality factors that regulate abundances of copepods (e.g. salinity) and cladocera, primarily *Bosmina* (e.g. salinity, cyanobacteria biomass), also controlled the ratio of copepods to cyclopoids and cladocera.

Metal pollutants chronically suppress zooplankton populations in the Elizabeth River and may be responsible for low zooplankton abundances in the oligohaline reach of the Potomac River. Other pollutants are known from the literature to affect estuarine zooplankton.

Zooplankton - Fish Linkages

Of the eight Chesapeake Bay striped bass nursery areas presently being monitored for zooplankton, none had "optimal" food levels required for normal striped bass larval growth during the spring (April - June). Only one (Choptank) has consistently met "minimum" food level requirements (i.e. median mesozooplankton abundance of 15-25 liter^{-1} during April - June). During the monitoring period, the Potomac declined to "below minimum" food levels and the James declined to "poor" food levels; the Patuxent and Rappahannock were consistently "below minimum" and the York and upper mainstem, "poor".

A significant positive relationship ($P < 0.05$) between mesozooplankton density in the spring and striped bass juvenile index the following summer occurred in the Choptank and Potomac rivers. The relationship for the Potomac is further supported by examination of historical zooplankton data sets. Other striped bass nursery areas did not show a similar relationship because zooplankton densities were all relatively low and precluded rigorous regression analyses over a wide range of zooplankton densities. While clearly recognizing that environmental factors such as temperature, flow, turbidity, and light penetration greatly affect striped bass larval survival, we conclude that food densities available to larvae is presently one of the factors regulating striped bass larval success in most bay nursery areas.

Zooplankton are the obligate prey of most larval fish and principal prey to the juveniles and adults of species presently dominating finfish communities in Chesapeake Bay. Summer abundances of mesozooplankton and finfish planktivores (obtained from the Maryland and Virginia juvenile seine surveys) showed significant but strikingly different relationships to each other in the tidal fresh and oligohaline reaches *not* impacted by toxics or rapidly increasing nutrient loadings. Planktivorous fish in tidal fresh waters exert strong top-down controls on their prey, as evidenced by *inverse* regression slopes, and are one of the factors governing mesozooplankton abundance and community structure in these areas. This echoes a pattern found repeatedly in freshwater lakes. *Positive* regression slopes in the oligohaline manifest strong bottom-up controls by prey on their predators and indicate mesozooplankton are one factor regulating summer fish populations there. The fundamental change in zooplankton - fish linkage at the fresh/oligohaline interface appears to be the result of major shifts in prey vulnerability, predator-prey overlap, and predator efficiency brought about by environmental changes at the tidal fresh/oligohaline interface. A straightforward relationship between finfish planktivore and mesozooplankton abundances was not found in the mesohaline reaches, apparently because invertebrate planktivores become increasingly important in this salinity regime.

Preliminary Zooplankton "Index of Biological Integrity"

A preliminary assessment was conducted to determine the feasibility of developing an "index of biological integrity" (IBI) for zooplankton. Selected stations in each salinity regime were subjectively classified as environmentally "stressed" or "nonstressed" because analyses of the indicators were not far enough along to objectively classify stations. The IBI scores were calculated for four salinity regimes and two seasons (spring and summer) using nine of the candidate zooplankton indicators. The results are promising, i.e. the stations with the most clearly documented "stressed" conditions had the most dramatic separation from other stations on the index scale. The findings did indicate the need for future refinement of the zooplankton IBI approach for the Chesapeake Bay. More objective methods for identifying "stressed" and "non-stressed" are available now and should clearly be employed. The effects of salinity and seasonality on the IBI scores need to be explored further to determine whether there are alternative means of "correcting for" these non-anthropogenic factors statistically prior to IBI calculations. Current and future zooplankton indicators should be thoroughly assessed for ecological meaning and sensitivity to environmental stress, as well as for power and robustness in the measurement of changes and long-term trends in water quality conditions and fish restorations.

Uses of the Indicators

The zooplankton environmental indicators will be integrated into the Chesapeake Bay Program's existing continuum of environmental indicators. The zooplankton component will forge a "trophic interface" between indicators of habitat and water quality and the living resources based environmental indicators.

The project results will also be used in the near future in refining the Chesapeake Bay Program time variable water quality model and in bay ecosystem processes models. These models are being developed as management tools for Chesapeake Bay and they rely on zooplankton monitoring data for input and calibration. Currently, some of these models use simplified zooplankton functions and are run without fully implementing the zooplankton variables. Expanded understanding of zooplankton as environmental indicators will allow for better implementation of zooplankton variables.

The Chesapeake Bay Program basin-wide Toxics Reduction Strategy includes funding for a joint EPA/NOAA toxics research program focused on ecological risk assessments for Chesapeake Bay toxic pollutant problems. This work has included studies of food chain pathways (including zooplankton) and the movement of toxic substances in the environment. Applying these research findings to the management of toxic substance load reductions will require quantitative information on the responses of key impacted communities. Zooplankton environmental indicators will provide one set of these food-chain based response measures.

Other potential uses for zooplankton indicators are the ongoing efforts to develop biocriteria for state waters and to establish Chesapeake Bay restoration goals could potentially use zooplankton environmental indicators. The indicators will also be useful in focusing and delineating restoration strategies for living resources in the Bay area.

General Recommendations

- Project PI should thoroughly integrate the results of this year's efforts to further develop zooplankton environmental indicators before recommending a final suite of zooplankton environmental indicators.

- Efforts to develop a zooplankton "index of biological integrity" (IBI) based on an array of zooplankton indicators should be continued. The resulting index should be included in the Chesapeake Bay Program's existing continuum of environmental indicators.

- The team should endeavor in the next year of the project to have the zooplankton indicators and IBI incorporated into ongoing modeling and assessment efforts.

- The feasibility of applying food web management strategies already developed for freshwater lakes to tidal fresh reaches in the bay should be investigated. The possibility that food web management strategies can be developed for brackish waters seems good and should be explored.

- Laboratory experiments on hypotrich oxygen sensitivity, viability and community integrity are necessary before the usefulness of hypotrichs as indicator of recent low DO events can be established.

- Stations with depressed zooplankton populations for which no other alternative causes seem to be present should be monitored for pollutants. Tentatively, these stations include the oligohaline and mesohaline stations in the Potomac (RET2.2, LE2.2) and the tidal fresh York (TF4.2).

- Reanalysis of historical monitoring data would contribute a long-term perspective of how phytoplankton-zooplankton and zooplankton-fish linkages have changed during the last two decades as man's impact on the Bay has increased. These analyses could easily be undertaken for the Calvert Cliffs area and the Potomac River because these data are readily available. Zooplankton indicator values should be calculated for historical monitoring data from Calvert Cliffs and the Potomac River, and analyzed for linkages with other trophic levels and for the effects of temperature and salinity which can confound assessments of estuarine "health".

Data Analysis Recommendations

- Zooplankton monitoring data should be carefully screened to identify data affected by salinity, temperature and flow before zooplankton indicators are calculated and used to assess Chesapeake Bay "health". The data should also be blocked according to salinity, and possibly season, before zooplankton indicators are calculated.

- Critical flow thresholds, above which zooplankton abundance (and similarly fish larvae abundance) is depressed, should be determined for each tributary and the upper bay. Net current rate, or residence time, for the river segment at each station may also be useful in determining critical thresholds.

- Further work needs to be done to quantify the effect of salinity on all of the various zooplankton environmental indicators.

- Further work needs to be done to clarify the effect of temperature on the indicators, and confirm the validity of the "spring" and "summer" groupings of the data.

- The spring food availability index should be expanded to reflect microzooplankton densities. While perhaps not important for striped bass beyond first feeding, microzooplankton densities are likely to be important for developing food availability indices for larvae of other species such as white perch.

- It is also recommended that *summer* food availability indices be developed for forage species that feed almost entirely on zooplankton, such as bay anchovy and silversides.

- Efforts to examine associations between spring zooplankton and summer striped bass juvenile index should be broadened. Stepwise regressions that lag information on flow, rainfall, and nutrients should be explored.

- Trophic relationships in all of the salinity regimes need to be further explored with the array of zooplankton indicators that are at hand. Zooplankton indicators also should be compared with the various trawl surveys. These future investigations can confirm suspected zooplankton - fish linkages at some stations (e.g. the mesohaline stations) and identify environmental variables that are disrupting the linkages at other stations.

Project Overview

In Chesapeake Bay, as well as in other estuarine and near coastal systems, zooplankton are a critical link between primary producers (phytoplankton) and higher consumers, especially forage fish and larval stages of large predatory fish. This role in the bay's ecosystem is the strongest argument for developing zooplankton indicators for managers to use in interpreting the ecological functions and biological integrity of Chesapeake Bay.

Zooplankton have been monitored bay-wide at a network of stations in Chesapeake Bay since 1984. The development of biological indicators has been a goal of zooplankton monitoring programs in the Chesapeake Bay since their inception. With several years of data now collected, zooplankton indicators designed for use as management tools can now be developed ("Chesapeake Bay Zooplankton Monitoring: Report on a Workshop Held in Easton, Maryland, September 23-24, 1991"). Specific characteristics of the zooplankton community appear to reflect differences or changes over time in several environmental parameters in the Bay, such as nutrient concentrations or exposure to toxic substances. Also, densities of particular zooplankton groups are one of several factors controlling survival of larval fish. Thus, characteristics of the zooplankton community are true biological indicators of the Bay's ecological health - measures of the Bay's ability to function under present-day and future stresses and disturbances.

Goals and Objectives

The primary goal of this project is to produce a set of zooplankton-based environmental indicators applicable to Chesapeake Bay. A second goal is to begin to promote the use of zooplankton data and environmental indicators in assessments of estuarine ecosystem health in the Chesapeake Bay and its tributaries.

The immediate objective is to further develop metrics capable of being used as environmental indicators of Bay "health". We recognize there are inherent limitations in how the indicators should be used. The indicators are still not fully evaluated at the end of this short-term project.

Report Structure

The report opens with a discussion of the limitations of zooplankton environmental indicators. The section calls attention to the high degree of natural heterogeneity in temperate zone estuarine ecosystems and to the primary sources of natural variation in zooplankton communities - flow, temperature and salinity.

The report then presents the individual indicators in two sections: zooplankton linkages with water quality, toxics and phytoplankton, and zooplankton linkages with finfish. For the most part, these chapters use the "ecosystem regression model" approach (*Current Approaches for Modeling Estuarine Ecosystem Process*, STAC/Chesapeake Bay Program, 1993) in analyzing the zooplankton indicators and identifying linkages. This approach avoids the mechanistic details of ecosystem processes while identifying trends in, and strong relationships between, environmental forcing functions and selected ecosystem responses. The ERM approach is holistic, empirical, and well suited for providing management with the tools it needs to measure status and change in the system.

The next section of the report presents a first attempt at creating a zooplankton "index of biotic integrity" (IBI) for Chesapeake Bay estuarine waters. Biotic integrity has been defined as the ability of an ecosystem to support and maintain "a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats of the region" (Karr and Dudley, 1981). The approach has gained considerable acceptance among living resources managers and scientists for streams world-wide, but still needs to be adapted for other aquatic ecosystems. Work on application of the IBI approach to fish and benthic communities in tidal fresh and estuarine ecosystems in Chesapeake Bay is already underway.

The following section addresses utilization of the indicators by bay researchers and managers. Effective ways of putting the Chesapeake Bay zooplankton monitoring data to use immediately are discussed, and the next steps in developing new zooplankton indicators are debated.

The final section of the report, placed at the beginning in lieu of an executive summary, reviews the project results through June 1993. Because of the time constraints of this FY93 year project, the principal investigators (PI) did not have the opportunity to integrate their results. Each chapter would benefit enormously, and possibly change, with such integration. For this reason, each chapter in this report should be viewed as a DRAFT.

Chronology

The U.S. Environmental Protection Agency identified the project's PI in its original funding proposal. During the course of the project, individuals with relevant expertise in zooplankton monitoring and data management in the Chesapeake Bay were invited to participate. Oversight and guidance of the zooplankton indicator project was provided by an *ad hoc* management workgroup. This workgroup was comprised of representatives from each of the following agencies: Virginia Department of Environmental Quality (DEQ), Maryland Department of the Environment (MDE), Washington, D.C. Department of Consumer and Regulatory Affairs (DCRA), Maryland Department of Natural Resources (MD DNR), and the U.S. Environmental Protection Agency (EPA).

A comprehensive search of the literature for zooplankton indicators was conducted by the Benedict Estuarine Research Laboratory and a list of possible references was sent to each of the project PI in November, 1992 (Appendix A). The PI reviewed the literature at a team meeting in January, 1993, and selected candidate indicators to test. Subsequently, the PI drafted a data request for each candidate indicator, describing the underlying hypotheses, pertinent data sets, data configurations, calculations and computer output required to test the candidate indicators with Chesapeake Bay data. The Applied Marine Research Laboratory at Old Dominion University assembled the pertinent data sets and began calculating the candidate indicator values from zooplankton data collected for the Virginia and Maryland Chesapeake Bay Monitoring Programs and for an earlier Potomac study. The Interstate Commission on the Potomac River Basin assisted by constructing an equivalence table for the Maryland, Virginia and NODC[1] species codes (Appendix B). At a second team meeting in late March, Ray Alden and Mike Lane reviewed

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[1] The NODC, or National Oceanographic Data Code, is currently being created and assigned to individual species by the National Oceanographic and Atmospheric Administration (NOAA). The systematic structure of the code allows data for related species (e.g. copepods) to be quickly extracted from a database and manipulated. This facilitated the calculation of candidate indicator values in this project because many of the zooplankton indicators are for groups of related species rather than for single species.

AMRL progress-to-date and received input from the group about mid-course corrections and changes. The PI presented preliminary results and further defined how the candidate indicators would be analyzed. The team discussed the value of the Calvert Cliffs data and agreed future analyses of this data and the historical Potomac data would benefit the project. The team came to a consensus on the report outline and the various writing assignments, agreeing that publications in peer-reviewed journals would ultimately be the preferred form of this work. In late April, the PI received the final indicator calculations from AMRL and began to analyze the data and write their chapters. In late May, first drafts of each chapter were received by Claire Buchanan, the chair, and incorporated directly into this report without benefit of thorough reviews by the entire team.

Spatial and Temporal Boundaries of the Zooplankton Environmental Indicators

by Ray S. Birdsong

The purpose of this section is to call attention to the primary sources of natural variation in Chesapeake Bay as a caution toward the judicious application of biometrics in the assessment of estuarine ecosystem health. Implicit in the rationale for biological monitoring is the assumption that any perceived environmental degradation is a product of human activity and potentially remedial. Estuaries, more than most aquatic environments, display a large amount of spatial and temporal heterogeneity. A variety of physical and biological factors of non-anthropogenic origin impinge upon the estuarine ecosystem and can alter metrics that are used to gauge community health. The strongest of these factors affecting zooplankton indicators in the Chesapeake Bay are flow rate, salinity, and temperature. Flow rate impacts zooplankton abundance in tidal fresh reaches, especially during the spring. Salinity determines the spatial distributions of freshwater and estuarine zooplankton communities. Temperature determines the temporal, or seasonal, occurrence of the communities. Care must be taken to apply the zooplankton bioindicators so that these three factors do not confound assessments of estuarine health.

Hydraulic Effects

Most tidal freshwater stations in the Bay display evidence of periodic, usually seasonal, hydraulic impact as plankton populations are subjected to rapid downstream transport by high river flow. Some downstream stations in the transition zone and the lower estuary may also be occasionally impacted by high flow events but salinity changes associated with increased river flow obscure the assessment of the hydraulic contribution. Three tidal freshwater stations of greatly differing flow characteristics in the Patuxent, York and James River systems were selected to illustrate the hydraulic effect. Unfortunately, no measurements of current speed are available at the stations at the time of zooplankton collection, so between station comparisons are indirect and utilize the monthly average flow (ft^3/sec) measured at the fall line. Since the current speed at a given flow is dependent upon the cross sectional area of the river, each station will display a unique flow rate above which the plankton community is noticeably impacted. Table 1 compares these three stations in several pertinent statistics for the period of March 1986 through May 1991. From inspection of the plots of zooplankton abundance v. flow (Figs. 1-3), the flow at which the zooplankton populations are impacted appears to be approximately twice the median flow rate.

In the Patuxent River with a low flow rate and no pronounced spring freshet, there is a striking spring zooplankton bloom with concentrations in March-May averaging 232% of those in June-February. At TF1.5 flow exceeded twice the median rate only in three of the 63 months analyzed and in none of these did it appear to impact the zooplankton concentration (Figure 1).

In the James River with an average spring flow rate over 23 times that of the Patuxent (11,600 versus 500 ft^3 sec^{-1}), the spring zooplankton concentration appears heavily impacted, with the March-May average concentration only 50% of that of June-February. During months with flows of twice the median flow or higher (20% of the sampled months) zooplankton concentrations at TF5.5 were severely depressed and displayed values of less than 5% of the annual mean (Table 1).

Station TF4.2, located in the Pamunkey River tributary of the York River, is intermediate between the James and Patuxent Rivers in both flow characteristics and the apparent impact on zooplankton concentrations. Similar to the Patuxent River, the March-May average zooplankton concentration is nearly twice that of June-February; however, periods of flow that exceed twice the median are correlated with zooplankton concentrations that are only 10% of lower flow periods (Figure 2).

In summary, zooplankton populations at riverine stations that experience frequent periods of high flow may be severely depressed during these events. The relative health of various tributary stations based on metrics such as zooplankton abundance or biomass alone could produce highly distorted assessments if the unique flow characteristics of the tributaries are not factored into the evaluation. A prerequisite of zooplankton analyses in the future should be a careful evaluation of flow effect, especially in spring. Critical flow thresholds, above which zooplankton abundance is depressed, should be determined for each station in the tributaries and upper bay. Net current (net flow x cross-sectional area of river) or residence time at each station should also be determined as it may prove useful in identifying physical characteristics of the system that govern flow and, ultimately, zooplankton communities.

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ZOOPLANKTON AS STAND-INS FOR UNDERSTANDING THE EFFECT OF SPRING FLOW EVENTS ON THE ICHTHYOPLANKTON, OR FISH LARVAE, OF ANADROMOUS FISH. Discuss how the zooplankton can recover relatively quickly from spring flow events because of rapid population growth rates at that time, however cohorts of fish larvae essentially have one chance to survive spring because they are transient members of the plankton. Depression of zooplankton abundances during high flow events can tell us something about ichthyoplankton susceptibility to high flow.
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Salinity Effects

Most zooplankton community characteristics vary along the salinity gradient. Zooplankton abundance, biomass, diversity and species composition may all be influenced by the salinity regime at a station. Figure 4 shows the distribution of Chesapeake Bay stations in relation to abundance and salinity. Zooplankton abundance and biomass values typically decline with increasing salinity. This effect is most apparent in the mesohaline and polyhaline portions of the Bay but is confused by the apparent effects of other factors. The plot of stations in Figure 4 appears to be divided into an upper group (diamond symbols) with high abundance values that decline with increasing salinity. A lower group (square symbols) comprising the Elizabeth, James, York and Rappahannock River stations plus the two mainstem stations just below the Susquehanna mouth appear to display anomalous abundance values for their respective salinity regimes. Some of these stations are subject to periodic hydraulic impact and this likely accounts for a portion of the apparent anomaly; however, other stress-related factors not yet6 identified are also probably involved. High concentrations of heavy metals and other toxic substances are known to occur at three of these stations: Elizabeth River stations SBE2 and SBE5 and Potomac River station RET2.3.

Zooplankton diversity, at least in the upper mesohaline and polyhaline sections of the Bay, increases with increasing salinity (Fig. 5). It is not clear whether this diversity gradient is directly related to salinity or is simply a reflection of the relative proximity of the stations to the Bay mouth, the source of a variety coastal species that enter the lower Bay.

Species composition and species dominance are clearly affected by the salinity regime. Species composition of the zooplankton communities in lower salinity waters appear particularly sensitive to salinity shifts and often react rapidly and dramatically. An especially instructive example of the effect of salinity on species composition and dominance is the alternating dominance of the copepods, *Acartia tonsa* and *Eurytemora affinis*, at oligohaline stations. Station TF3.3 in the Rappahannock River, for example, ranges from freshwater to lower mesohaline (Fig. 6). *Eurytemora affinis*, the dominant copepod during periods of low or no salinity, is replaced by *Acartia tonsa* during higher salinity events. *Bosmina* (Cladocera), who frequently dominate at freshwater stations show a similar sensitivity to periodic salt water intrusions.

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Temperature Effects

To date, no detailed analyses have been done to correlate temperature, or rate of change in temperature, to zooplankton indicator values. Many of the indicator analyses (below) have focused on two periods called "spring" and "summer". These time periods correspond to two distinct portions of the annual temperature cycle. "Spring" comprised the months of April, May and June. This happens to be when average water temperatures are above 10°C, an apparently critical temperature for many zooplankton species, and climbing rapidly. "Summer" comprised the months of July, August and September. Average water temperatures during these months are high and relatively stable. Average temperatures fall rapidly during October, November and December, are low and stable during January and February, and begin to climb in March.

That temperature affects the production, standing crop and species composition of the zooplankton community is well established. What is not so obvious is how temperature, other water quality factors, abundance of phytoplankton and planktivores and the life history patterns of the constituent species interplay to produce the annual seasonal zooplankton pattern that is common to the mesohaline and polyhaline portions of the Bay. Figure 7 shows the typical seasonal abundance pattern at a Chesapeake Bay mesohaline station. The two periods of peak abundance occur in spring and summer. The spring bloom commences when the water temperature is at the annual minimum and the summer bloom when it nears the annual maximum value. While there are differences in the species composition of the two bloom periods, some dominant species such as *Acartia tonsa* frequently contribute heavily to both blooms. Annual abundance typically reaches its nadir in December when water temperatures have dropped sharply. A second period of low abundance consistently occurs in May-June when water temperatures display their most rapid rate of increase. During this period planktivore abundance also displays a marked increase [references needed here]. One additional but less severe drop in abundance occurs in late summer (usually August in the lower Bay) and is not associated with temperature change. Abundance subsequently rebounds somewhat in September and October. This pattern of a late summer dip followed by a rebound becomes less consistent in salinities below 15 ppt. The August decline occurs close to the time of the annual dissolved oxygen minimum and, at least in the lower Bay, appears more consistent and pronounced at stations where hypoxia is more severe.

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Table 1. Comparison of stations TF1.5, TF4.2 and TF5.5 in flow characteristics and several mesozooplankton community statistics for the period March 1986 through May 1991.

	TF1.5 (Patuxent R.)	TF4.2 (York R.)	TF5.5 (James R.)
Median flow ($ft^3\ sec^{-1}$)	370	1,200	5,700
Maximum flow	1,300	5,100	33,500
Avg. flow (March - May)	500	1,900	11,600
Avg. mesozooplankton concentration ($10^3\ m^{-3}$)	26.3[1]	7.6	9.7
Avg. mesozooplankton concentration ($10^3\ m^{-3}$) at flows <2X median	26.2[1]	9.9	12.9
Avg. mesozooplankton concentration ($10^3\ m^{-3}$) at flows >2X median	28.1	1.0	0.6
Avg. mesozooplankton concentration ($10^3\ m^{-3}$) (March - May)	44.0[1]	11.6	5.3
Avg. mesozooplankton concentration ($10^3\ m^{-3}$) (June - February)	18.9	6.2	10.6

[1] A concentration of 706,100 m^{-3} in May 1991 was excluded from the averages.

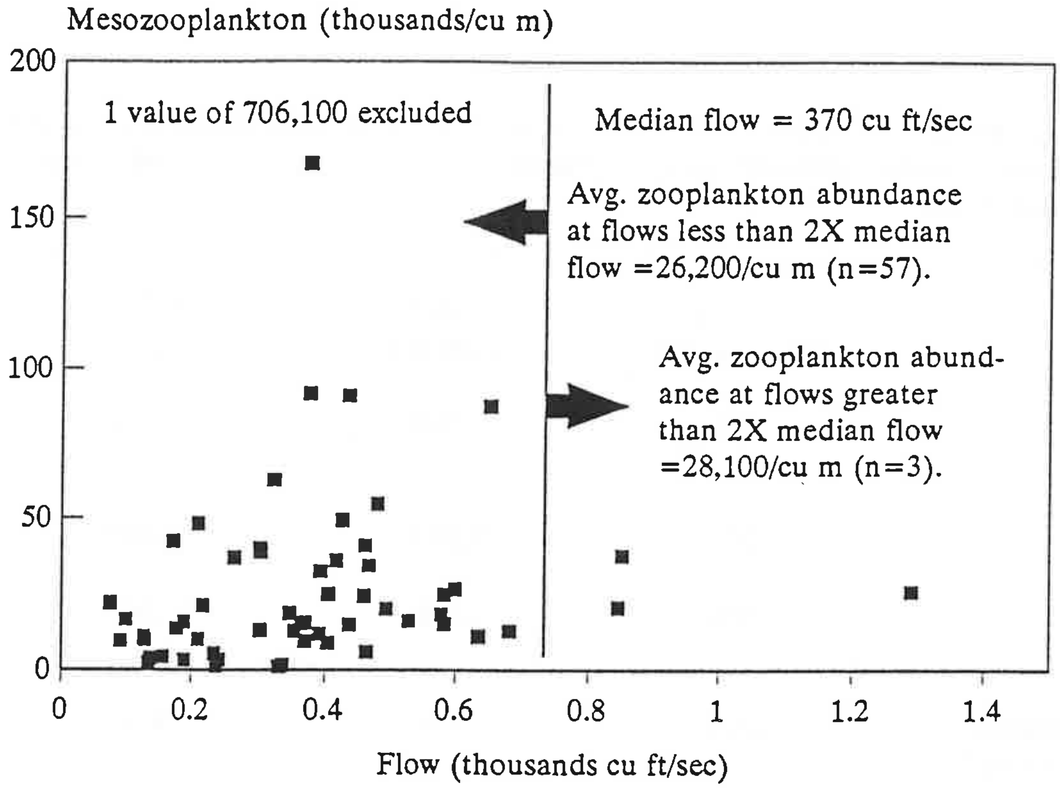

Figure 1. Mesozooplankton abundance versus flow at Patuxent River station TF1.5.

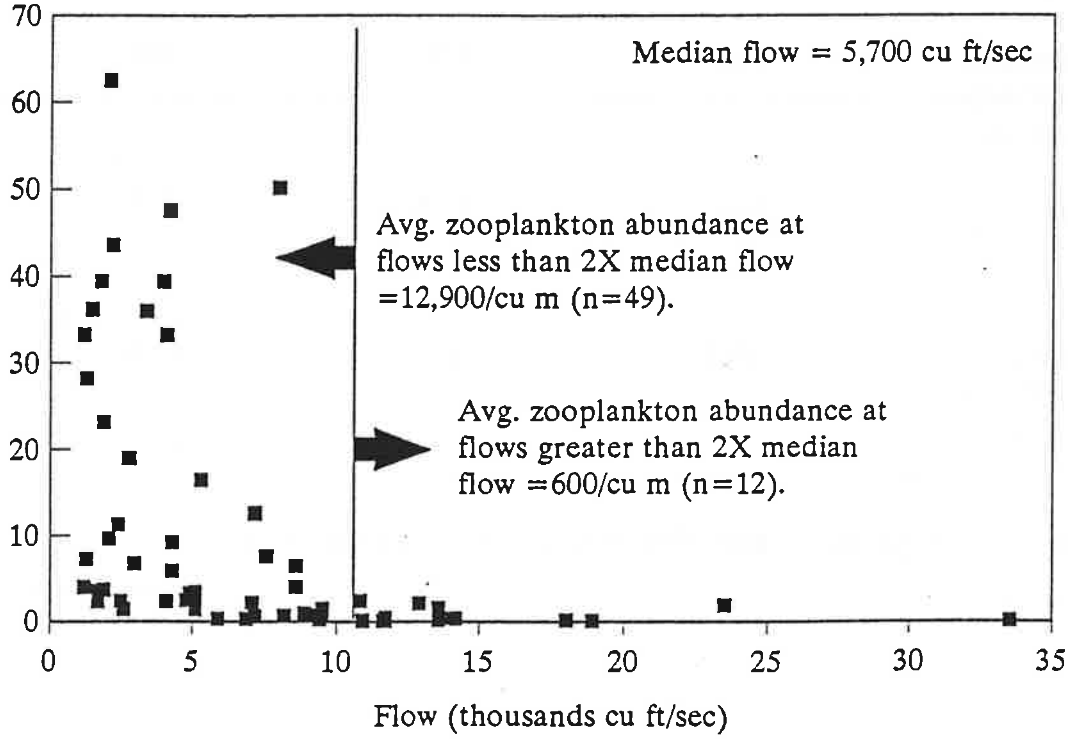

Figure 2. Mesozooplankton abundance versus flow at James River station TF5.5.

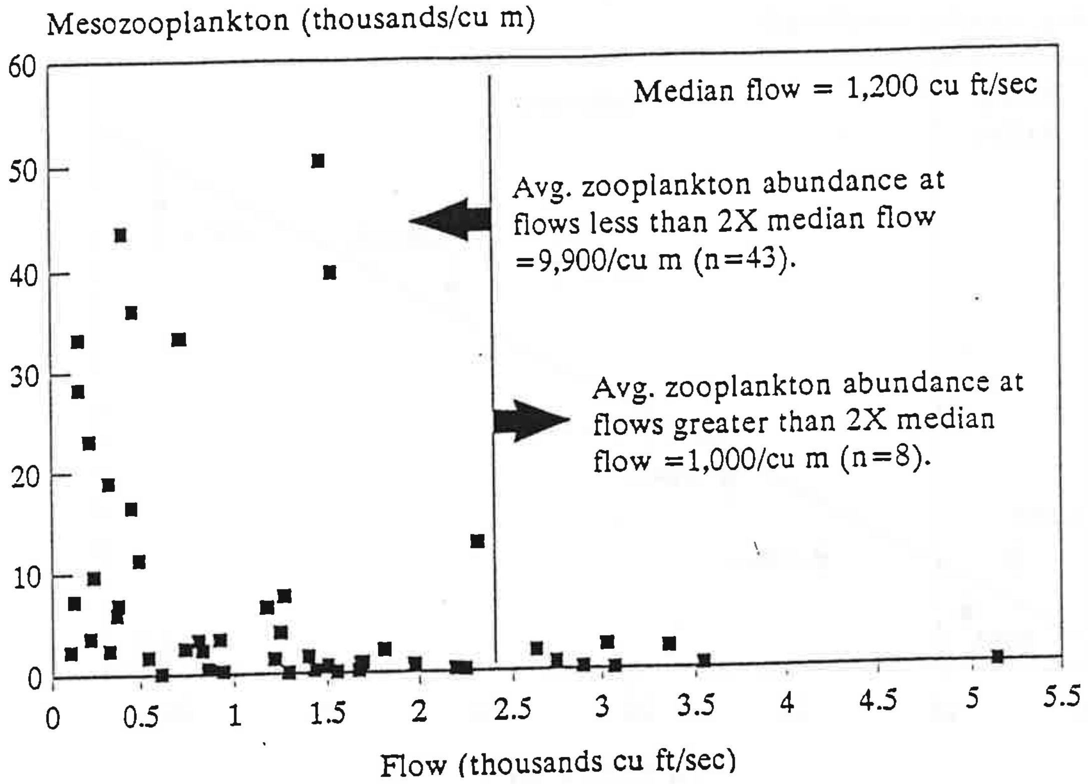

Figure 3. Mesozooplankton abundance versus flow at York River station TF4.2.

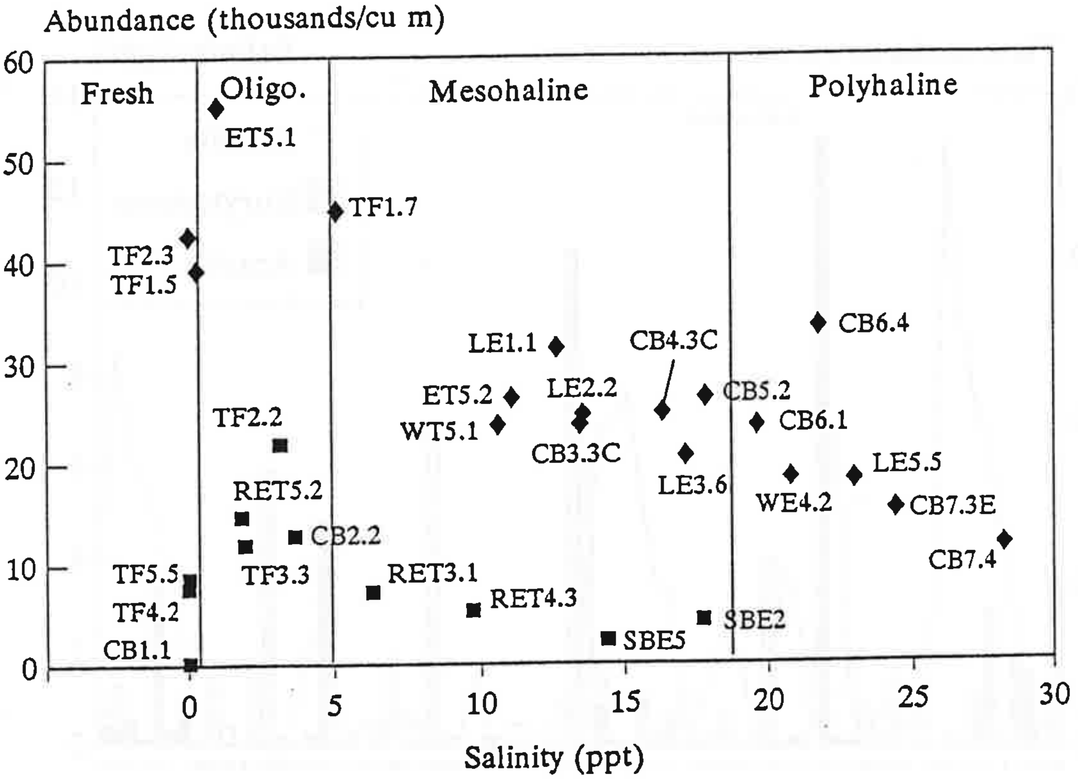

Figure 4. Average mesozooplankton abundance versus average salinity at all Chesapeake Bay plankton monitoring stations.

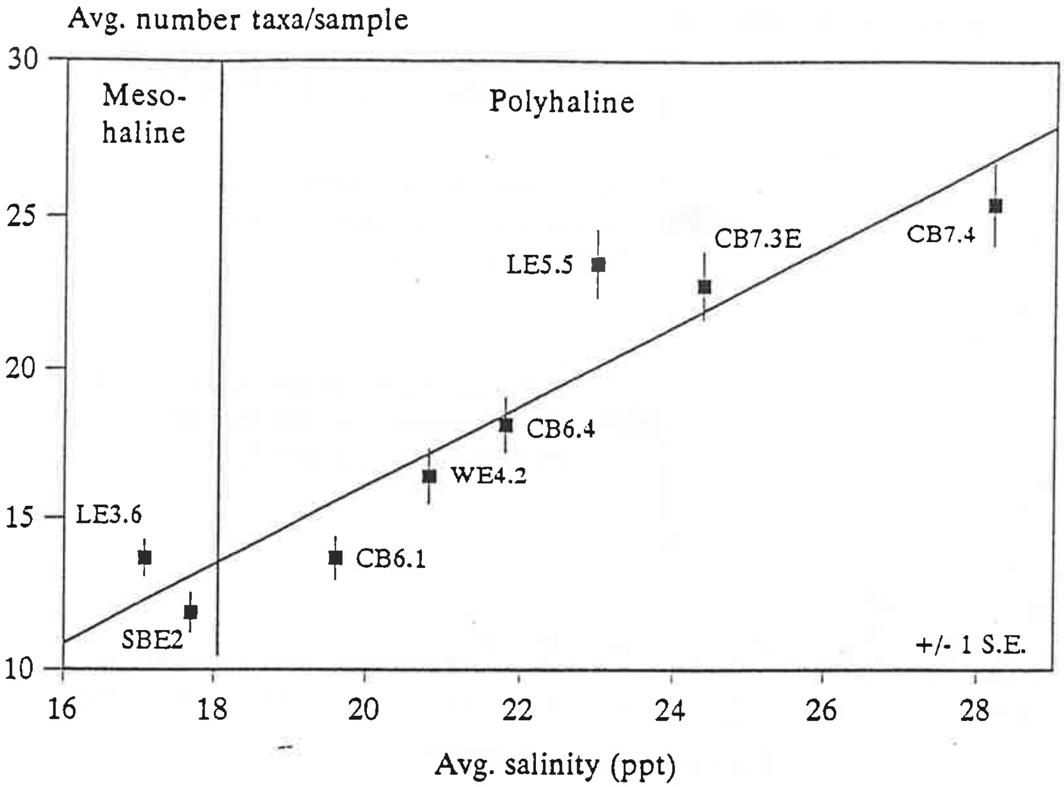

Figure 5. Average mesozooplankton diversity versus salinity at upper mesohaline and polyhaline stations in the Virginia portion of the Chesapeake Bay.

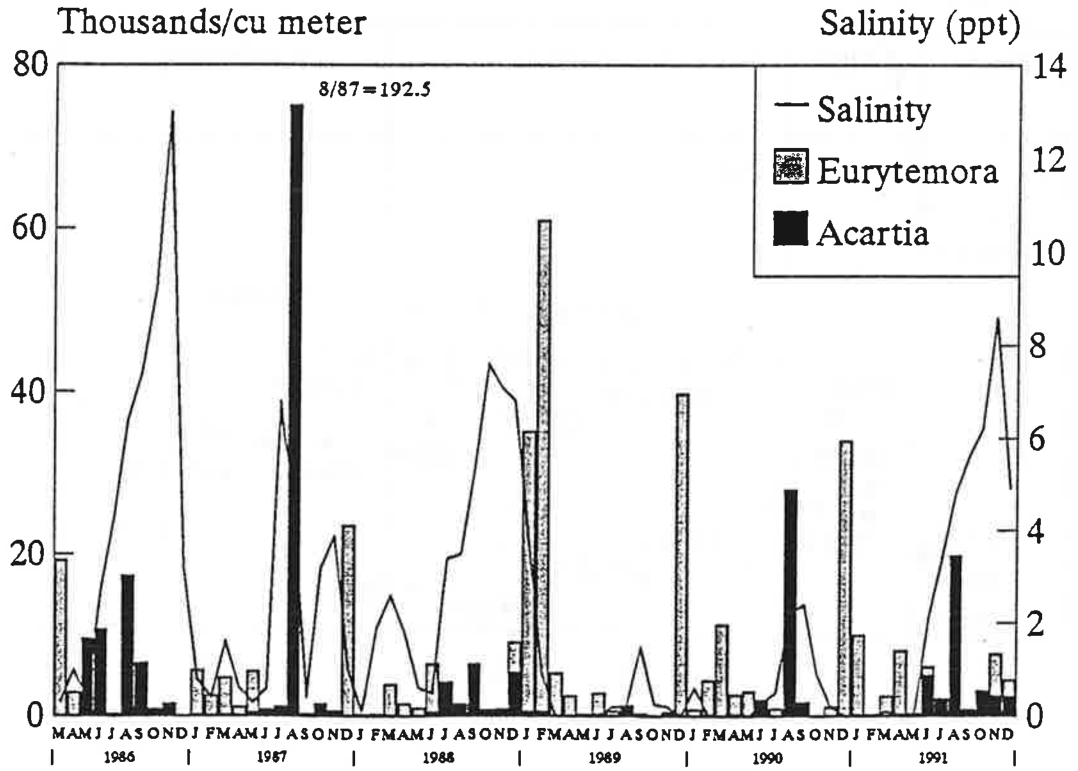

Figure 6. Abundance of *Acartia* and *Eurytemora* versus salinity at Rappahannock River station TF3.3.

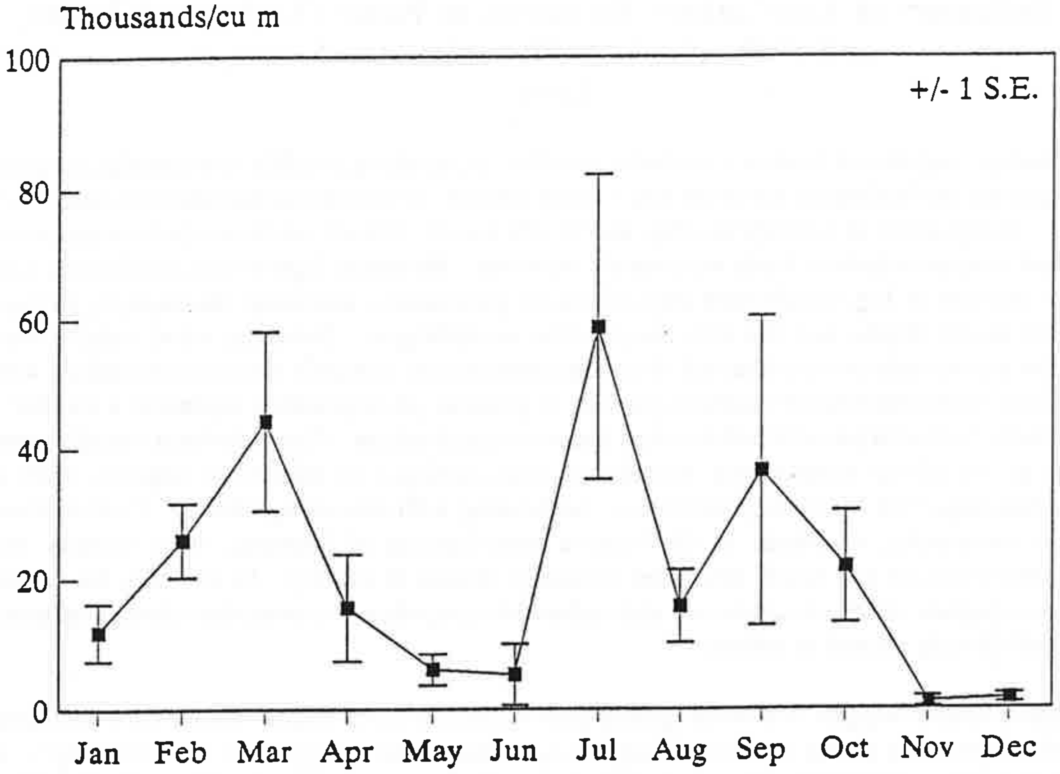

Figure 7. Mesozooplankton abundance averaged by month for station LE3.6, located at the mouth of the Rappahannock River (July 1985 - December 1991).

Indicators of Zooplankton Responses to Water Quality and Toxics, and of Zooplankton-Phytoplankton Linkages

Summary

Multiple regression models were formulated for establishing possible relationships between a suite of water quality and biological variables and various indices for microzooplankton and mesozooplankton from four salinity zones in Chesapeake Bay and its tributaries. Direct relationships between zooplankton and nutrient concentrations or loads were rarely observed. However, significant correlations were noted between a number of log-transformed phytoplankton parameters, including chlorophyll, phytoplankton carbon and secchi depth, and the two zooplankton assemblages. Strongest relationships were noted between the phytoplankton variables and the microzooplankton; similarly strong relationships were noted for tidal-fresh mesozooplankton assemblages but, in general, phytoplankton explained a smaller fraction of the variation in the larger zooplankton than the microzooplankton. Phytoplankton variables were more important in describing zooplankton assemblage characteristics in tidal-fresh regions, with linkages becoming less important (declining correlation coefficients) with increasing salinity. Contributions of the cladoceran community, examined as the relative contributions of *Bosmina*, were directly related to phytoplankton biomass and secchi depth and inversely related to salinity. In contrast, the contributions of calanoid copepods relative to cladocera and cyclopoid copepods were inversely related to phytoplankton biomass and directly related to salinity.

These results suggest that water quality-phytoplankton-zooplankton linkages are well established in low salinity portions of the watershed, suggesting nutrient and energy flow is "bottom-up". Because of the relative strong correlations between variables routinely estimated in the Water Quality Monitoring Program and zooplankton, reasonable predictions of zooplankton abundance and biomass could be made for a number of low salinity areas currently sampled in the on-going MDE and VA programs.

Hypotrich ciliate abundances were strongly related to hypoxic-anoxic conditions at 6 stations experiencing low dissolved oxygen at least a few times each year. Planktonic hypotrichs were also noted at several stations with dissolved oxygen (DO) concentrations ≥ 2 mg liter^{-1}, however, suggesting that hypotrichs might provide short-term memory of very low DO events in an area. Additional studies are proposed for determining the usefulness of these indicators in future monitoring efforts.

Metal pollutants chronically suppress zooplankton populations in the Elizabeth River, and probably the Potomac River oligohaline as well. Unmonitored pollutants are likely to be involved in suppressing zooplankton populations at other stations for which no alternative causes seem to be present.

Zooplankton Linkages to Water Quality and Biological Variables Collected in the Water Quality Monitoring Program

by Kevin G. Sellner and Raymond W. Alden III

Introduction

Since the pioneering work of Vollenweider (e.g., Vollenweider and Dillon, 1974) and Schindler (1974) regarding nutrient-chlorophyll relationships in freshwater lakes, researchers and managers have been trying to relate changes in species composition of the pelagic and benthic biota to nutrient stock variability. Vollenweider's and Schindler's success in predicting phytoplankton biomass as a function of nutrient loads and concentrations encouraged exploratory analyses for identifying bottom-up controls of phytoplankton species, biomass and productivity (e.g., O'Brien and DeNoyelles, 1974; Carlson, 1977; Stockner and Shortreed, 1985; Smith, 1983; Gophen et al., 1990; Peters, 1991) and, in turn, the planktonic herbivores that feed on these substrates (O'Brien and DeNoyelles, 1974; Makarewicz and Likens, 1979; McCauley and Kalff, 1981; Bays and Crisman, 1983; DeCosta et al., 1983; Blancher, 1984; Hanson and Peters, 1984; Pace, 1984; Yan, 1986; Vanni, 1987; Siegfried et al., 1989; Zankai, 1989).

The next step, establishing water quality linkages to the zooplankton, has been attempted in many freshwater systems. Phosphorus, as the major nutrient limiting phytoplankton in most freshwater lakes, was found as a good predictor for zooplankton biomass and community composition, most often via P control of chlorophyll and phytoplankton species composition. Hanson and Peters (1983) established strong relationships between total phosphorus and zooplankton biomass in several Canadian lakes. Multiple regression results found that TP and total depth explained 75% of the variation in total zooplankton biomass. Yan (1986) estimated that TP was the best predictor ($r^2=0.63$) of total crustacean zooplankton biomass in several other Canadian shield lakes. In a comprehensive review of freshwater systems, Pace (1984) reported that zooplankton biomass strongly covaried with total phosphorus.

Phytoplankton control of zooplankton, ultimately based on phytoplankton assimilation of nutrient stocks, has been modeled more frequently. For example, McCauley and Kalff (1981) observed that phytoplankton biomass explained 40% of the variation in zooplankton biomass in eastern Canadian lakes. Chlorophyll or a surrogate for phytoplankton, secchi depth, explained 57% and 59%, respectively, of the variation in zooplankton biomass in several other Canadian systems (Hanson and Peters, 1984). Bays and Crisman (1983) noted that annual mean chlorophyll was an excellent predictor of total zooplankton biomass in more than 50 Florida lakes. Thus, water quality-phytoplankton-zooplankton linkages have been established in many freshwater lacustrine habitats.

Strong relationships have also been identified between water quality, phytoplankton and specific zooplankton groups. With some debate (Matveeva, 1991), rotifer abundance and species composition have been suggested as indicators of system trophy (Dzyuban and Kuznetsova, 1979; Mäemets, 1983; Sládeček, 1983; Blancher, 1984; Foissner, 1988) with increasingly eutrophic conditions selecting specific rotifer assemblages. Further, increasing P has been directly implicated in the abundance of rotifers (DeCosta et al., 1983; Johansson, 1983) and cladocera (O'Brien and DeNoyelles, 1974; DeCosta et al., 1983; Vanni, 1987). Siegfried et al. (1989) determined that rotifer abundance was directly related to total phosphorus levels as well as chlorophyll concentrations. On a species level, *Synchaeta* biomass could be predicted from phytoplankton carbon for a portion of the Baltic Sea (Johansson, 1983). Zankai (1989) found that rotifer numbers were 10-fold higher in areas of Lake Balaton with high primary productivity.

Cladocerans have been strongly tied to eutrophy as well. Chlorophyll a concentrations were directly related to average cladoceran dry weights in enriched ponds of the great plains; cladoceran biomass increased with increasing fertilizer addition (O'Brien and DeNoyelles, 1974).

These results suggest that quantifiable linkages between water quality and phytoplankton and zooplankton may exist in many aquatic systems, particularly freshwater environments. The Biological Monitoring Component of the Chesapeake Bay Water Quality Monitoring Program has compiled a suite of water quality and biological parameters over the last nine years in most of MD's and VA's watershed. These data sets include variables that have been successfully used in predicting planktonic biomass and productivity in limnology (see above) and provide the opportunity for assessing their importance as predictive tools in the rivers and estuaries of the Bay currently and historically receiving large nutrient loadings through cultural eutrophication.

In the present study, multiple regression analyses were undertaken to determine predictable patterns between water quality and zooplankton biomass and community composition in the watershed. Water quality and biological monitoring data from each salinity zone within the Bay watershed were analyzed separately. The analysis was designed to include those parameters previously identified as important in freshwater systems (e.g., TP, phytoplankton biomass, secchi depth) as well as those parameters that might mimic those parameters in saline waters. Other variables in saline reaches of the system included the major nutrient limiting summer productivity in saline reaches, nitrogen, as well as some measure of water column stratification and potentially deleterious oxygen concentrations. Total depth was also included as one variable in fresh and saline habitats.

Methods

Multiple regression models were developed to explore the relationships between zooplankton bioindicators and environmental variables. Details of the statistical programs are presented in Appendix C. Briefly, the overall approach taken can be summarized as follows. Many of the variables, whether dependent (i.e., the bioindicators) or independent (i.e., the environmental variables) are correlated to temperature. Therefore, to reduce the potential for spurious relationships due to common covariance with temperature, the data sets were "corrected" for temperature effects prior to the regression analyses. The temperature effect on each variable was assessed independently by regression analysis and the residuals from these analyses were output as "temperature-corrected data" for all subsequent analyses. Since salinity and season also greatly influence both dependent and independent variables, the regression models were run for each of the four salinity regimes (tidal fresh, oligohaline, mesohaline, and polyhaline) for the seasons of "spring" (March-May) and "summer" (June-September).

Initially, multiple regression models of the bioindicators versus a suite of environmental variables were run with "raw" temperature-corrected data. A second set of models were run with log-transformed temperature-corrected data to examine whether nonlinear relationships provided a better fit. A final set of models were developed which contained only the variables found to be significant in the initial runs. It should be noted that this approach was considered to be exploratory in nature. As with most environmental data sets, the biological and environmental variables used in the regression models probably did not meet all of the assumptions of parametric statistics (e.g., normality, homogeneity of variances, independence of error terms, etc.). Therefore, while regression analyses tend to be quite robust, violations of the assumptions for these data sets could have affected Type I (α) or Type II (β) errors. Due to time and resource limitations and the preliminary nature of the analyses, these effects were not explored. Most violations of assumptions (particularly those involving autocorrelation and other types of nonindependence of errors) tend to elevate Type I errors (i.e., produce a greater probability of apparently significant values than would be expected by chance alone). Furthermore, the large number of individual regression statistics produced by the numerous model runs for the bioindicators in each salinity regime-season combination would tend to produce a percentage of significant relationships by chance alone. Thus, the statistical criterion of $\alpha=0.01$ was selected in an attempt to make these exploratory assessments somewhat more conservative. Nonetheless, the reader is cautioned that some of the "significant" relationships observed may be due to these statistical factors. Therefore, the relationships that have been emphasized in this exploratory study were those that were consistent with previously reported (*a priori*) correlations or current ecological theory, as well as those that "made sense" with the other patterns that emerged from the various models. This degree of subjectivity was deemed to be warranted in the context of this preliminary, hypothesis-generating study.

Four regression models were developed, based largely on previously observed relationships reported in the literature (see INTRODUCTION):

Model 1 was described as follows:

$$(1)\ \ \mathrm{IND} = \beta_1\mathrm{TN} + \beta_2\mathrm{TP} + \beta_3\mathrm{CHL\ a} + \beta_4\mathrm{BDO} + \beta_5\mathrm{DSAL} + \beta_6\mathrm{CYN\ C} + \beta_7\mathrm{SEC} + \beta_8\mathrm{TDEP}$$

where IND is the zooplankton indicator, TN and TP are total nitrogen and phosphorus ($\mathrm{mg \cdot L^{-1}}$), respectively, CHL a is active chlorophyll a ($\mu\mathrm{g \cdot L^{-1}}$), BDO is bottom dissolved oxygen concentration ($\mathrm{mg \cdot L^{-1}}$), DSAL is the change in salinity from surface to bottom, CYN C is biomass of the cyanobacteria ($\mu\mathrm{gC \cdot L^{-1}}$) and SEC and TDEP are the secchi depth and total depth (m), respectively.

Model 2, relating zooplankton variables to loads in the tributary stations, was as follows:

$$(2)\ \ \mathrm{IND} = \beta_1\mathrm{TNTRANS} + \beta_2\mathrm{TPTRANS} + \beta_3\mathrm{SEC} + \beta_4\mathrm{TDEP} + \beta_5\mathrm{DSAL}$$

where TNTRANS and TPTRANS represented the product of fall line flow and nutrient concentration ($\mathrm{kg \cdot mon^{-1}}$).

Model 3 focused on phytoplankton biomass as chlorophyll as the primary biological descriptor:

$$(3)\ \ \mathrm{IND} = \beta_1(\mathrm{CHL\ a}) + \beta_2\mathrm{SEC} + \beta_3\mathrm{TDEP} + \beta_4\mathrm{DSAL}$$

where DSAL was not used in the model in tidal fresh stations.

Model 4 was developed for examining phytoplankton carbon as the primary controlling factor for the zooplankton:

$$(4)\ \ \mathrm{IND} = \beta_1(\mathrm{TOT\ C}) + \beta_2\mathrm{SEC} + \beta_3\mathrm{TDEP} + \beta_4\mathrm{DSAL}$$

and as in Model 3, DSAL was not included in application of the model in tidal fresh stations.

Relationships between a variable and the indicator were derived from estimation of the F statistic, computed from model mean square $\cdot$ (error mean square)$^{-1}$. Variation in the INDICATOR, computed from model sum of squares $\cdot$ (corrected sum of squares)$^{-1}$, as a function of variation in the independent variables was reported as r^2.

Results

Model 1. (Zooplankton as a function of nutrients, phytoplankton biomass, oxygen, salinity and depth)

Total Mesozooplankton Biomass and Abundance. There was little apparent consistent relationship between any of the variables in Model 1 and total mesozooplankton biomass. Total phosphorus was positively related to total mesozooplankton biomass during summer in stations in oligohaline and polyhaline regions of the watershed (r^2=0.10-0.18). Because *Acartia* is numerically dominant from late spring through summer and therefore responsible for the majority of the mesozooplankton biomass, there are also significant relationships between TP and total mesozooplankton abundance: total mesozooplankton abundance is positively related to TP in spring in the polyhaline reaches of the system and in summer in mesohaline and polyhaline areas (r^2's=0.25, 0.0.05 and 0.18, respectively). These patterns reflect coincidence and not cause and effect. *Acartia* spp. numbers and biomass increase from spring through summer, paralleling P recycling in the system. Total mesozooplankton biomass in spring in polyhaline areas of the Bay was also weakly related to bottom dissolved oxygen concentrations (r^2=0.19) while total depth was a significant but poor predictor of biomass in spring oligohaline reaches of the system (r^2=0.09).

Ratio of Calanoids to Cyclopoids + Cladocerans. With increasing eutrophic conditions, the numbers of cladocerans and cyclopoids increase while densities of calanoid copepods decline (Makarewicz and Bertram, 1991). Thus, the ratio of calanoid copepods to cyclopoid copepods and cladocerans should decline with increasing environmental stress resulting from high nutrient loads. For those regions with consistently large contributions of cyanobacteria, i.e., the tidal fresh Potomac River, the ratio should therefore be low during bloom periods and increase with effective management elimination of the nuisance algae. Unfortunately, there was no consistent result for tidal fresh regions during summer. The ratio was positively related to TP concentrations (p=0.0001), contrary to the expected pattern, and inversely related to TN (p=0.0003) and cyanobacteria biomass (p=0.0006), in agreement with the expected pattern. Thirty-five percent of the variation in the ratio was explained by the regression model $\log RATIO = 14.21 \cdot \log TP - 2.74 \cdot \log TN - 0.13 \cdot \log CyanC + 1.14$.

In more saline reaches of the watershed, phosphorus was not important in explaining variability in the ratio of calanoids to cladocerans and cyclopoids. In spring, the ratio in mesohaline reaches of the system was positively related to TN, bottom DO and negatively related to cyanobacteria biomass (r^2=0.28). Considering that calanoids, nitrogen (as DIN) and bottom DO are all high at this time and that the contribution of cyanobacteria carbon to total phytoplankton carbon in mesohaline areas should be small, the authors urge cautious acceptance of this relationship as a useful zooplankton bioindicator for spring <u>mesohaline</u> regions.

Salinity, depth and cyanobacteria biomass could also partially explain the proportion of calanoids to cyclopoids plus cladocerans in oligohaline stations during spring and summer. In spring, calanoid densities would definitely increase (*Acartia* increase) with increasing salinity, leading to a direct relationship between the ratio and salinity; cladocerans also decline as salinity increases. A negative relationship with depth may be explained by the concentrating effect of rapid shoaling in mixing areas of some tributaries, leading to the concentration of many calanoids common to deeper estuarine waters downriver into shallow water columns of the upper rivers. Thirty-five percent of the variation of the ratio data could be explained by these two parameters. In summer, the ratio of calanoids to cyclopoids and cladocera was also inversely related to total depth and to cyanobacteria biomass, as expected (see discussion above). In addition, however, the ratio was directly related to salinity, via the contributions of copepods from the estuary, and to chlorophyll, probably through the net particle trapping mechanism inherent to the freshwater-seawater interface of most estuaries.

The inverse relationship between salinity and cladocera is exemplified by the results of the regression analyses of percent *Bosmina* versus model variables. The relative contribution of this taxon was inversely related to salinity in oligohaline regions in spring and summer (r^2's=0.22, 0.26, respectively), indicating the detrimental effect of increasing cation concentration on this planktonic herbivore. Further, this taxon was positively related to biomass of the cyanobacteria in oligohaline summer waters, the pattern expected from the ratio and cyanobacteria.

Microzooplankton Biomass and Abundance. Spring microzooplankton (>44 μm) abundance and biomass in tidal fresh regions of MD's portion of the watershed were partially explained by ambient chlorophyll concentrations and inversely related to bottom DO levels ($r^2=0.61$). The relationship to chlorophyll was not surprising considering that similar relationships have been noted in other freshwater systems. However, the inverse relationship to bottom DO, when oxygen concentrations in bottom waters were appreciably aerated, was unexpected but probably relates to decreasing oxygen saturations as water temperatures increase in the spring. Bottom oxygen levels also explained mesohaline microzooplankton biomass in the spring, although only modestly ($r^2=0.06$).

Phytoplankton biomass, as chlorophyll, was correlated with microzooplankton biomass in the summer in oligohaline reaches of MD's watershed, with biomass inversely related to total depth ($r^2=0.52$). The strong tie to food has been noted previously, while the relationship to depth can be explained by dilution of planktonic stocks as water columns increase in depth. Variation in microzooplankton biomass was weakly related to chlorophyll stocks in summer, mesohaline Bay waters ($r^2=0.15$).

As the principal members of the microzooplankton, rotifers (as biomass) followed the same pattern identified for total microzooplankton biomass and abundance in the spring, i.e., 43% of the variation in biomass could be explained by chlorophyll and bottom DO. That chlorophyll also explains 28% of rotifer biomass in spring oligohaline reaches of MD's system probably reflects spring transport of high freshwater plankton into oligohaline reaches of the system. Mesohaline rotifer biomass in summer was inversely related to secchi disc depth, a function of phytoplankton-dominated surface waters ($r^2=0.33$). The tie to secchi depth and suspended phytoplankton complements the positive relationship noted between total microzooplankton biomass and chlorophyll in summer, mesohaline areas of MD's water; however, there was no relationship noted between rotifer biomass and chlorophyll.

Model 2. (Tidal-fresh and oligohaline zooplankton as a function of flow-related variables)

Regression analyses indicated possible relationships between flow-controlled nutrient loads and zooplankton. Total mesozooplankton abundance was inversely related to total phosphorus loads in tidal-fresh reaches of the system in spring ($r^2=0.53$). In contrast, microzooplankton biomass was inversely related to total N loads ($r^2=0.33$). Rationalization of these results is difficult. However, the relationships might reflect <u>flow</u> and not load. That is, the transport term was derived from the product of flow and concentration with the flow term dominating. The inverse relationship might therefore reflect an inverse relationship to flow, with high flow washing the animals from the region (see R. Birdsong chapter).

Flow-induced vertical salinity gradients in oligohaline reaches of the system resulted in marked reductions in cladocerans, including *Bosmina*. As a result of increasing salinity reducing contributions of cladocerans, percentage contributions of *Bosmina* declined in spring and summer as an inverse function of the salinity gradient (r^2's=0.28, 0.25 respectively). In addition, the ratio of calanoids to cyclopoids and cladocerans was directly related to salinity stratification, indicative of gravitational circulation in the estuaries in spring and to a lesser extent, summer (r^2's=0.32, 0.13, respectively). In summer, the relative contributions of *Bosmina* was also partially explained by secchi depth, with higher cladoceran contributions a function of increasing secchi depth, probably due to lower flow leading to lower suspended sediments and a greater portion of phytoplankton in the suspended particulates.

Model 3. (Zooplankton versus water column particulates and vertical stratification)

Direct relationships between zooplankton and potential food, as phytoplanktonic chlorophyll concentrations, was most rigorously addressed in Model 3. As expected from previous freshwater research, microzooplankton, and to a lesser extent, mesozooplankton were directly explained by chlorophyll alone or the pigment and other factors.

Mesozooplankton Abundance and Biomass. Total mesozooplankton abundance was directly related to chlorophyll and secchi depth in tidal-fresh reaches of the system in spring (r^2=0.42). In summer tidal-fresh regions, mesozooplankton abundance and biomass were inversely related to secchi depth (r^2's=0.35, 0.18, respectively), a function of phytoplankton and smaller non-algal suspended particulates. *Bosmina,* as a percentage of mesozooplankton, was directly related to secchi depth in tidal-fresh (r^2=0.14) and oligohaline regions in summer (r^2=0.22), possibly coincident with large contributions of cyanobacteria in oligohaline areas of the Potomac River from 1985-1988. In summer, increasing salinity in the region lead to a reduction in the contributions of the cladoceran and partial explanation of the declining relative contribution of *Bosmina.*

Ratio of Calanoids to Cyclopoids + Cladocera. The importance of increasing saline contributions in oligohaline reaches is further supported by the direct relationship between calanoids to cyclopoids and cladocera and vertical salinity gradient in spring and summer oligohaline reaches (r^2's=0.35, 0.23, respectively).

The ratio of calanoids to cyclopoids plus cladocerans was inversely related to secchi depth in three salinity zones in summer, tidal-fresh, mesohaline and polyhaline regions, as well as the polyhaline region in spring, suggesting increasing calanoid densities with shallowing euphotic depth. This trend is not unexpected considering the direct relationship of *Bosmina* to secchi depth; as secchi depth declines, *Bosmina* would decrease, increasing the ratio of calanoids to cyclopoids + cladocera.

Microzooplankton Abundance and Biomass. The most dramatic and consistent results were observed between microzooplankton and chlorophyll. Of the 18 possible tests relating microzooplankton to chlorophyll, chlorophyll was found significant in explaining variations in microzooplankton 14 times. No relationships, or weak relationships, were noted between spring microzooplankton abundance and biomass and chlorophyll for mesohaline areas, not surprising considering the large number of non-herbivorous microzooplankton found in saline samples, e.g., many tintinnids and other ciliates. However, even summer mesohaline microzooplankton was positively related to chlorophyll (r^2=0.15), likely due to rotifer biomass in the period. Variations in chlorophyll could explain 13%-52% of the variation in microzooplankton abundance and biomass as well as rotifer biomass in the region. The only other variables contributing to microzooplankton were secchi depth, a surrogate for chlorophyll in mesohaline environments (2 times), and total depth (once).

The ratio of microzooplankton biomass to mesozooplankton biomass was positively correlated with secchi depth in tidal-fresh regions of MD's watershed in spring and summer (r^2's=0.36, 0.16, respectively). These results probably reflect the large contribution of microzooplankton biomass to total zooplankton during the 1985-1988 cyanobacteria-dominated period of the upper Potomac.

Model 4. (Zooplankton as a function of phytoplankton carbon, secchi and total depth and salinity stratification)

Relationships between zooplankton and phytoplankton biomass as carbon were observed and were in general similar to zooplankton-chlorophyll relationships identified in Model 3.

Mesozooplankton Abundance and Biomass. In general, phytoplankton carbon was less strongly linked to mesozooplankton abundance and biomass than observed for chlorophyll and the grazers and more importantly, mesozooplankton was inversely related to the available phytoplankton stocks. R^2's were low, ranging from 0.13-0.18, for model results between the variables carbon and total depth and the dependent mesozooplankton abundance in oligohaline reaches in spring and summer. In polyhaline waters during summer, the relationship between phytoplankton carbon and mesozooplankton abundance was weaker yet, with r^2=0.07. Mesozooplankton biomass was similarly explained by variations in phytoplankton carbon, total depth and secchi depth, with oligohaline spring and summer results comparable (r^2's=0.19 and 0.18). In tidal-fresh reaches of the system, secchi depth was the only variable that was related to mesozooplankton abundance or biomass; secchi depth was inversely related to these mesozooplankton stock estimators for both spring and summer.

In contrast to total mesozooplankton abundance and biomass, the relative contributions of *Bosmina* to the total community was directly related to secchi depth in tidal-fresh and oligohaline regions during summer and as well, to total phytoplankton carbon in oligohaline summer reaches of the watershed. Increasing salinity stratification was inversely related to the contribution of *Bosmina*, as noted in the results above.

The importance of calanoid copepods to contributions of the cyclopoids plus the cladocerans reflected these responses to total phytoplankton carbon and secchi depth: as phytoplankton carbon and secchi depth declined, calanoids increased while cladocerans (*Bosmina*) decreased. This result is consistent with the inverse relationship noted between the ratio of calanoids to cyclopoids and cladocera and cyanobacteria carbon in summer, tidal-fresh waters (see Model 1 results). Surprisingly, the inverse relationship of the ratio to carbon and/or secchi depth was noted not only in tidal-fresh, phytoplankton-rich areas but also through spring and summer oligohaline and mesohaline regions. In polyhaline regions in spring and summer, secchi depth continued to be inversely related to the ratio of calanoids to cyclopoids plus cladocera but, at least for spring, the ratio was directly related to total carbon pools. This pattern in the most saline regions might reflect the fact that the majority of particles responsible for secchi depth are phytoplankton and therefore, as phytoplankton increases (carbon pools are larger), secchi depth shallows due to the accumulation of the phytoplankton food particles. Again, the inverse relationship between secchi depth and the ratio was also observed in results from Model 3.

Microzooplankton Abundance and Biomass. Relatively strong linkages were observed between microzooplankton abundance, biomass and rotifer biomass and total phytoplankton carbon. In contrast to the inverse relationships noted between mesozooplankton abundance and biomass and phytoplankton carbon noted above, microzooplankton standing crops were directly related to the available phytoplankton carbon, or alternatively, inversely to secchi depth, a result of increasing particle concentrations, i.e., phytoplankton. Strongest linkages between microzooplankton and phytoplankton were noted in tidal-fresh areas, generally declining with increasing salinity. Phytoplankton carbon and secchi depth explained 30-48% of the variation in microzooplankton abundance or biomass in tidal-fresh regions in the spring and summer; phytoplankton carbon and secchi depth in oligohaline regions and secchi depth only in mesohaline regions explained 13-49% of the variation in microzooplankton densities and biomass for oligohaline and mesohaline reaches.

As noted for Model 3, the ratio of microzooplankton to mesozooplankton biomass was directly related to secchi depth in tidal-fresh areas in spring and summer (r^2's=0.36 and 0.16, respectively), reflecting the a more rapid increase in microzooplankton to a declining secchi depth than observed for the mesozooplankton.

Discussion

At first glance, the weak but positive relationships noted between total mesozooplankton biomass as well as abundances of mesozooplankton (primarily *Acartia* spp.) and TP in saline reaches of the system (Total mesozooplankton biomass and TP, Total *Acartia* spp. biomass, Model 1) were somewhat surprising in that previous research had noted this pattern in eutrophic freshwater systems (Hanson and Peters, 1983; Yan, 1986). Increasing TP in summer more than likely results from high DIP flux from anoxic sediments of the saline Bay and its tributaries (Boynton et al., 1993) and mixing into the overlying water column, coincident with but not the cause of high zooplankton biomass in the summer. Zooplankton regenerated P is substantial, but dominated by microzooplankton activity (W.R. Boynton, unpubl. data). Mesozooplankton regenerate approximately 10% of summer recycled P, further supporting coincidence rather than a cause-and-effect relationship. Tight P-ZP relationships might have been expected in tidal fresh portions of the Potomac River, for example, where high P loads and TP concentrations are routinely noted, coincident with high phytoplankton biomass (as chlorophyll or carbon) and the associated zooplanktonic herbivores. Absence of the pattern, however, might reflect the shift from rotifers and small *Bosmina* during cyanobacteria dominance from 1985-1998 to larger copepods after 1988 when cyanobacteria and total phytoplankton carbon declined in the system.

The ratio of calanoids to cyclopoids plus cladocerans followed the expected pattern with cyanobacteria biomass and salinity. Cyanobacteria form dense blooms in summer particularly in tidal fresh and oligohaline Potomac River areas from 1985-1988, favoring high *Bosmina* contributions and at least in theory, fewer copepods. With increasing salinity downriver and in the Bay, *Bosmina* will be eliminated leading to a high ratio, both patterns noted in results from Models 1 and 2.

One explanation for the inverse relationships between any of the zooplankton parameters and spring bottom dissolved oxygen concentrations reflects temperature-controlled saturation of the water and not oxygen concentrations *per se*. Olson (1987) found most of the variation in zooplankton in mesohaline Chesapeake Bay could be explained by the annual temperature cycle and had temperature been one of the variables in the models, temperature would have been strongly related to zooplankton abundances and biomass. Temperature-controlled saturation declines with increasing temperatures in the spring, as zooplankton densities and biomass increase. The negative relationships reflect this pattern. However, additional insight is required as all water quality and biological parameters were "corrected" for temperature prior to the analyses.

Microzooplankton, at least those >44 μm, were more strongly linked to phytoplankton biomass than total mesozooplankton. In Model 1, chlorophyll was strongly related to microzooplankton abundance and biomass and rotifer biomass in spring and summer, observed in previous freshwater systems (see Introduction) as well as the tidal-fresh Potomac River (Sellner et al., in prep.). The linkage between rotifers and the community they dominate and phytoplankton biomass has been indicative of eutrophic freshwater environments. Even stronger relationships, and certainly more consistent throughout all salinity zones studied (tidal-fresh to mesohaline regions), were estimated from Model 3, using chlorophyll, secchi depth, total depth and salinity stratification as variables.

The strong linkages between the >44 μm microzooplankton and chlorophyll, phytoplankton carbon and secchi depth derived from Models 1, 3 and 4 indicate that at least for tidal-fresh and oligohaline reaches of MD's portion of the watershed, bottom-up control of the smallest herbivores is likely. In addition, the relatively high correlations noted between phytoplankton variables and microzooplankton, comparable to those measured in Florida and Canadian lakes, suggest that reasonable estimates of zooplankton might be possible for other systems where water quality parameters are currently or have been measured. The response was largely attributable to rotifers, which co-dominate the microzooplankton assemblage with copepod nauplii.

Except for high correlations between tidal-fresh mesozooplankton abundances and biomass and phytoplankton as chlorophyll or secchi disc depth, correlation coefficients were generally lower for mesozooplankton and phytoplankton variables, not surprising considering that the diverse mesozooplankton assemblage includes herbivores, carnivores and omnivores. In addition, food-limited conditions are rarely observed in at least MD's portion of the watershed (Sellner and Jacobs, in prep.) so linkages between nutrient stocks or their assimilators, the phytoplankton, would be more difficult to identify. Because bottom-up control of zooplankton is infrequent in MD's portion of the Bay, predicting mesozooplankton abundances and biomass from measured water quality parameters will be less exact than for microzooplankton.

Increasing salinity also appeared to weaken the linkages between phytoplankton variables and zooplankton, as correlations declined from tidal-fresh to polyhaline areas. The lower correlation between phytoplankton and zooplankton probably reflects synergistic effects of a much larger suite of variables in the estuary, as compared to the more constant water quality associated with a constant and low ionic strength. In the estuary, nutrients are sub-optimal and water depths (SML and total) much greater than present in most of the tidal-fresh areas. Water quality varies substantially, at short- and long-time scales. DO can be at supersaturated levels at the surface in blooms while immediately below the bloom, bottom waters can be devoid of oxygen (e.g., summer Baltimore Harbor). This stratification leads to drastic differences in Eh and element solubilities, further complicating plankton dynamics in the estuary. These types of problems are less frequently observed in tidal-fresh regions of the system. The increasing complexity of the estuary leads to many factors acting simultaneously to govern distributions of any parameter and hence, the low correlations noted between any phytoplankton variable and the zooplankton.

Attempts at modelling zooplankton as a function of flow-related variables, e.g., nutrient loads, secchi depths, salinity stratification, were unsatisfactory. Nutrient loads and zooplankton were highly correlated with r^2's from 0.25-0.53. However, there were no consistent patterns: mesozooplankton abundance was inversely related to total P load while microzooplankton was indirectly related to N loads in tidal-fresh spring areas. As noted above, inverse relationships to loads might actually reflect wash-out of zooplankton from tidal-fresh regions of the system (see R. Birdsong chapter). Considering that zooplankton in other freshwater systems have been directly related to P loading (see Introduction), inverse relationships are difficult to explain.

Recommendations

The results presented from the regression analyses for the four models suggest rather strong relationships between phytoplankton and zooplankton in low salinity reaches of the system in specific seasons. Future work might focus on individual basins, e.g., the tidal-fresh region of the Potomac River, to see whether some systems have stronger relationships between water quality and plankton than others; the results of the present analyses could be driven by one system. Another area to be considered is the potential linkage between water quality, primary productivity and only herbivorous zooplankton in that other systems have strong responses of the primary planktonic herbivores to photosynthetic production (McCauley and Kalff, 1981); in the present analyses, all rotifers and all mesozooplankton taxa were employed. In the future, the zooplankton might be pre-sorted into herbivores, carnivores, omnivores, etc. and then the models re-run with herbivorous zooplankton only. Development of an index similar to Carlson's Trophic State Index (Carlson, 1977) might also be considered for establishing nutrient-secchi-phytoplankton linkages on seasonal to annual scales, followed by comparisons with temporally consistent zooplankton characters.

One major problem is developing a testable model of the factors effectively regulating zooplankton in more saline reaches of the watershed. In the present study, only weak correlations were noted between most of the water quality and phytoplankton variables and zooplankton. Do mesohaline and polyhaline zooplankton assemblages shift prey preferences dependent on prey availability, i.e., if microzooplankton are very abundant, do mesozooplankton select the small animals versus phytoplankton? Certainly, recent evidence suggests that microzooplankton are major contributors to mesozooplankton diets in several environments, including Chesapeake Bay (Gifford and Dagg, 1988; Stoecker and Capuzzo, 1990; White and Roman, 1992). There are bacterivorous, herbivorous, carnivorous and omnivorous rotifer populations as well (e.g., Guiset, 1977; Pourriot, 1977) so separation of rotifer populations into feeding modes might also be possible.

Developing models for mesohaline and polyhaline regions based on freshwater results could also be unreasonable. Could models found representative for freshwater lakes are too simple for the complex estuarine system? Nutrient concentrations in the estuary during spring and summer are relatively stable while turnover is extremely rapid. Establishing nutrient-plankton relationships in these areas must include some recycling term since primary production is derived from regenerated nutrient pools (see Malone, 1992). Nutrient recycling thus becomes one of the major obstacles in modelling water quality-plankton interactions in estuarine portions of the watershed.

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Microzooplankton as Bioindicators of Low Dissolved Oxygen

by Kevin G. Sellner and Stella G. Brownlee

Introduction

Hypotrich ciliates have been identified as microaerophiles and are frequently found in hypoxic or anoxic waters of several environments (Finlay, 1982; Fenchel et al., 1990). The ciliates emerge from pore waters into the overlying water column in response to anoxia and the accumulation of sulfide in the sediments and several taxa, e.g., *Euplotes* spp., grow most rapidly at oxygen tensions from 6-8% atm sat (Fenchel et al., 1989). Unique ciliates have also been reported for other oxygen-poor environments of the Baltic and Adriatic (Revelante and Gilmartin, 1990; Setälä, 1991). As a side effect of increasing nutrient loads, excessive productivity and high oxygen demand leading to anoxia, the presence/absence of hypotrich ciliates in the plankton was examined as an "indicator" of bottom anoxia. If hypotrichs appeared only during or immediately following bottom water anoxia, the ciliates could prove an effective planktonic indicator of low DO intrusions. If, on the other hand, hypotrichs were not restricted to low DO concentrations but appeared when DO levels were substantially elevated, the ciliates would not prove an effective indicator of DO history.

Methods

Hypotrich ciliate abundances estimated for composite samples collected below the pycnocline (including *Euplotes* spp.) were plotted versus bottom dissolved oxygen concentration for those stations experiencing seasonal bottom water hypoxia or anoxia (CB3.3C, CB4.3C, CB5.2, XDE5339, MLE2.2, WT5.1). Coincidence of hypotrichs with rapidly declining DO levels were summed for the period 1985-1992.

Results

The coincidence of low bottom DO and hypotrichs indicated that these ciliates were a unique indicator of hypoxia or anoxia in each of 6 stations experiencing excessive seasonal oxygen demand. In the main Bay (Fig. 8a), hypotrichs were never observed when DO exceeded 0.5 mg liter^{-1} and appeared to be excellent predictors of very low oxygen tensions. In eutrophic Baltimore Harbor (Fig. 8b), hypotrichs (up to 600 liter^{-1}) were only observed when coincident bottom DO concentrations were $\leq$0.4 mg liter^{-1}. Indications that hypotrichs are indicators of even rare hypoxic/anoxic events are obvious from Fig. 9a, for the lower Choptank River. On the single occasion that hypoxia was observed, hypotrichs appeared in the bottom sample.

There are indications that hypotrichs might act as signals for short-term low DO events or, alternatively, that there might be hypotrich populations that are not restricted to hypoxic or anoxic waters. Hypotrichs were noted in bottom waters of the lower Patuxent River in summer, 1990 when DO concentrations were $<$0.5 mg liter^{-1} (Fig. 9b). However, in 1991 and 1992, hypotrichs were also noted in bottom composite samples when ambient DO concentrations were $>$8 mg liter^{-1}; although data are not shown, a similar pattern was noted for CB4.3C. The overlap of hypotrichs with aerated waters suggests that the animals are remnants of a low DO event occurring at a time when sampling was not possible or that a hypotrich assemblage might be found in aerated waters. This pattern was also seen when average hypotrich abundances over the entire water column were plotted against bottom DO. In contrast to the tight coupling between minimally aerobic water and hypotrichs in comparisons of only below pycnocline hypotrich densities and DO of the bottom water, water column hypotrich abundance was not as tightly linked to the extremely low oxygen tensions in the mouths of several of the larger MD tributaries. For example, hypotrichs were observed in the mouth of the Choptank in July, 1986 (Fig. 10) when coincident bottom DO levels were approximately 2.4-2.8 mg liter^{-1}.

Discussion

The appearance of hypotrich ciliates in bottom waters of the Bay (Fig. 8a), the mouth of the Patapsco (Fig. 8b), Choptank (Fig. 9a), Patuxent (Fig. 9b) and Potomac Rivers was limited to periods when bottom DO concentrations were substantially < 1 mg liter^{-1}. This pattern is consistent with distributions of a number of unique ciliates as well as *Euplotes* sp. in eutrophic Baltic fjords. Fenchel et al. (1990) found this taxon in the oxycline of one of the fjords and suggested that the ciliate was a microaerophile typical of oxygen environments containing 0-40% atm sat.

The appearance of hypotrichs in bottom waters of the mouth of the Choptank River (Fig. 9a) is likely an indicator of intrusion of anoxic bottom waters from the Bay. Bottom DO concentrations were rarely < 1 mg liter^{-1} at this station and only for a short time. The hypotrichs may represent remnant populations resulting from intrusion of hypoxic-anoxic bottom water from the main stem that fills the mouth of the river during lateral oscillations of the pycnocline (Sanford et al., 1990). These intrusions have been identified previously, documented by dying oysters in the lower Choptank (Seliger et al., 1984) and short-term rapid increases in bottom salinity (Sanford et al., 1990).

However, results presented for the mean water column hypotrich densities versus bottom DO (Fig. 10) as well as hypotrichs during aerated periods of 1991 and 1992 in the lower Patuxent River and at CB4.3C indicate that there may be an active hypotrich assemblage at the base of the surface mixed layer (see Dolan, 1988; Fenchel et al., 1990; Revelante and Gilmartin, 1990). Certainly, oxygen minima have been noted in the vicinity of the pycnocline through the Monitoring effort (see CSC data base) and ciliates co-occur with oxyclines in other environments (e.g., Revelante and Gilmartin, 1990). The contribution of water column oxygen minima and microaerophilic ciliates should be considered in future sampling programs. Mixing events could also distribute hypotrichs characteristic of the bottom layer throughout the water column. Alternatively, there might also be aerobic hypotrich populations as Fenchel et al. (1989) have documented wide tolerances to oxygen tensions with three *Euplotes* species. These possibilities support implementation of a small laboratory and field program to determine (1) fine-scale distributions of DO and hypotrichs at oxygen minima near the pycnocline, (2) the range of DO concentrations the hypotrichs tolerate (see Fenchel et al., 1989) as well as (3) animal integrity on exposure to DO concentrations > 2 mg liter^{-1}.

Recommendations

Two factors should be considered in use of hypotrichs as indicators of microaerophilic water columns. First, the sampling design in the Plankton Component of MD's Monitoring Program is inappropriate for detecting hypotrich increases in microaerophilic conditions immediately above the bottom or alternatively, from oxygen minima at or near the pycnocline. Sampling is undertaken by compositing water from at least 5 depths in either the surface mixed layer or below the pycnocline, thus diluting the hypotrich signal with animals from all aerated sample depths. Effective characterization of bottom water hypoxia-anoxia via the presence of hypotrichs would be ascertained by quantifying these ciliates at the oxygen minimum zone and immediately above the bottom. Secondly, duration of anoxia and response of the hypotrichs is currently unknown. It is conceivable that hypotrichs migrate into the overlying water column and remain free-swimming as a function of the duration of the low oxygen event. If the period is long with no reaeration, hypotrichs would reach greater densities higher into the water above the sediments, implying that on re-aeration, hypotrichs might still be present in the plankton for considerable time after the reintroduction of oxygen. Experimental observations are required to determine (1) oxygen concentration leading to emergence of hypotrichs from the sediments, (2) period of exposure to the low DO concentration necessary for emergence from the sediments, (3) hypotrich response (e.g., motility, viability, etc.) to reaeration, (4) duration of planktonic stage following reaeration and (5) presence of a viable hypotrich assemblage in oxygen-deficient regions near the pycnocline. The latter two points are particularly relevant considering the presence of the ciliates at bottom DO concentrations >2 mg liter^{-1}, as in the mesohaline stations of the Choptank and Potomac Rivers.

In summary, the inverse relationship between hypotrich ciliates and DO appears very promising and the feasibility of using these organisms in identifying previous low DO events needs to be investigated in a short term laboratory program. In addition, fine-scale distributions of these organisms should be ascertained in the vicinity of the pycnocline and immediately above the bottom sediments in order to identify unique populations of these environments.

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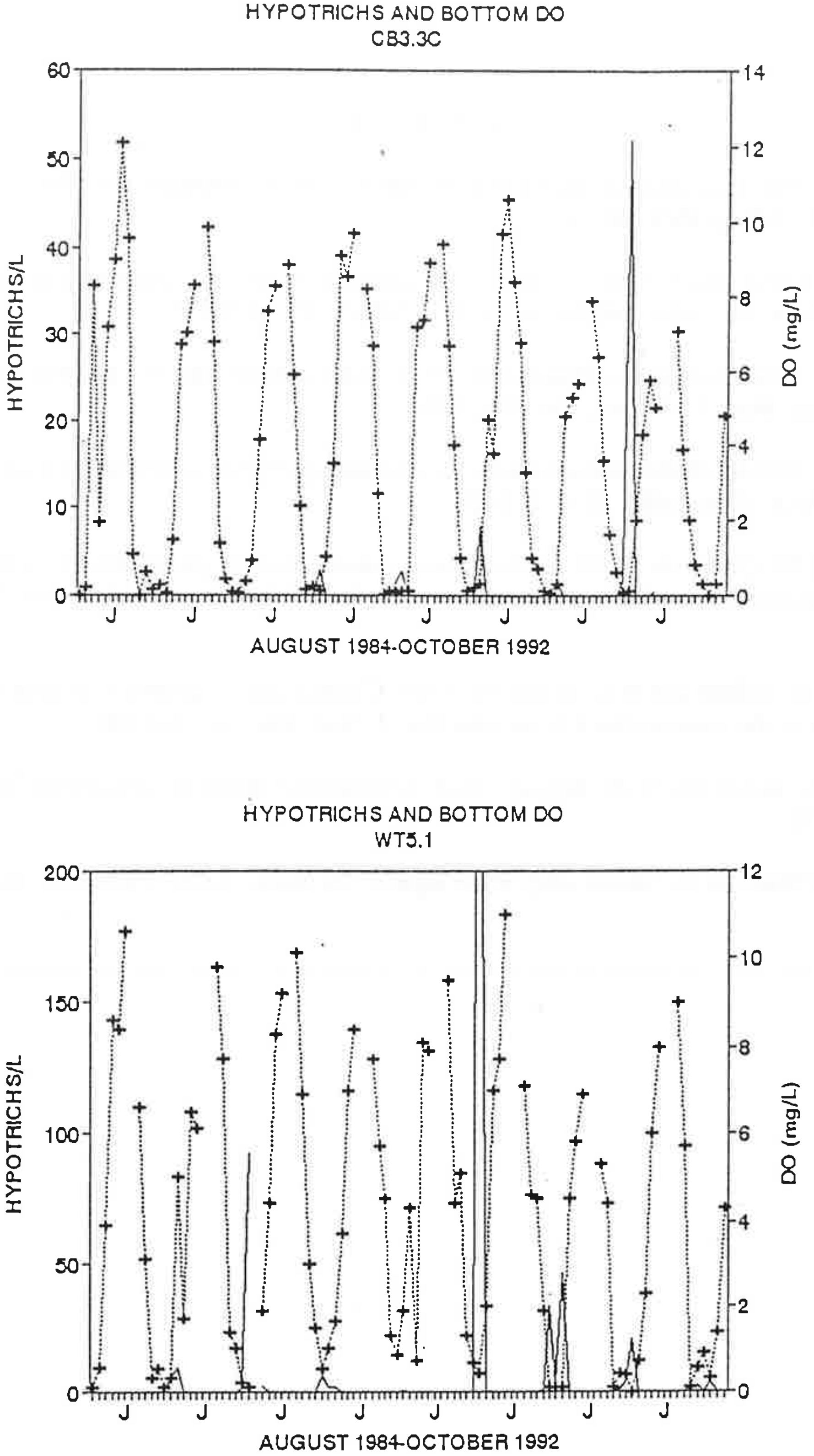
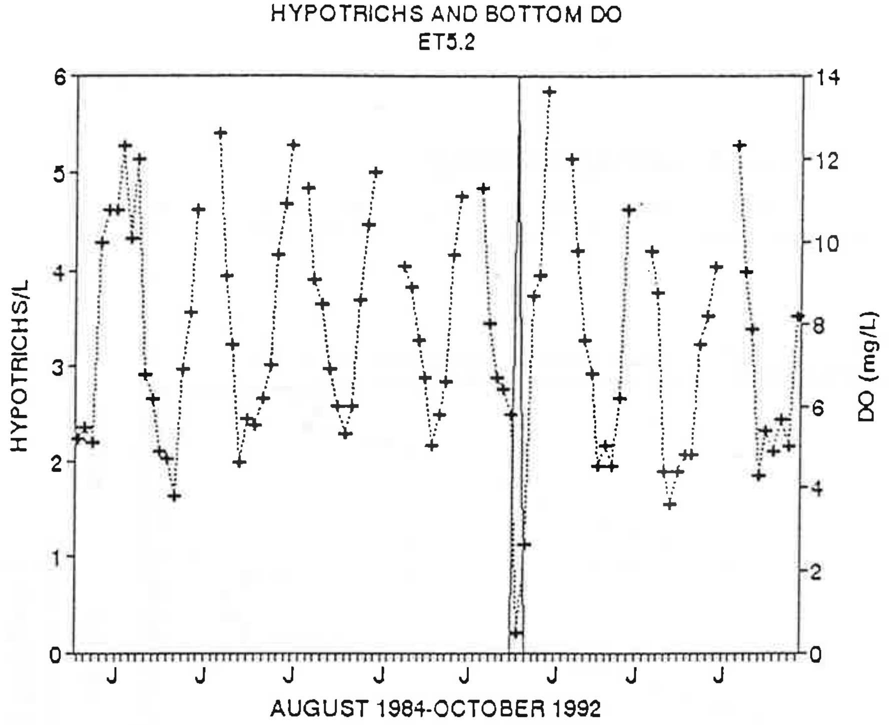


Figure 8. Hypotrich ciliate abundance in sub-pycnocline waters and bottom dissolved oxygen concentration at station (a) CB3.3C in the deep trough of Chesapeake Bay and (b) the mouth of the Patapsco River, station WT5.1.



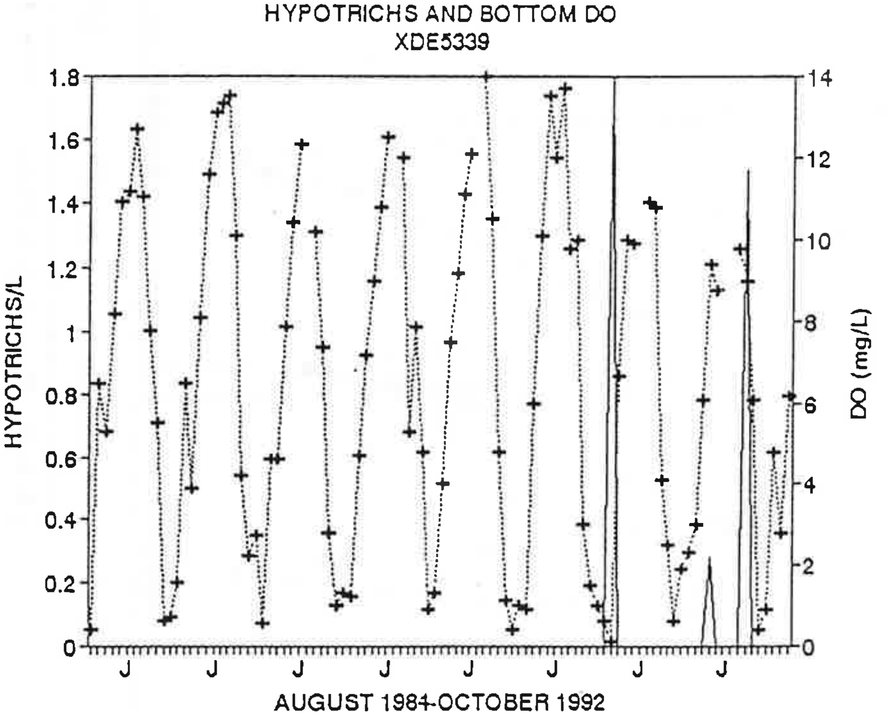
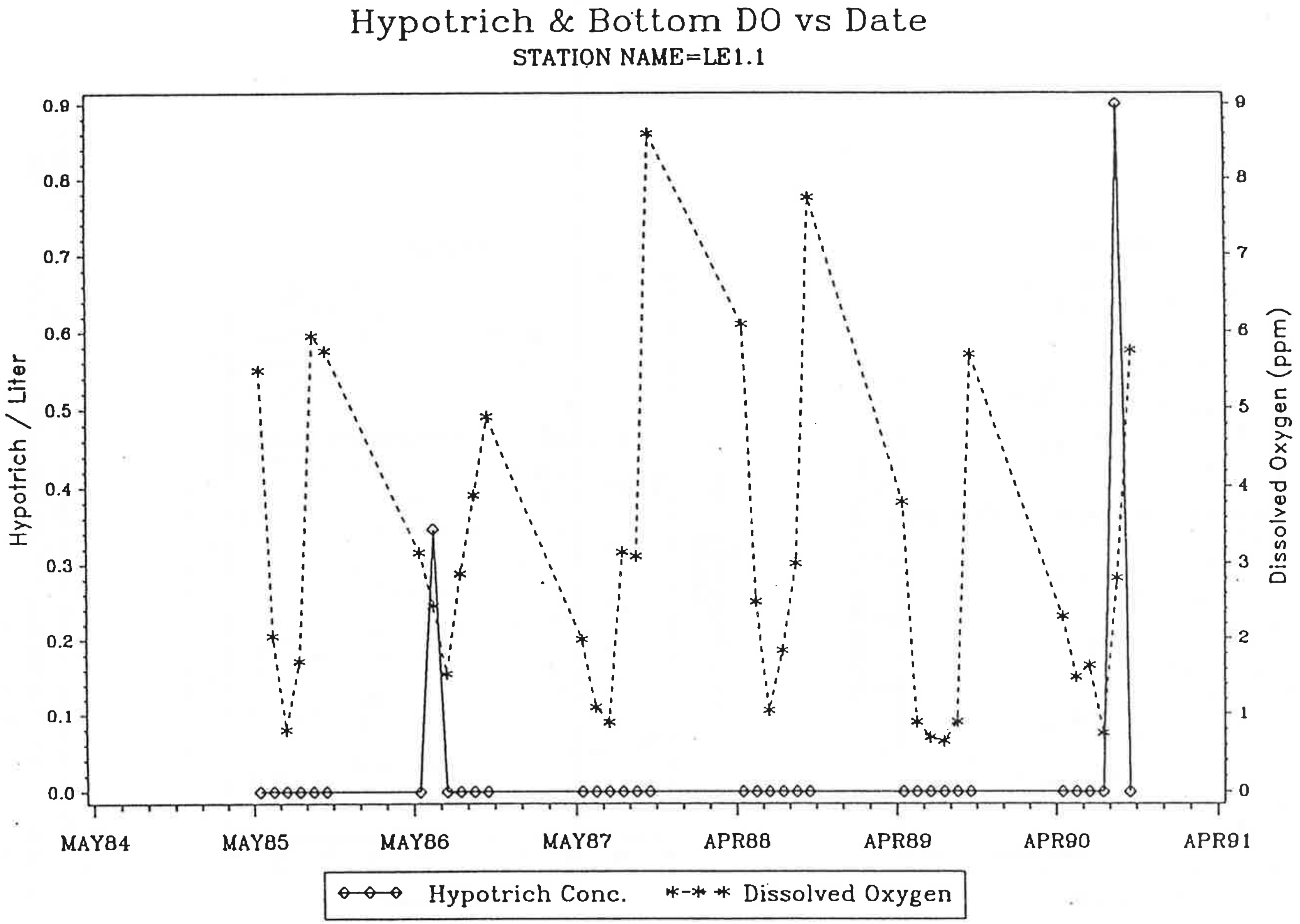


Figure 9. Hypotrich ciliate abundance in sub-pycnocline waters and bottom dissolved oxygen concentration at (a) station ET5.2 in the mouth of the Choptank River estuary and (b) station XDE5339 in the lower Patuxent River estuary.



Hypotrich & Bottom DO vs Date

STATION NAME=LE1.1

Figure 10.Mean hypotrich abundance over the water column versus bottom DO concentration for station ET5.2.

Zooplankton as Bioindicators of Toxic Substances

by Ray Birdsong

An abundant literature exists (reviewed by Bradley and Roberts, 1987) that shows estuarine zooplankton to be highly sensitive to a variety of toxic substances frequently found in estuarine waters. These include heavy metals (especially mercury, copper, silver and cadmium), a variety of pesticides (the antifoulant TBT is highly toxic to *Acartia tonsa* and *Eurytemora affinis*) and oxidants (notably chlorine and bromine and their residual compounds). Increased toxicity to zooplankton due to the synergistic interactions between various toxics and with salinity and temperature is also known (Bradley and Roberts, 1987).

We have little knowledge of the overall impact of toxics on the Bay's ecosystem or the zooplankton population. Because of expense, routine monitoring of toxic substances is not conducted by the Chesapeake Bay Program. Temporally or spatially limited introductions of toxic substances into the system are likely to produce unexplained anomalies in zooplankton metrics or simply have no detectable impact on the zooplankton community, if the source is far from a monitoring station. However, chronic toxic pollution is at least sometimes detectable by the present monitoring program. A case study is provided by the Elizabeth River, a tributary of Hampton Roads, that is highly industrialized along its course through Norfolk and Portsmouth. Alden, et al. (1988) identified 24 EPA priority pollutants from the Elizabeth River system. Two zooplankton monitoring stations (SBE2 and SBE5) were established in the Southern Branch, the most industrialized subunit, in February 1989. Zooplankton abundance and biomass at SBE2 and SBE5 are chronically depressed (biomass values average 17% and 15%, respectively, of the closest station outside the system, LE5.5 at the mouth of Hampton Roads (see Fig. 4 for comparative abundance values). Sunda et al. (1990) compared Southern Branch water with that from near LE5.5 and found copper to be 3.4-5.3 times and zinc 20-33 time that of LE5.5. When bioassays were conducted using *Acartia tonsa* nauplii the survival was much lower with Elizabeth River water than with Chesapeake Bay water. When the chelators EDTA and NTA were added to the water to detoxify the metals, survival of nauplii significantly improved in the Elizabeth River bioassays. These results strongly implicate copper and zinc as major contributors to the chronic depression of zooplankton in the Elizabeth River system.

Circumstantial evidence suggests zooplankton at the Potomac RET2.3 monitoring station are suppressed by toxics. 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 of RET2.3. Known stressors in the water column at Morgantown and the Dahlgren Naval Weapons Laboratory include tributyltin (TBT), copper, nickel, and possibly mercury and lead, in excess of EPA water quality criteria.

The Elizabeth and Potomac River examples suggest that unmonitored pollutants may be involved in suppressing zooplankton populations at some other stations as well. We recommend that stations with chronically depressed populations for which no alternative causes seem to be present be screened for toxic substances.

xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx

We need to identify these stations after integrating our chapters.

xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx

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Indicators of Zooplankton Linkages with Finfish

Summary

Of the eight Chesapeake Bay striped bass nursery areas presently being monitored for zooplankton, none had "optimal" food levels required for normal striped bass larval growth during the spring. Only one (Choptank) has consistently met "minimum" food level requirements (i.e. median mesozooplankton abundance of 15-25 liter^{-1} during April - June). During the monitoring period, the Potomac declined to "below minimum" food levels and the James declined to "poor" food levels; the Patuxent and Rappahannock were consistently "below minimum" and the York and upper mainstem, "poor".

A significant positive relationship ($P < 0.05$) between mesozooplankton density in the spring and striped bass juvenile index the following summer occurred in the Choptank and Potomac rivers. The relationship for the Potomac is further supported by examination of historical Potomac River zooplankton data sets. The upper bay and the remaining tributaries did not show a similar relationship because zooplankton densities experienced were all relatively low and precluded good regression analyses over a wide range of zooplankton densities.

Significant regressions between mesozooplankton and finfish post-larval planktivore abundances (obtained from the Maryland and Virginia seine surveys) were evident during the summer growing period, indicating the mesozooplankton-planktivore linkage is tightly coupled at this time. The regressions had either shallow or steep slopes if controlling factors such as salinity or flow influenced the zooplankton. Regressions were absent at tidal fresh and oligohaline stations known to be impacted by toxins or increasing nutrient loads. A straightforward relationship between finfish planktivore and mesozooplankton abundances was not found in the mesohaline reaches, apparently because invertebrate planktivores become increasingly important in this salinity regime. Other analytical techniques (e.g. step-wise multiple regression) may be able quantify the portion of the mesozooplankton variability caused by finfish planktivores in the mesohaline.

Regressions between summer abundances of mesozooplankton and finfish planktivores showed strikingly different slopes in the tidal fresh and oligohaline reaches of the bay. Planktivorous fish in tidal fresh waters exert strong top-down controls on their prey, as evidenced by *inverse* regression slopes, and need to be considered one of the factors governing mesozooplankton abundance and community structure in these areas. *Positive* regression slopes in the oligohaline manifest strong bottom-up controls by prey on their predators and, in view of the predominance of planktivorous Y-O-Y and planktivore species, indicate mesozooplankton need to be considered one of the factors governing summer fish populations there. The fundamental change in zooplankton - fish linkage at the fresh/oligohaline interface appears to be the result of major shifts in prey vulnerability, predator-prey overlap, and predator efficiency brought about by environmental changes.

The inverse relationship in tidal fresh waters echoes a pattern found repeatedly in freshwater lakes. Food web management strategies developed for freshwater lakes that incorporate principles of the "trophic cascade effect" can probably be applied directly to tidal fresh reaches in the bay. The *positive* regression slopes in the oligohaline, and the apparently complex relationship between the mesozooplankton and a diverse array of vertebrate and invertebrate predators in the mesohaline, suggests that food web management strategies developed for freshwater lakes may *not* be directly transferable to oligohaline and mesohaline waters in the bay. However, these salinity regimes would probably benefit from a larger influx of *mesozooplankton* abundances, as would fish larvae inhabiting

the tidal fresh. Trophic relationships in all of the salinity regimes need to be further explored with the array of zooplankton indicators that are at hand. Zooplankton indicators should be compared with the various trawl surveys, also. These future investigations may confirm suspected zooplankton - fish linkages at some stations (e.g. the mesohaline stations) and identify environmental variables that are disrupting the linkages at other stations.

Food Limitation of Striped Bass Larvae in Spring

by Fred Jacobs

Introduction

A number of studies have demonstrated that the availability of adequate zooplankton abundances during critical feeding periods of larval fish is important in determining their survival, growth, and/or year class success (e.g. Cushing 1972; Houde 1978; Hunter 1981; Kernehen et al. 1981). Recently, Tsai (1991) concluded that for striped bass larvae, the higher the prey density, the greater the rate of survival and growth, and ultimately recruitment. Chesney (1989), in a study that examined the relationship of food and other variables on larval striped bass, indicated that while striped bass larvae are well adapted for survival in highly turbid, turbulent systems, low densities of zooplankton are likely to greatly diminish the probability of larval survival.

The importance of mesozooplankton in the larval striped bass diet was underscored in a feeding study by Hjorth (1988). Adults and copepodids of the copepod *Eurytemora affinis* were the prey most consistently selected, constituting 60% or greater of the larval fish diet at eleven of fourteen stations. The selection of *E. affinis* was striking because the taxon comprised less than 10% of the total zooplankton numbers. Tsai (1991) indicated optimum growth and survival of striped bass larvae when fed *E. affinis* adults, with poorer results for copepodids and nauplii. He concluded that one *E. affinis* adult was equivalent to 1.5 copepodids or 11.1 nauplii, in terms of nutritive value.

Microzooplankton such as nauplii and rotifers appeared to be more important in the diets of species such as white perch as compared to those of striped bass larvae (Hjorth 1988). While early stage striped bass and white perch both prefer *E. affinis* adults, their response to a lack of that taxon being present appears to be different. Striped bass curtailed feedings, while white perch switched to smaller prey such as nauplii and rotifers. These findings are somewhat in contrast with Tacacs (1992) who found rotifers and nauplii to be important in the first feeding stages of both striped bass and white perch. In any case, it is clear that following first feeding, mesozooplankton become increasingly important in the diet of larval fish, and particularly for striped bass.

Starvation experiments have generally indicated that striped bass larvae can withstand relatively long periods without food (Eldridge et al 1981; Rogers and Westin; 1981). They are able to utilize energy stored in an oil globule which is prominent in striped bass larvae but not in other taxa such as white perch. White perch larvae consequently cannot withstand as long a low food period as striped bass. White perch can somewhat overcome this lack of energy reserves by feeding on microzooplankton which are plentiful in the bay systems.

Short-term lack of food, while not starving striped bass larvae, may still negatively impact survival. Starved fish do not grow. As vulnerability to predation and disease generally decreases with size, it is important for the larvae to spend a minimal amount of time at its most vulnerable size. Therefore, exposure to adequate zooplankton concentrations would affect larval fish survival either directly or indirectly, as rapid growth of larvae will enhance the probability of survival. The Chesapeake Bay Zooplankton Monitoring workshop report (Buchanan 1992) indicated that high spring mesozooplankton abundances may be associated with high recruitment of finfish in the summer, as reflected by the juvenile striped bass index. For the Maryland tributaries and the upper mainstem of the Bay, the relationship approached a significant association between the variables but was skewed by certain pairs of points. The relationship is examined in greater detail in this report.

Methods

As described in the Chesapeake Bay Zooplankton Monitoring workshop report (Buchanan, 1992), there appears to be a quantifiable relationship between zooplankton densities observed in the field and the abundance of striped bass or white perch larvae. An analysis of three fish larvae databases obtained from J. Uphoff (MD DNR), E. Houde and E. Rutherford (University of Maryland) and R.C. Jones and D. Kelso (George Mason University) indicated that when high densities of striped bass or white perch were observed, high densities of mesozooplankton and/or microzooplankton were also often encountered.

Based on the studies outlined above and others, it was thought that the development of a food availability index may be a useful bioindicator of the critical feeding densities required for striped bass survival in the spring. A number of sources (e.g. Beaven and Mihursky 1980; Setzler-Hamilton et al. 1981; Chesney 1989; Tsai 1991; Eldridge et al. 1981; Miller 1978; Uphoff 1989; etc.) were reviewed to develop subjective categories of feeding ranges for striped bass larvae. While clearly recognizing that environmental factors such as temperature, flow, turbidity, light penetration, etc. greatly affect larval feeding rates, the body of the literature was sufficiently consistent to suggest that distinct feeding categories could be created. These ranges while admittedly not perfect can provide valuable guidance for resources managers in determining the status of fish resource food availability during critical larval first-feeding periods of striped bass.

Only the mesozooplankton fraction of the zooplankton was considered in the following analyses because of the striped bass larvae preference for mesozooplankton. The "optimal" category was a density of greater than 25 organisms per liter. A "minimum" requirement of zooplankton fell between 15 and 25 liter^{-1}; "below minimum" ranged from 5 to 15 liter^{-1}; and "poor" was less than 5 liter^{-1}. If the microzooplankton fraction had been considered, the values for each category would have been considerably higher because of the presently high proportion of microzooplankton in the zooplankton community.

The index was applied to the Chesapeake Bay Monitoring Program mesozooplankton data for the spring (April - June) of each year for which data were available. A median density for each year was calculated for individual tributaries and the upper mainstem using the monthly mesozooplankton densities at monitoring stations overlapping striped bass early life stage nursery areas (Table 2). Median density was thought to be more representative than the mean because the data were often skewed, with a few high and many low values. It was felt that median value represented a truer density of what food larvae would experience most of the time.

A separate slopes model was used to further examine the potential relationship between spring mesozooplankton abundance and summer juvenile striped bass index, by river system. Log-transformed mean zooplankton densities for the April - June period and striped bass juvenile index were the main variables.

Results and Discussion

Food availability for striped bass larvae in spring. The Choptank River was the only tributary that consistently met the "minimum" striped bass larval food requirement over the 1985 - 1991 period. In

1988 and 1990, zooplankton densities at the two zooplankton monitoring stations in the Choptank spawning area were within the "optimal" range, and were at "minimum" levels in 1985, 1986, and 1991 (Figure 11). Only in 1987 and 1989 were the striped bass food availability indices "below minimum". The food availability index appears to have declined in the Potomac and James rivers in recent years. After being at "optimal" levels in 1986, the index fell in the Potomac (Figure 12), reflecting a substantial decline in the zooplankton populations at the oligohaline station RET2.3. Similarly in the James River, relatively high index values early in the study period were followed by years characterized as "poor" (Figure 13). The annual median value for the two Patuxent stations was consistently "below minimum" (Figure 14). Zooplankton abundances at the upper Patuxent station (PXT0402) were often in the "minimum" range, however those at the lower station (XED4892) were typically "poor" and consequently brought the annual median value down. Annual indices in the Rappahannock and York fluctuated between "below minimum" or "poor" (Figures 15 and 16). The upper bay stations were consistently "poor" (Figure 17).

Figure 18 illustrates the overall food availability index for striped bass larvae for each spawning area in which zooplankton are monitored, for the entire monitoring period. Overall index values were obtained by calulating the average of the median spring values for each tributary and the upper Bay. These findings suggest that although zooplankton densities in the spawning areas may sometimes climb to optimal levels, *overall* densities of zooplankton are sub-optimal for normal striped bass larval growth in all spawning reaches except the Choptank. Declining index values in the Potomac and James rivers have resulted in "below minimum" and "poor" averages, respectively, in the second half of the study period. Overall, the Patuxent and Rappahannock Rivers were "below minimum" while spawning areas in the upper mainstem and York (i.e. Pamunkey) were "poor".

While the upper bay fell into the "poor" category, one of its two stations (MCB1.1) may not truly reflect zooplankton densities of the region as it is located in the mouth of the Susquehanna River. The station is probably greatly influenced by Susquehanna River flow and the resulting short residence time. Hence, zooplankton abundances are depressed at this station. In an attempt to eliminate this shortcoming, an additional more representative station in the Upper Bay has been added to the monitoring program.

Association between spring mesozooplankton abundance and summer striped bass juvenile index. A separate slopes model examining variability by system (i.e. Potomac, Patuxent, Choptank and upper bay) indicated a significant positive relationship ($P<0.05$) between mesozooplankton density and striped bass juvenile index occurred in the Choptank when the 1989 data point was removed, and in the Potomac (Figure 19). (The Choptank 1989 data was deleted because one seine haul yielded over 1000 fish and could have been considered an outlier.) Forty eight percent of the variability in the entire model (i.e. $r^2 = .48$) was explained by the relationship between the variables. The relationship for the Potomac is further supported by examination of historical Potomac River zooplankton data sets from other sources, namely the Chesapeake Biological Laboratory (Setzler-Hamilton) data for 1980, 1981, 1982, 1985, and 1986, and the Douglas Point Power Plant Research Project data for 1974. Although these studies used a 72 um net and zooplankton densities were thus considerably greater than obtained in the monitoring program data (1985-1991) where a 202 um net was used, the results could be compared to the striped bass juvenile index separately (Figure 20). It is interesting to note that the highest juvenile index values of 10 in 1982, and 9.9 in 1986, corresponded to extremely high zooplankton densities of 298 and 247 zooplankton liter^{-1}.

Other Maryland river systems did not show such a clear relationship between zooplankton and the juvenile index. The Patuxent River juvenile index values were generally low, regardless of zooplankton densities. The upper bay, a spawning reach with only low mesozooplankton densities in the spring which had a food availability index of "poor" (above), showed no relationship (Figure 21). The range of mesozooplankton densities here is too small to show the relationship.

Mesozooplankton spring densities in the York and Rappahannock tributaries were generally low, in the range of 15 liter^{-1} or less, thereby precluding good analyses for a relationship between zooplankton density and striped bass juvenile index. The paucity of high spring mesozooplankton densities means that regressions of the data will be driven by isolated high values, if they occur, and are probably incorrect. The James River did not showed high juvenile index value (Figure 22) despite a few years of relatively high mesozooplankton abundances. The Rappahannock and York never had a year with high mesozooplankton abundances (Figure 23 and 24).

A stepwise regression was conducted to examine the relationship between mesozooplankton, microzooplankton, flow variables and the striped bass juvenile index in Maryland where microzooplankton data is available. Flow values computed from tributary and main bay locations were weighted for a given 25-year average flow. This weighing allowed for meaningful comparisons among different river systems with very different flow characteristics. Results of the stepwise regression conducted on data collected from 1985-1990 indicated a significant r^2 of 0.36 between total spring flow (March, April and May) and the striped bass juvenile index. No other single variable met the significance criteria for the model ($P<0.15$). Furthermore, when the 1989 Choptank and upper bay data were deleted from the analysis, mesozooplankton density was the single variable that met the significance level for the model (i.e. mesozooplankton density was significantly related to juvenile index).

xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx
See Pauline's comments re this section
xxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxxx

Clearly, more research is needed in this area. Uphoff (1989) indicated that year class strength was related to rainfall and temperature. Similarly, efforts on The Hudson River have suggested that environmental variables such as temperature and flow can affect development and survival of striped bass eggs and larvae (Boreman 1983; Klauda et al. 1980). Because year class strength is relatively far removed from spring zooplankton, perhaps larval survival would be a better variable for comparison with mesozooplankton density or biomass. However, the calculation of larval survival would have to be calculated from either intensive sampling efforts (e.g. weekly), or by applying mortality rates obtained from literature values. While such work is currently under study, the relationship between mesozooplankton, flow, temperature, and striped bass larval survival is a subject worthy of future expanded research efforts.

Recommendations

The food availability index should be expanded to reflect microzooplankton densities. While perhaps not important for striped bass beyond first feeding, microzooplankton densities are likely to be important for developing food availability indices for larvae of other species such as white perch. It is also recommended that food availability indices be considered for forage species that feed almost entirely on zooplankton, such as bay anchovy and silversides.

Efforts to analyze associations between spring zooplankton and the striped bass juvenile index should also be broadened. Stepwise regressions that lag information on flow, rainfall, nutrients, etc. against the variables should be considered as should examinations of zooplankton to ichthyoplankton densities.

TABLE 2. STATIONS UTILIZED IN ZOOPLANKTON FOOD AVAILABILITY ANALYSIS

River	Stations
Upper Bay	CB 1.1 (MCB 1.1), CB 2.2 (MCB 2.2)
Choptank	ET 5.1 (MET 5.1)
Potomac	TF 2.3 (XEA 6596), RET 2.2 (XDA 1177)
Patuxent	TF 1.5 (PXT 0402), TF 1.7 (XED 4892)
Rappahannock	TF 3.3 (1R)
York	TF 4.2 (1Y)
James	TF 5.5 (1J)

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List of Figures

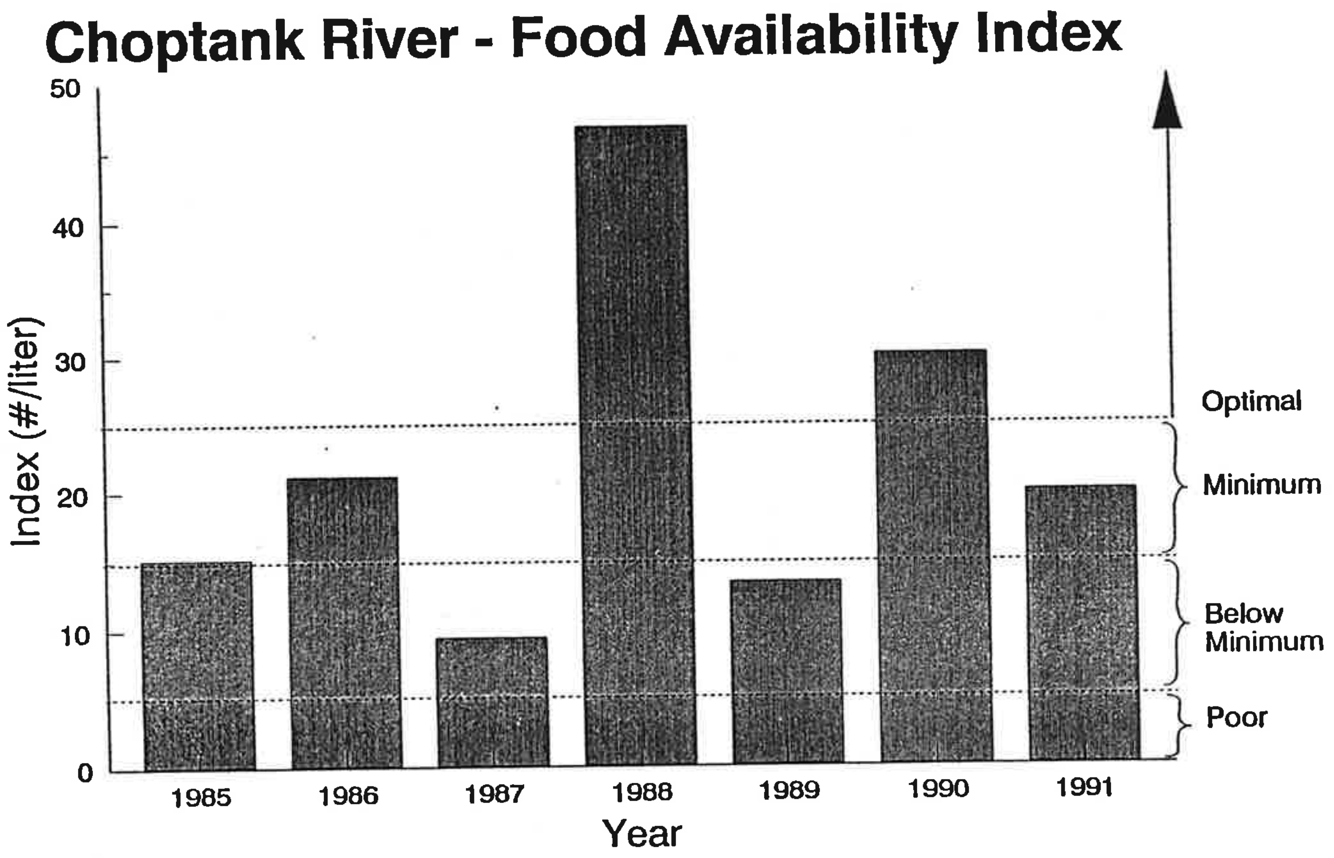

Choptank River - Food Availability Index
Index (#/liter)
50
40
30
20
10
0
1985
1986
1987
1988
1989
1990
1991
Year
Optimal
Minimum
Below Minimum
Poor

Figure 11.

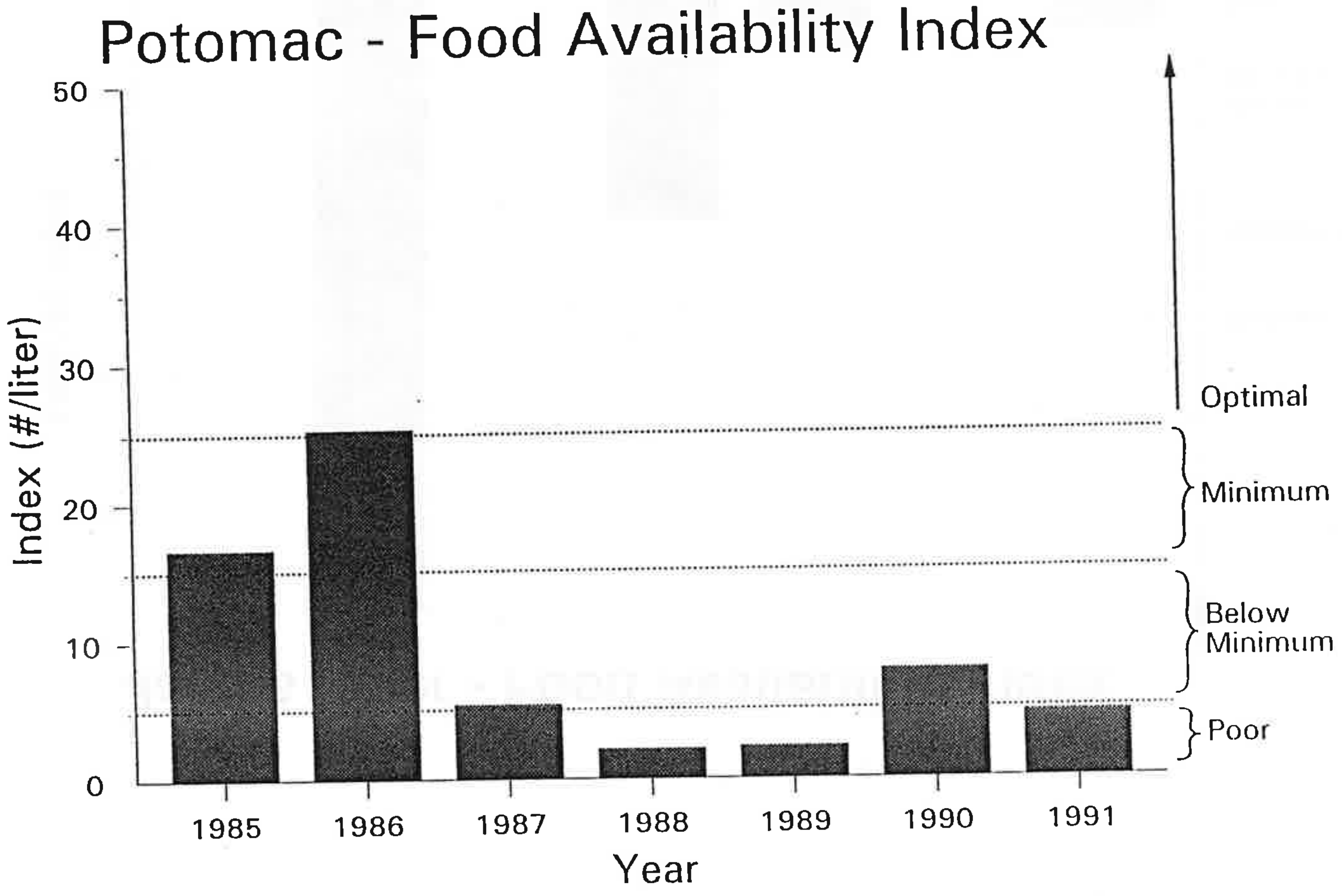

Potomac - Food Availability Index
Index (#/liter)
50
40
30
20
10
0
1985
1986
1987
1988
1989
1990
1991
Year
Optimal
Minimum
Below Minimum
Poor

Figure 12.

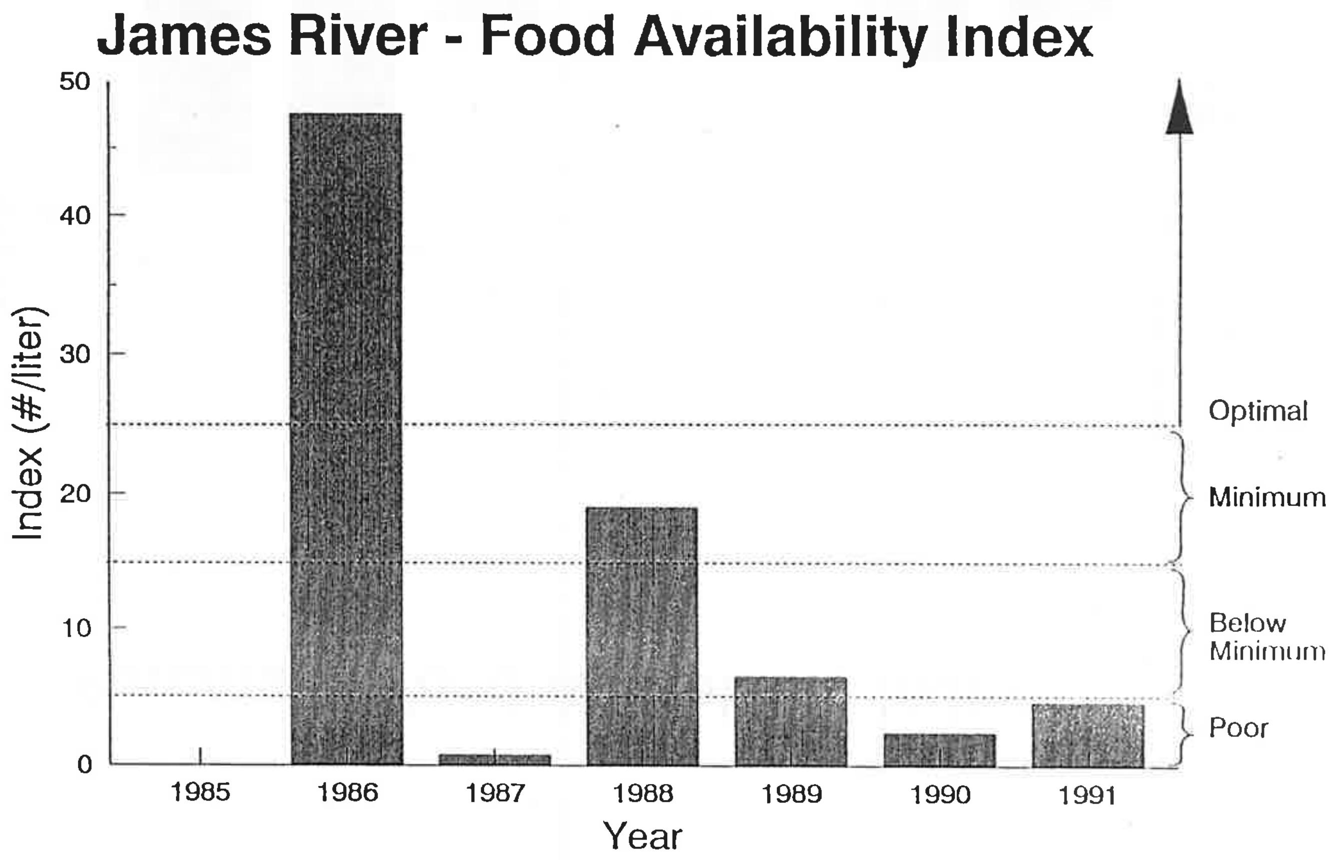
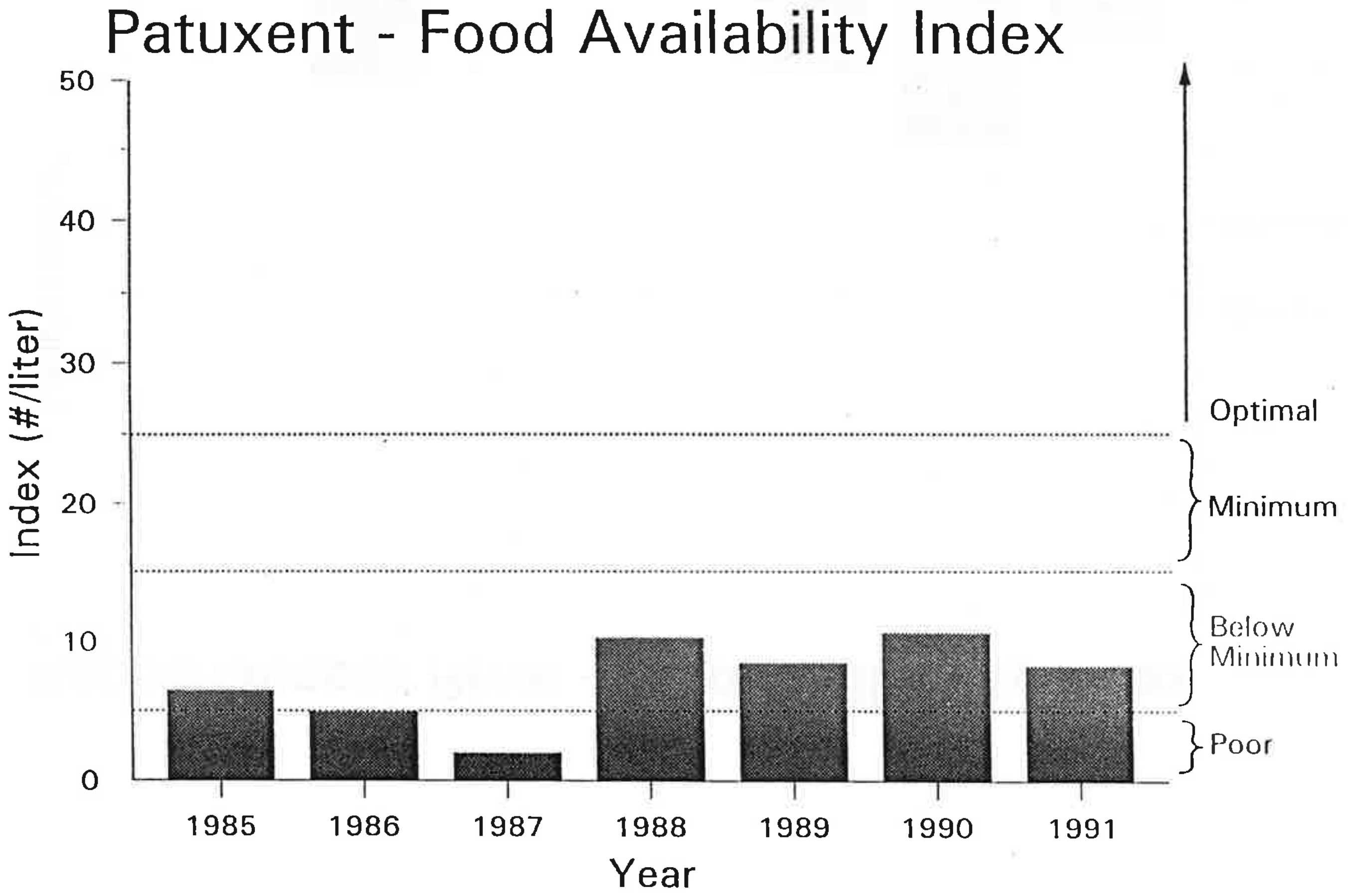

James River - Food Availability Index
Index (#/liter)
50
40
30
20
10
0
1985
1986
1987
1988
1989
1990
1991
Year
Optimal
Minimum
Below Minimum
Poor

Figure 13.



Patuxent - Food Availability Index

Figure 14.

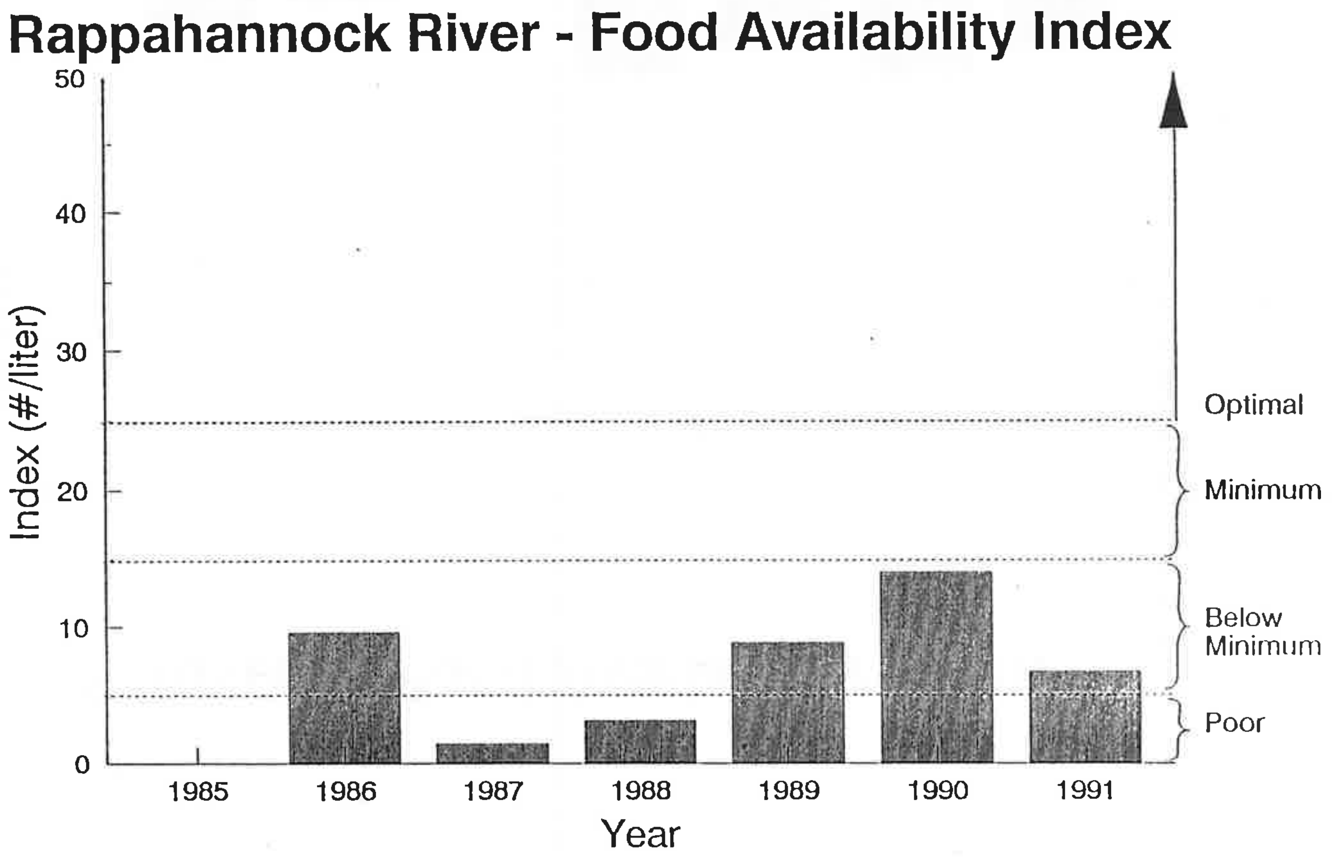

Rappahannock River - Food Availability Index
Index (#/liter)
50
40
30
20
10
0
1985
1986
1987
1988
1989
1990
1991
Year
Optimal
Minimum
Below
Minimum
Poor

Figure 15.

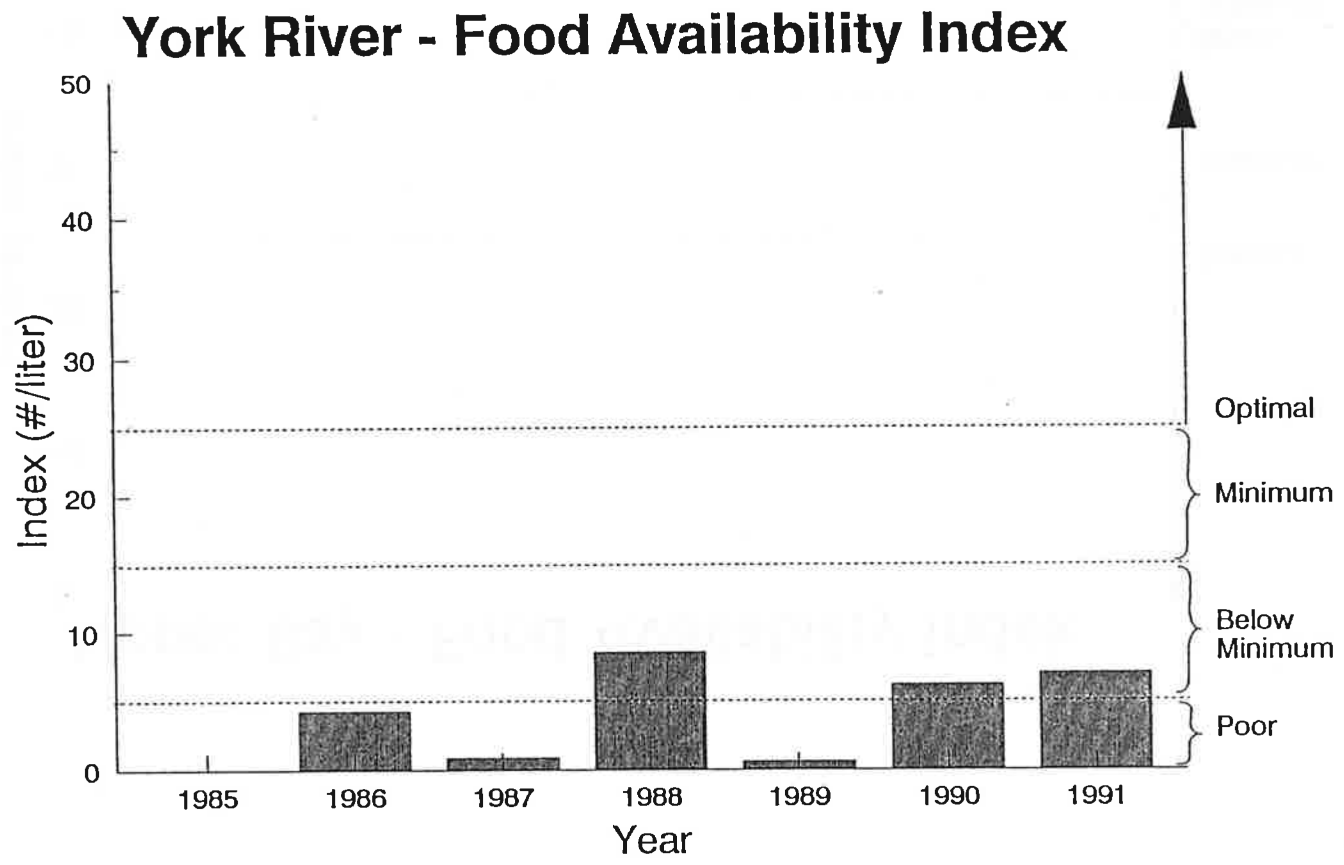

York River - Food Availability Index
Index (#/liter)
50
40
30
20
10
0
1985
1986
1987
1988
1989
1990
1991
Year
Optimal
Minimum
Below Minimum
Poor

Figure 16.

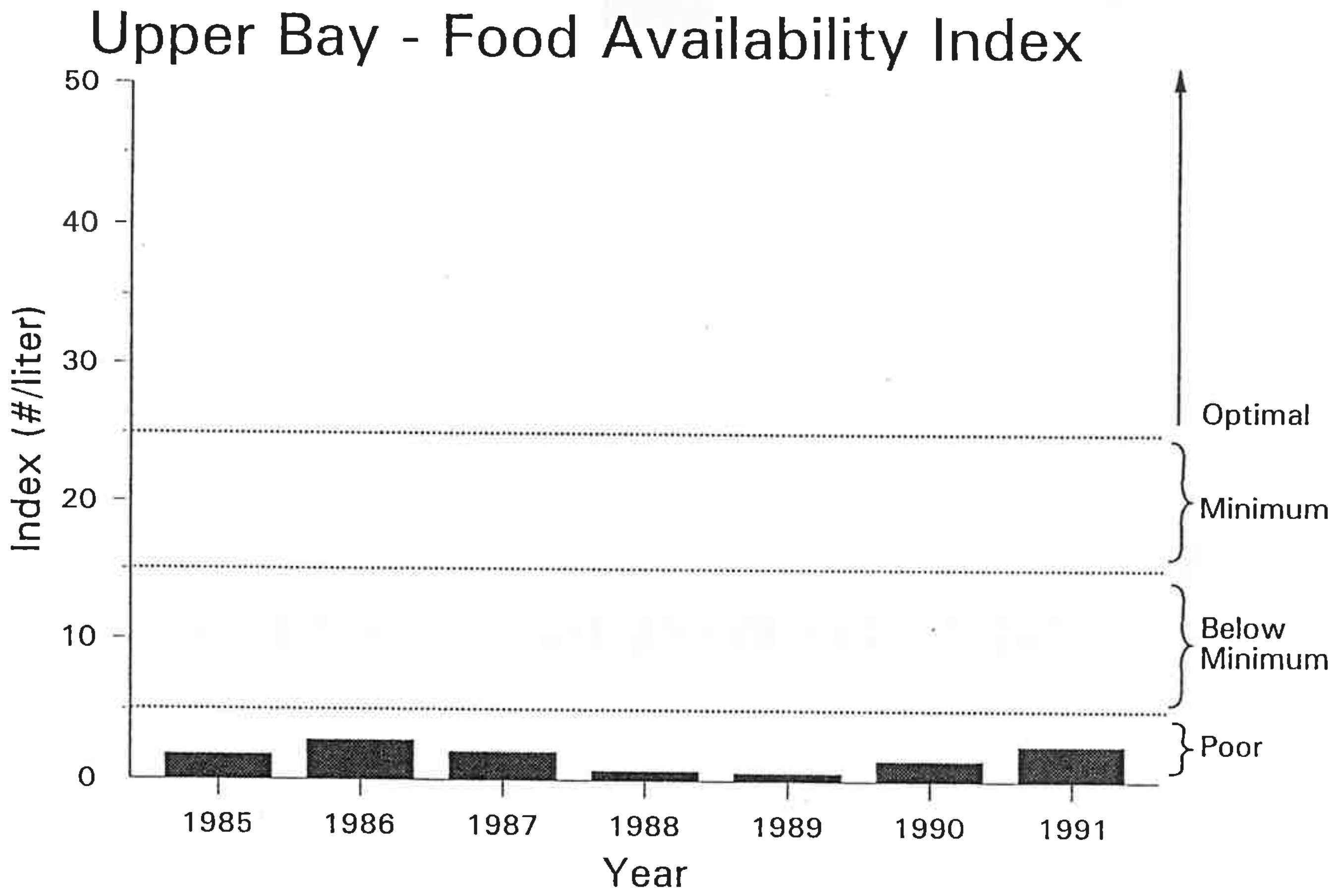
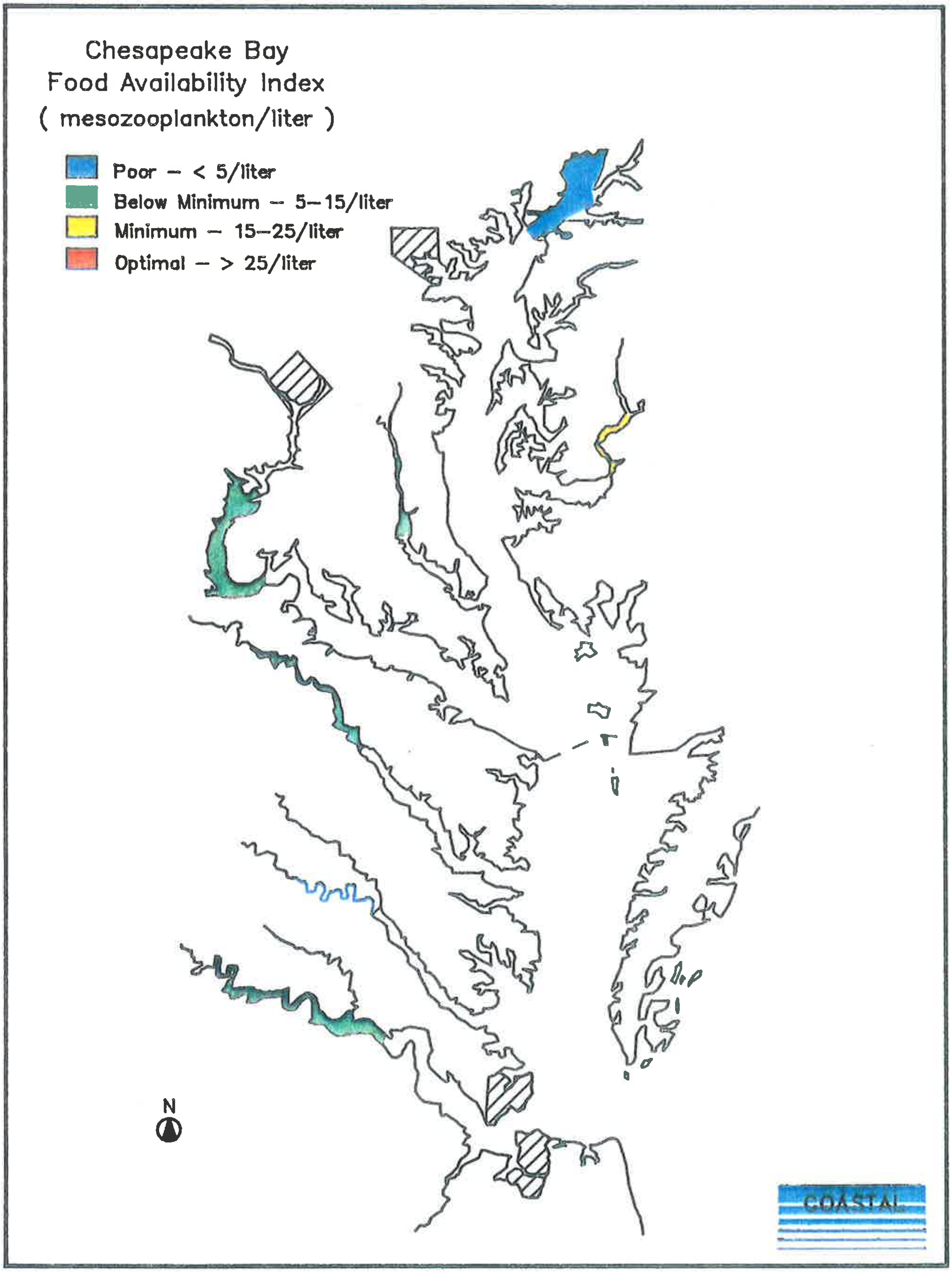

Upper Bay - Food Availability Index
Index (#/liter)
50
40
30
20
10
0
1985
1986
1987
1988
1989
1990
1991
Year
Optimal
Minimum
Below Minimum
Poor

Figure 17.


Chesapeake Bay
Food Availability Index
(mesozooplankton/liter)
Poor – < 5/liter
Below Minimum – 5–15/liter
Minimum – 15–25/liter
Optimal – > 25/liter
N

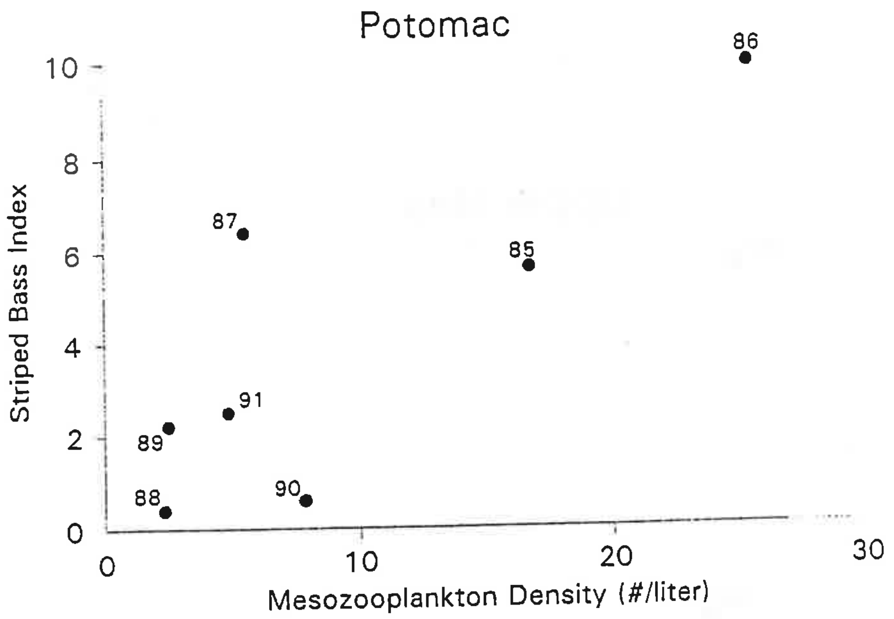


Figure 19

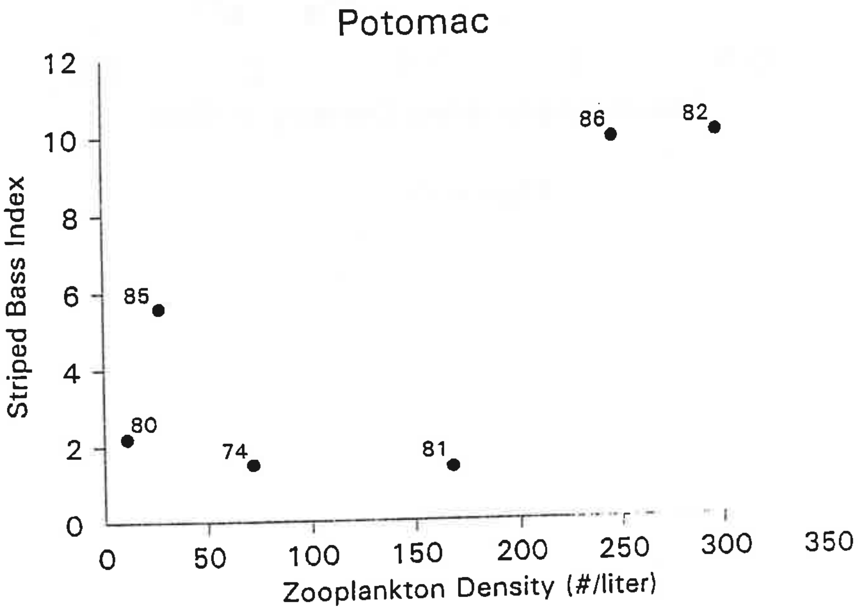


Figure 20

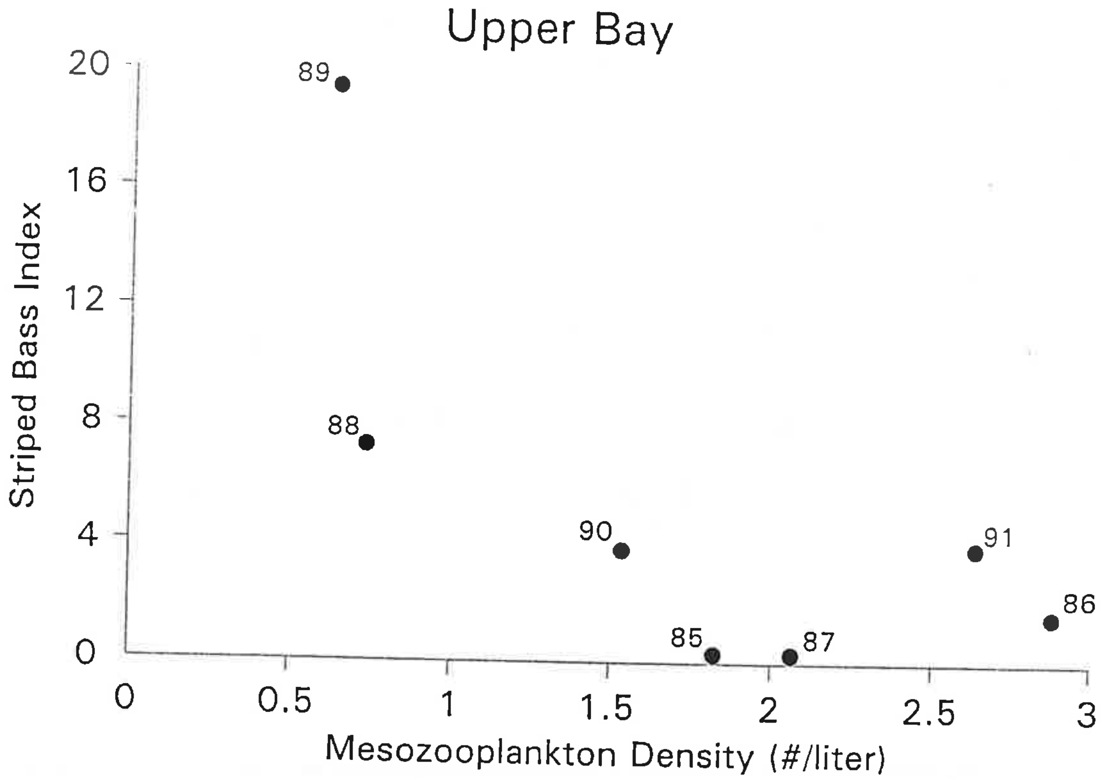

Upper Bay
Striped Bass Index
20
16
12
8
4
0
89
88
90
91
86
85
87
0
0.5
1
1.5
2
2.5
3
Mesozooplankton Density (#/liter)

Figure 21.

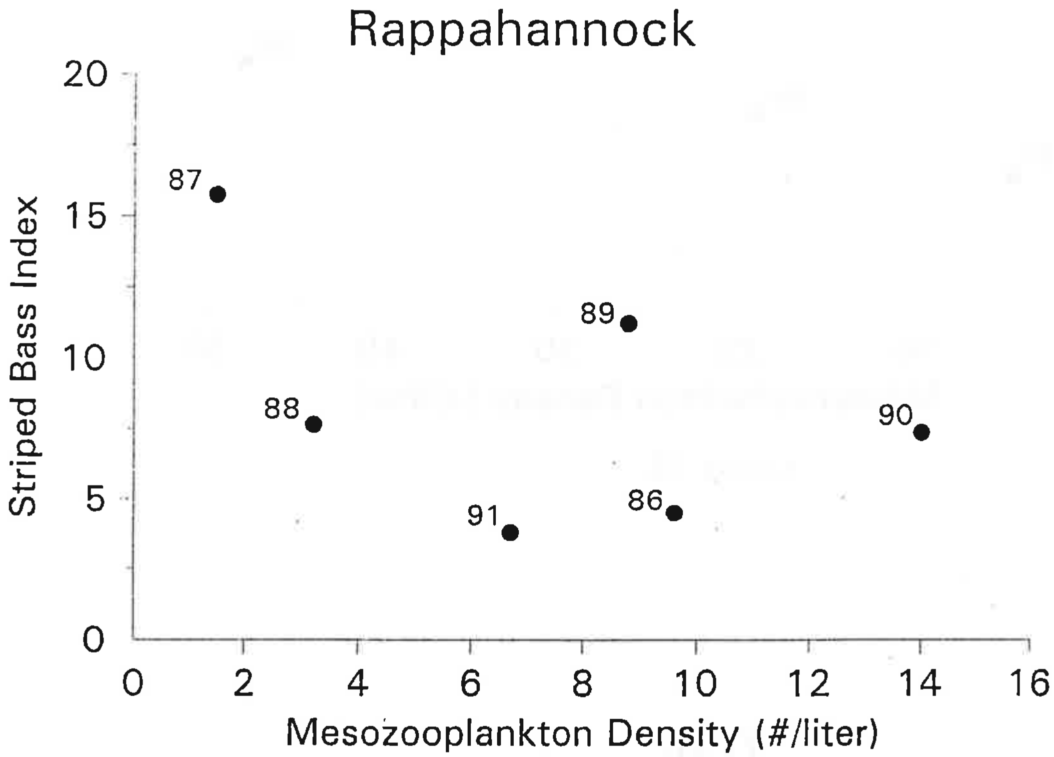

Rappahannock
Striped Bass Index
Mesozooplankton Density (#/liter)
0
5
10
15
20
0
2
4
6
8
10
12
14
16
87
88
89
90
91
86

Figure 22.

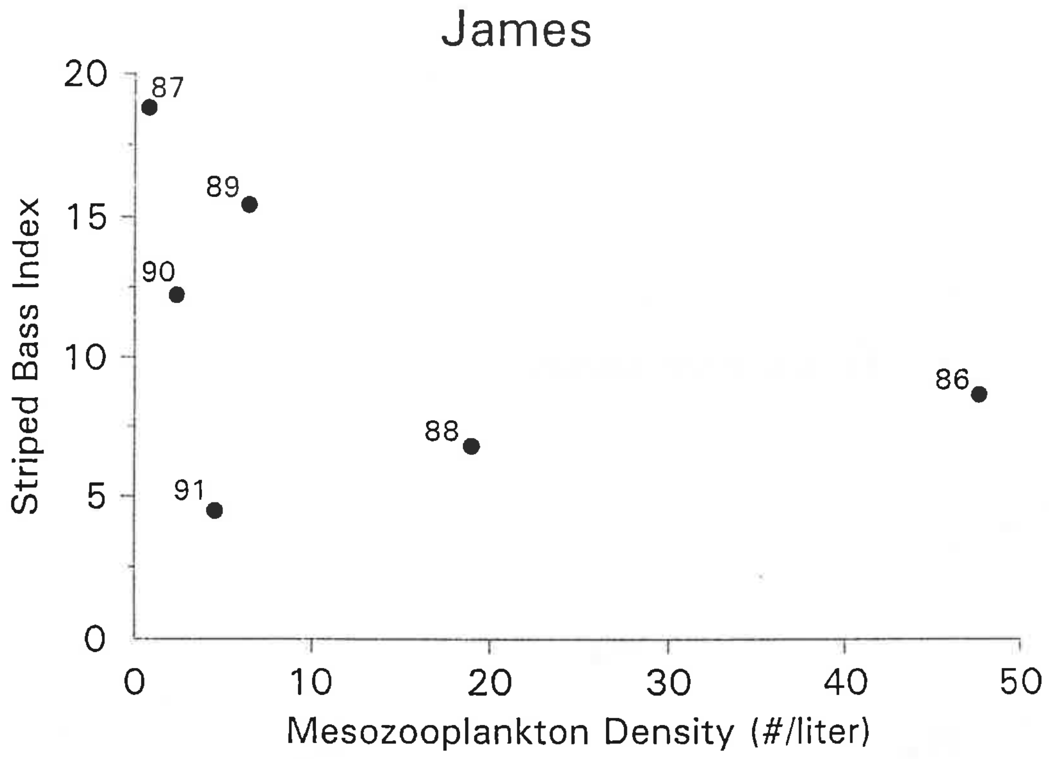


Figure 23.

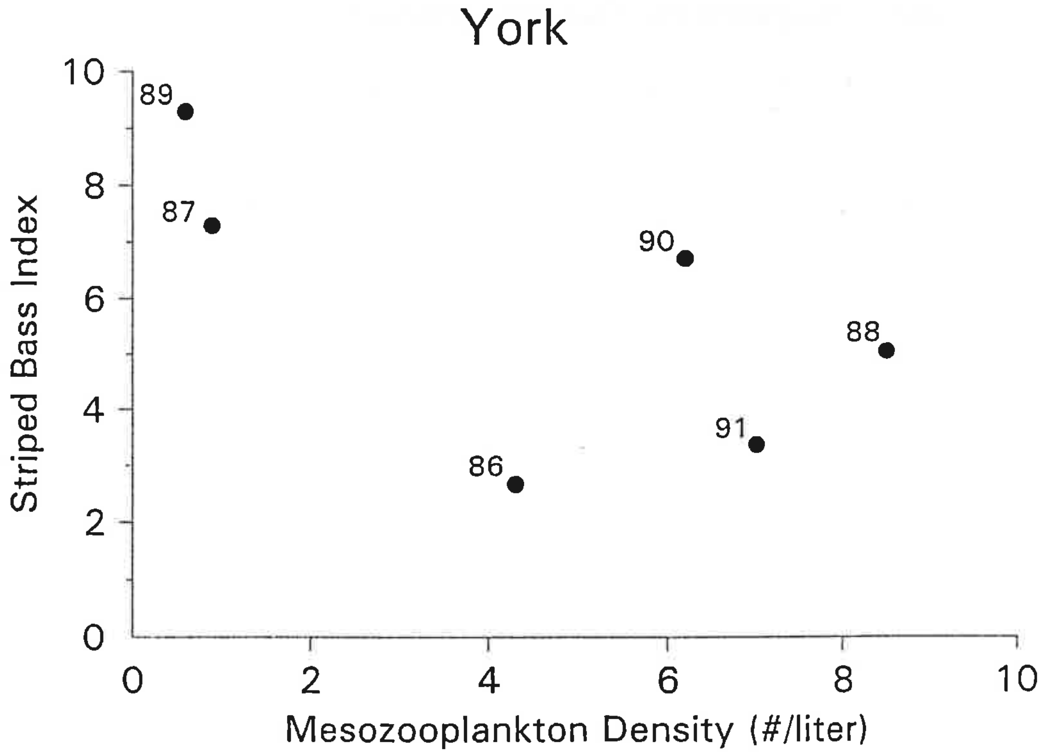


Figure 24.

Preliminary investigations of associations between zooplankton monitoring data and the Maryland and Virginia summer seine surveys

by Claire Buchanan and Pauline Vaas

Introduction

The species presently dominating finfish communities in Chesapeake Bay feed principally upon zooplankton during their juvenile and adult stages. Bay anchovy and Atlantic silversides, currently the most common resident species (Carmichael et al. 1992), are strict planktivores their entire lives. Atlantic menhaden, the dominant species, consume zooplankton during early life stages in coastal waters, then develop specialized brachial structures after entering the estuary which allow them to filter phytoplankton and detritus as well as zooplankton. American shad and river herring, which were historically abundant and are presently overexploited and/or habitat impaired, feed principally on zooplankton during their growing periods in the estuary. Facultative predators on zooplankton are also presently abundant in Bay fish communities, e.g. the sunfishes and many of the minnows and killifish. Finally, zooplankton are the obligate prey of most finfish larvae regardless of what prey they may switch to as they metamorphose. For example, Striped bass larvae feed on zooplankton in spring and early summer, become facultative predators of invertebrates as juveniles near the end of their first summer, and are strict piscivores by one year (refs.). The zooplankton role of obligate prey to most larval fish and to the juveniles and adults of many common species in Chesapeake Bay, as well as their role of facultative prey to other common species indicates strong trophic linkages may exist between the Bay zooplankton and finfish communities.

We suspected the zooplankton data from the Chesapeake Bay Monitoring Program could give us empirical, holistic insights of plankton - fish linkages in the Bay when correlated with selected data from state finfish surveys. Furthermore, the observed relationships could be the basis for new indicators of finfish community structure in the Bay. Such indicators have been developed elsewhere for use in fisheries management. Galbraith (1975) used the abundance of *Daphnia* spp. to predict the survival and "fishing quality" of rainbow trout *Salmo gairdneri* in some 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". 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). Zooplankton indicators could be useful to Chesapeake Bay management considering the Bay-wide coverage of the zooplankton monitoring program and the present difficulties in comparing the various state finfish surveys. Tributary differences in water quality, hydrology and salinity could be expected to help sort out the influences of other controlling factors and clarify the environmental limits within which strong trophic linkages are possible.

For this project, we focused on the interactions between finfish species which feed predominantly on plankton their entire lives (strict planktivores), and their prey the mesozooplankton. We explored two hypotheses:

1) significant regressions between mesozooplankton and finfish post-larval planktivore abundances will be evident during the summer growing period (indicating the mesozooplankton - planktivore linkage is tightly coupled at that time);

2) the regressions will have either shallow or steep slopes, or be weak or non-significant if other controlling factors (eg. salinity, water quality, high flow, predation by another group) strongly influence the mesozooplankton, finfish planktivores, or both.

We assumed trophic linkages between fish and zooplankton would be strong, and therefore most evident, between planktivores which are essentially specialized predators, and their prey species which comprise most of the mesozooplankton. We looked for evidence of trophic linkages in July, August and September because planktivory is most intense at this time, when planktivore species are actively feeding, growth rates are at their annual maxima and Y-O-Y, most of which are to some extent planktivorous, contribute substantially to the overall predation pressure on mesozooplankton.

Methods

Only two long-term finfish data sets were available at the time of this project to compare to the zooplankton monitoring data: the Maryland Estuarine Juvenile Finfish Survey and the Virginia Juvenile Striped Bass Survey. Both are shoreline seine surveys done in bay tributaries and the upper bay, ranging from tidal freshwater to mesohaline reaches and focused on the spawning and nursery grounds of commercially important anadromous fish. Seine hauls are done at the numerous stations in the months of 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 Monitoring Atlas (Heasly et al. 1989) and elsewhere. The seine survey data were selectively sorted so as to better match the information to the zooplankton monitoring data, i.e. only seine stations located near a zooplankton monitoring station were used (Table 3). Several tributary zooplankton stations and all of the bay mainstem stations south of CB2.2 could not be matched with seine stations and are therefore not included.

Maryland and Virginia seine survey protocols are different. At each seine station, Maryland collects three rounds of seine hauls, 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 stations were not sampled. To prevent the gaps from biasing finfish estimates, we excluded station-year data if the following criteria were not met:

> *Virginia.* If at least three hauls out of the ten total hauls were missing for a particular year, the station results were not used for that year. Only one station had at least three hauls missing. An additional 21 station-year combinations has two of the ten hauls missing, but these were not deleted.
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> *Maryland.* If at least two hauls out of six total hauls were missing for a particular year, the station results were not used for that year. If a zooplankton monitoring station was paired with only one seine station, all of the data were kept whether or not there were missing hauls. There were nine station-year combinations that were deleted from the Maryland data.

For each year, the fish data were summed for all hauls by station since different species could occur in the hauls over the course of the summer. The sums of the stations grouped together to match a zooplankton monitoring station were then averaged. The sums were not normalized, hence the Virginia data (sum of 10 total hauls) are approximately 40% greater than comparable Maryland data (sum of 6 total hauls).

After sorting by station, we selectively sorted for species that are known to be principally planktivores or strict piscivores. Fish species that facultatively feed on zooplankton after larval first-feeding stages were excluded, as were species that were *not* obligate piscivores after age 1 year or during their time in the estuary. A list of these planktivore and piscivore species was arrived at with the help of Jim Uphoff and Steve Jordan of Maryland Department of Natural Resources and Jim Cummins of the Interstate Commission on the Potomac River Basin (Table 4). This selective sorting was difficult to do for the "strict piscivore after age 1 year" group of the Virginia data because the state does not record ages for juvenile fish. Consequently, we grouped *all* the juveniles of these species (see Table 4) in the piscivore category for Virginia, even though they are often mixotrophic as Y-O-Y, and Virginia counts of "strict piscivores" are slightly high.

Finally, the *average* station abundance of mesozooplankton for July, August and September of each year was regressed against each year's average planktivore abundance for the matching seine station(s). Historical mesozooplankton data were available for the upper Potomac TF2.3 station for 1974 (unpublished data obtained from Versar, Inc. and described in Ecological Analysts, 1974) and 1981 (Buchanan and Schloss, 1983). Only data collected in the vicinity of TF2.3 was averaged for these years.

Results

Recognizing the enormous impact of salinity gradients on zooplankton community structure, we first compared stations that experienced similar salinities during the summer months (July, August, September). Distinct patterns were in fact found in three of the salinity regimes defined by the Venice system, namely 0 - 0.5 ppt (tidal fresh), 0.5 - 5 ppt (oligohaline), and 5 - 18 ppt (mesohaline). The term "planktivore" refers to finfish planktivores in the following discussion, except when noted otherwise.

Tidal fresh (0 - 0.5 ppt salinity). Of the four zooplankton monitoring stations that are entirely in tidal freshwater, TF5.5 (James) and TF4.2 (York) are in two of the Bay's smaller tidal fresh regions whereas TF2.2 (Potomac) and CB1.1 (upper Bay) are in the Bay's largest. The four stations typically experienced low planktivore abundances, but zooplankton abundances were relatively high at one station (Potomac), modest at another (James) and exceptionally low at two (upper Bay, York). Summer zooplankton community structure in the tidal fresh was diverse compared to oligohaline and mesohaline communities, and although the cladoceran *Bosmina* was often the dominant species, it rarely comprised more than 70% of the population.

Both the larger tidal fresh regions showed a significant, inverse relationship between mesozooplankton and planktivore abundance during the summer months, i.e. mesozooplankton abundance decreased as planktivore abundance increased (Figures 25 and 26). The inverse relationship indicates mesozooplankton abundance responds to planktivore abundance or more specificially to planktivore predation pressure. The tidal fresh Potomac station experienced significant water quality and habitat improvements during the 1984 - 1991 period which may be obscuring the mesozooplankton - planktivore relationship somewhat. Nutrient loadings and ambient concentrations decreased after phosphorus removal was implemented upriver at the Blue Plains Sewage Treatment Plant in Washington, D.C. in the early 1980's, and acreage of submerged aquatic vegetation nearly tripled in this segment of the river in the mid 1980's, further improving water quality (Carter et al. 1988) and allowing a resurgence in Largemouth bass, a top predator that relies heavily on SAV for habitat (Fewlass). Concurrently, microzooplankton abundance and biomass have declined, and mesozooplankton community structure has shifted towards fewer Cladocera.[2]

The upper Bay station had considerably lower mesozooplankton densities than the tidal fresh Potomac, however the station is located in the high flow zone in the mouth of the Susquehanna River and although it may accurately track trends in the upper Bay it appears to underestimate the average mesozooplankton densities there (Sellner and Jacobs, personal communication). The upper Bay, also known as the "Susquehanna Flats", had high but fairly unchanging nutrient concentrations between 1984 and 1990 (mainstem report) which appeared to suppress mesozooplankton abundances (discussed elsewhere in this report). The very high correlation coefficient (r^2) suggests the mesozooplankton - planktivore link is tightly coupled here despite the impact of high flow and eutrophication on mesozooplankton abundance.

Neither the James nor the York tidal fresh stations show a significant correlation between mesozooplankton and planktivores (Figures 27 and 28). Mesozooplankton abundances in the tidal fresh James (TF5.5) are approximately half those found in the tidal fresh Potomac (TF2.3). It is interesting to note that both planktivore and mesozooplankton abundance decrease similarly from 1986 to 1991. We suspect that linkages between predator and prey exist to some degree at this station, but some changing environmental variable is frequently resetting the regression slope and intercept of the relationship. One possible stressor may be the significant increase in nutrient loadings at the James River fall line since the Chesapeake Bay Program monitoring began. Future analyses may clarify what is happening here.

The tidal fresh York (TF4.2) had the lowest mesozooplankton abundances of any zooplankton monitoring station in Chesapeake Bay, except the flow-dominated upper Bay CB1.1 station. High flows can reduce mesozooplankton densities in the spring at TF4.2, but flow was apparently not a controlling factor in July, August and September during the 1986-1991 study period. The lack of a relationship between the planktivores and mesozooplankton combined with especially low abundances suggests some factor, or combination of factors, has an impact large enough to disrupt linkages between the two populations here. Further indications that the zooplankton community is stressed at this location include frequent population crashes (zooplankton densities $< 1000\ m^{-3}$). The station was also associated with the lowest planktivore abundances found in Virginia, or bay, seine surveys. Some possible candidates are [...water quality??... suggestions anyone?]

Oligohaline (0.5 - 5 ppt salinity). The relatively diverse zooplankton community of the tidal fresh shifts to an *Acartia* dominated, estuarine community as it enters the oligohaline. Freshwater species decline and are replaced by estuarine species. There was no consistent pattern of change in summer mesozooplankton abundances between tidal fresh and oligohaline stations when both occur in the same system. Overall, summer abundances dropped in the Potomac, rose in the upper bay, and remained approximately the same in the James. Summer mesozooplankton abundance declined somewhat between the fresh/oligohaline and the oligohaline/low mesohaline stations in the Patuxent. In contrast to the mesozooplankton, summer planktivore abundances rose in all the systems, sometimes substantially.

Correlations between mesozooplankton and planktivore abundances for the six predominantly

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[2] Biological data collected by the District of Columbia is presently being computerized and will be available soon to further document the planktivore - mesozooplankton relationship in the tidal fresh Potomac estuary.

oligohaline[3] stations in the Bay and its tributaries suggest an entirely different relationship than that found in the tidal fresh. Oligohaline stations demonstrated a strong *positive* correlation between summer planktivore abundance and mesozooplankton abundance during the study period (Figures 29 - 32) except the oligohaline/low mesohaline Potomac (XDA1177) and the oligohaline/low mesohaline James (RET5.2) (Figures 33 and 34). Specifically, planktivore abundance was high when mesozooplankton abundance was high. If we postulate the mesozooplankton and planktivore trophic levels are indeed coupled here, and a third factor is not causing the two to vary in a similar fashion, the positive slopes of the regressions then indicate mesozooplankton abundance is one factor governing planktivore abundance during the summer. The significances of the regressions imply a substantial degree of bottom up control by the mesozooplankton on the planktivores. One underlying reason for the entirely different relationship in the oligohaline is likely to be the dramatic shift in zooplankton community structure from a relatively diverse, freshwater community in the tidal fresh to an *Acartia* dominated, more estuarine community in the oligohaline. These changes would have significant ramifications to the mesozooplankton - planktivore relationship.

The very shallow slope of the regression at the Patuxent fresh/oligohaline station (.00473) suggests other factors play an important role in regulating the planktivores at that station. One factor may be the transitional nature of the zooplankton community structure. Of all the oligohaline stations, the zooplankton species composition at this station is most like those in tidal fresh stations except for frequent incursions by *Acartia*. Similarly, summer planktivore abundances here are more comparable to those in the tidal fresh. Nutrient loadings and ambient concentrations are relatively high here, indicating another possible cause for low mesozooplankton abundance. Summer flow does not appear to influence the relationship.

The absence of a mesozooplankton - planktivore relationship Potomac (RET2.3) and James (RET5.2) suggests the linkage between the two trophic levels has been uncoupled at these stations by other, strong controlling factors. Toxics are a likely candidate for the Potomac station. 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 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* is known to be sensitive to trace metals and population crashes elsewhere in the Bay have been associated with elevated metal concentrations (Sunda et al. 1990). The lack of a regression between mesozooplankton and planktivores at the James oligohaline station may be a result of rapidly increasing fall-line nutrient loadings, as appears to be the case upriver at the tidal fresh station. Abundances of both mesozooplankton and planktivores are low in the James relative to those in other oligohaline stations.

A positive slope in a significant regression between predator and prey abundances usually indicates prey abundance controls predator abundance to a large extent. An equally valid interpretation is a third variable governs both predator and prey in a similar way. In oligohaline reaches, there do not

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[3] Summer salinities at none of these stations were strictly oligohaline (0.5 - 5 ppt). The uppermost stations in the Choptank and the Patuxent typically experienced both fresh and oligohaline conditions during the summer. The middle Patuxent and Potomac stations experience both oligohaline and low mesohaline (5 - 10 ppt) conditions. And the upper Bay station MCB2.2 experienced fresh, oligohaline and low mesohaline conditions. Data for summers with predominantly tidal fresh conditions have been removed from the MCB2.2 regression.

appear to be any factors that could similarly control mesozooplankton and planktivore abundances. Numerous piscivores can crop juvenile and adult planktivores but not mesozooplankton, and could be expected to bring mesozooplankton - planktivore regression slopes (as shown in Figures 29 - 34) to a vertical line. A substantial population of facultative planktivores in the oligohaline (e.g. striped bass Y-O-Y, mummichog, sticklebacks, sheepshead minnow, and the rainwater, striped, marsh, and spotfin killifishes) could conceivably regulate both planktivores and mesozooplankton and this possibility remains to be examined in these systems. A cursory look at the fish communities in Maryland (Carmichael et al. 1992) suggests this possibility is unlikely there but needs to be rigorously examined. Jellyfish predators can influence both mesozooplankton and planktivores, however they are rare in the oligohaline (Lippson et al. 1979). Likewise, most meroplankton planktivores can impact mesozooplankton and ichthyoplankton but their populations are small in the oligohaline and typically pulsed. It appears, then, that the first interpretation of the positive regression slopes in the oligohaline may be the correct one; namely, mesozooplankton abundance exerts significant bottom-up control on planktivore abundance.

Tributary mesohaline (5 - 18 ppt salinity). The mesohaline covers extensive stretches of water 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 mesohalines 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, with this data we can look for mesozooplankton -planktivore linkages at only four tributary mesohaline sites.

Summer densities of zooplankton at the tributary mesohaline stations varied tremendously, with MET5.2 (Choptank) and LE1.1 (Patuxent) typically exhibiting relatively high averages and MLE2.2 (Potomac) and RET3.1 (Rappahannock) exhibiting modest averages. Densities in the mesohaline were comparable to those in the oligohaline on a tributary by tributary basis. In other words, the Choptank and the Patuxent had high densities in the oligohaline and also downstream in the mesohaline. Likewise, the Potomac had modest densities in the oligohaline as well as mesohaline. *Acartia tonsa* again dominated the summer mesozooplankton in this salinity regime. Planktivore abundances near zooplankton monitoring stations were moderate to low in the mesohaline, except in the Choptank. All the stations stratify to some extent during the summer, the Choptank and Rappahannock sporatically 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 intruded the Choptank station from the Bay mainstem. Flow did not affect the July - September abundances.

No correlations were evident between mesozooplankton and planktivore abundance at any of the four tributary mesohaline stations, including the Choptank which experienced the largest range of both the predator and prey abundances (Figures 35 - 38). The implication here is that the trophic linkage between strict planktivores and their principal prey, the mesozooplankton, is either masked or uncoupled by the effects of other factors. There are several factors in the tributary mesohaline waters that exert strong controls on mesozooplankton populations: a diverse, abundant collection of zooplankton predators, and chronic summer hypoxia/anoxia. 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. Multiple regression of the various predators, bottom dissolved oxygen and mesozooplankton may determine how much of a role total predator abundance plays in regulating mesozooplankton abundance during July - September and whether bottom dissolved oxygen affects the overall predator-prey relationship.

Discussion

Results of these straightforward correlative analyses of Chesapeake Bay zooplankton monitoring data and the Maryland and Virginia juvenile finfish seine survey data seem to support both the hypotheses stated above, namely 1) significant regressions between mesozooplankton and finfish planktivore abundances will be evident during the summer growing period (indicating the mesozooplankton - planktivore linkage is tightly coupled at that time), and 2) the regressions will have either shallow or steep slopes, or be weak or non-significant if other controlling factors (e.g. salinity, water quality, high flow, predation) strongly influence the mesozooplankton, the finfish planktivores or both. The lack of signficiant correlations at mesohaline stations, and at tidal fresh and oligohaline stations that appear to be seriously affected by toxins or rapidly changing nutrient loads, suggests there are environmental limits outside of which straightforward trophic linkages between strict finfish planktivore and mesozooplankton cannot be discerned and may even be uncoupled.

The strikingly different slopes in the tidal fresh and oligohaline regressions indicate a fundamental change takes place in zooplankton - fish linkages at the fresh/oligohaline interface, and this change may be relevant to fisheries management strategies in the tributaries and upper Bay. Planktivores in tidal fresh waters exert strong top-down controls on their prey, as evidenced by inverse regression slopes, and need to be considered one of the factors governing mesozooplankton abundance and community structure in these areas. This echoes a pattern found repeatedly in freshwater lakes. Convérsely, positive slopes in the oligohaline manifest strong bottom-up controls by prey on their predators and, in view of the predominance of planktivorous Y-O-Y and planktivore species, indicate mesozooplankton need to be considered one of the factors governing fish populations there. The fundamental change in zooplankton - fish linkage at the fresh/oligohaline interface appears to be the result of major shifts in prey vulnerability, predator-prey overlap, and predator efficiency brought about by environmental changes. The relative dominance of predator and prey responses determines whether correlations between predator and prey populations are positive, negative, or absent when predation pressures are strong (Williamson et al. 1989). Since planktivore predation appears to be strong in tidal fresh and oligohaline waters, except the York (TF4.2), a simple comparison of the tidal fresh and oligohaline habitats and communities highlights the factors probably effecting shifts in predator/prey interactions. 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.). This loss is somewhat compensated for by higher turbidity in the oligohaline which shrinks the reactive zones of visual planktivores (although not of Atlantic menhaden, the dominant Bay species). Prey vulnerability is also lowered in the oligohaline by a major, salinity-induced shift in zooplankton species composition from a relatively diverse, freshwater community dominated by cladoceran species to an *Acartia*-dominated, estuarine community. *Acartia* are tolerant of a wide range of salinities. They are omnivores capable of selectively consuming detritus, net phytoplankton and even smaller zooplankton (refs.) and are therefore well adapted to utilizing the enormous amounts of detritus generated as freshwater species die out. They are also copepods and therefore better adapted for escaping fish predators than are the slower moving Cladocera which rely more on vertical migration and transparency to avoid predation. A shift towards an *Acartia* dominated community would reduce the influence of both salinity and predation as controlling factors on the overall mesozooplankton population. Our initial conclusion, therefore, is that fundamental changes at the fresh/oligohaline interface cause predator and prey responses to shift significantly with respect to each other, consequently changing the dominant direction of trophic control from top-down in the tidal fresh to bottom-up in the oligohaline. The high correlation coefficients of the regressions tell us linkages between planktivore predators and their mesozooplankton prey are strong in both salinity regimes.

The diverse array of zooplankton predators present 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). The situation is complex and dynamic. 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 nettle, can comsume 470 copepods per hour (Bishop 1967) and population maxima in mid-summer have been negatively associated with east coast estuarine copepod abundances (Mountford 1980, refs.). The impact of this invertebrate planktivore is reduced when *Chrysaora quinquecirrha*, a jellyfish predator of the sea nettle and zooplankton, reaches its annual maximum (refs.). Multiple regressions need to be done for the tributary mesohaline stations to determine if the *combined* predation of invertebrate *and* finfish planktivores show a clear, consistent relationship to mesozooplankton abundance.

Similar analyses of historical monitoring data (1976 - 1980) from mesohaline waters of the Chesapeake Bay mainstem near Calvert Cliffs indicates this method has promise. The Calvert Cliffs studies done by the Academy of Natural Sciences show evidence of planktivore - mesozooplankton relationships. Olson (1987) used weekly and monthly data in step-wise 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), an invertebrate predator of zooplankton *Neomysis* (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 1970's, and whether these linkages have changed in the last 15 years of increasing eutrophication.

The mesozooplankton - planktivore relationships described for the Chesapeake Bay estuary indicate a "trophic cascade effect" in action in the Bay. This concept was derived from recurring patterns of trophic interactions observed in freshwater lakes over many decades and recently synthesized into an overaching 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[4] (Carpenter et al. , 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. 1984, Gophen et al. 1990, Elser and

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[4] 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 (from *Chesapeake Bay Stategy for the Restoration and Protection of Ecologically Valuable Species*, 1993, Chesapeake Bay Program).

Carpenter 1988, Mills and Green 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 pre-manipulation 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, i.e. an inverse relationship. Development and maintenance of a sizeable 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) will 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 rather than to the microbial loop.

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, suggests 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 strict planktivores.

Recommendations

Many avenues of investigation remain to be explored with Chesapeake Bay biomonitoring data. First, only the juvenile summer seine surveys have been used so far in our effort link zooplankton and finfish monitoring programs. There are a number of trawl surveys, done throughout the Bay during different seasons, whose data would give us 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 strict planktivores and their obligate prey, the mesozooplankton. For example, finfish planktivory elicits specific changes in zooplankton size frequency distributions, abundance of invertebrate planktivores, and prey vulnerability responses (refs.). The piscivore : planktivore ratio frequently correlates with mesozooplankton abundance and the proportion of large-bodied zooplankters (refs.). Zooplankton abundance and composition in turn can influence fish abundance, age-class size, and community structure (refs.). Preliminary calculations suggest these kinds of analyses will work in Chesapeake Bay tidal waters. For example, zooplankton size frequency distributions in the tidal fresh shift towards smaller sizes during the July - September time period (Figures 39 and 40). In the *Acartia* dominated oligohaline and mesohaline, a more appropriate way to investigate compression of the zooplankton size frequency distribution may be to look at the ratio of nauplii : copepodites : adult copepods. The abundance of *Leptodora kindtii* (Cladocera), a very large zooplankton predator of other zooplankton in the tidal fresh and species easily spotted and caught by finfish, decreases in abundance when finfish planktivores become abundance, indicating an possible imbalance in the proportion of invertebrate and vertebrate planktivores (Fig 41). The Calvert Cliffs data analyses (Olson 1987), as well as the research of Cowan and Houde (1992) and others, suggests these methods may prove especially useful in analyzing plankton - finfish linkages in the mesohaline. Finfish and invertebrate planktivores have very different relatiohships with their prey in estuarine waters, and each can possibly obscure effects of the other in simple ecosystem regression models such as was done for this paper. In summary, future examination of other evidences of plankton - fish interactions may confirm suspected linkages at some stations and identify environmental variables that are disrupting the linkages at other stations. The time limitations of the current project has prevented us from exploring the possibilities.

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Table 3. Station matches for zooplankton and juvenile finfish seine surveys in Maryland and Virginia. ( ) indicates 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 4. Strict planktivore and piscivore finfish species in Chesapeake Bay. All ages, except where noted. [1] indicates ages 1 year and older.

Planktivore	Piscivore
Alewife	Atlantic needlefish
American shad	Black crappie[1]
Atlantic menhaden	Bluefish
Atlantic silverside	Largemouth bass[1]
Atlantic thread herring	Longnose gar[1]
Banded killifish	Striped bass[1]
Bay anchovy	Yellow perch[1]
Blueback herring	
Bridle shiner	
Comely shiner	
Gizzard shad	
Golden shiner	
Spottail shiner	
Striped anchovy	
Pipefish	

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Figure 39. Size frequency distribution of the tidal fresh Potomac mesozooplankton (Cladocera and Copepoda) community, averaged by month for the entire monitoring period and converted to percentage of the total.

Figure 40. Size frequency distribution of the tidal fresh Patuxent mesozooplankton (Cladocera and Copepoda) community, averaged by month for the entire monitoring period and converted to percentage of the total.

Figure 41. Overall summer abundance of *Leptodora kindtii* versus planktivore for the entire monitoring period, at stations experiencing <2 ppt salinity during the summer. The regression equation, r^2 value, and the probability that the slope is not equal to zero are given in the graph. See text for details concerning calculation of averages.

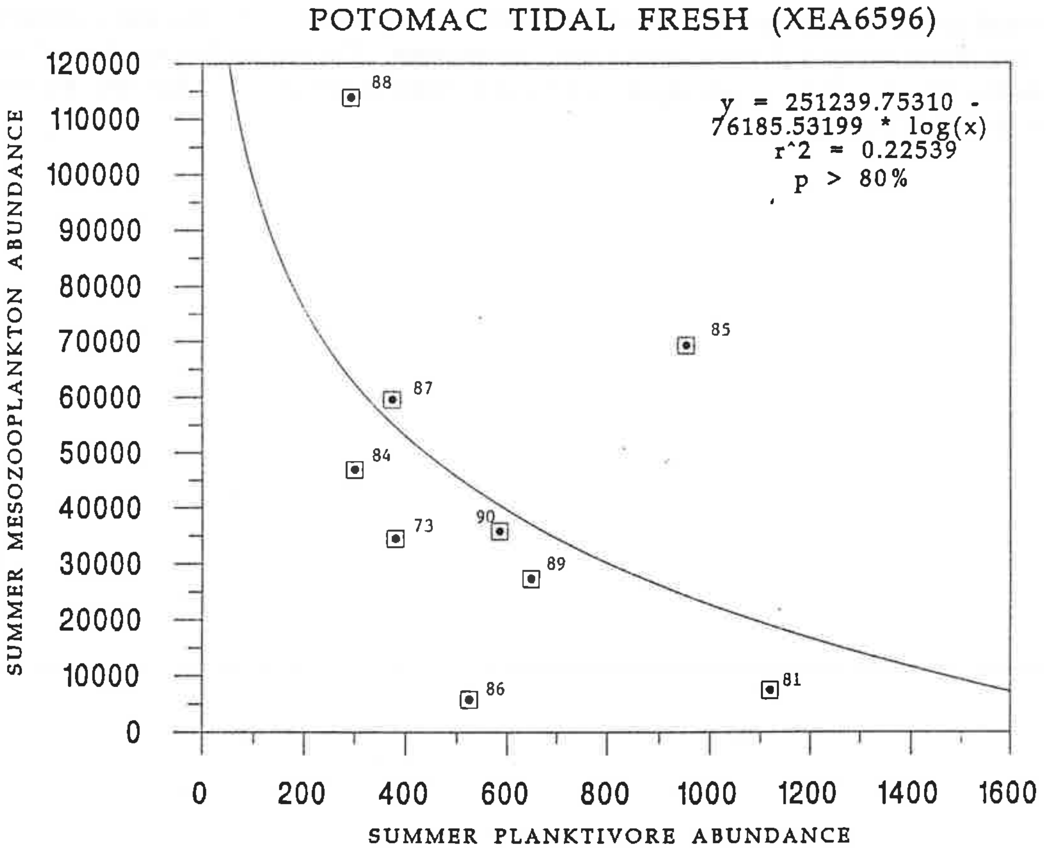
POTOMAC TIDAL FRESH (XEA6596)
SUMMER MESOZOOPLANKTON ABUNDANCE
SUMMER PLANKTIVORE ABUNDANCE
0
10000
20000
30000
40000
50000
60000
70000
80000
90000
100000
110000
120000
0
200
400
600
800
1000
1200
1400
1600
y = 251239.75310 -
76185.53199 * log(x)
r^2 = 0.22539
p > 80%
88
87
84
73
90
89
86
85
81

Figure 25.

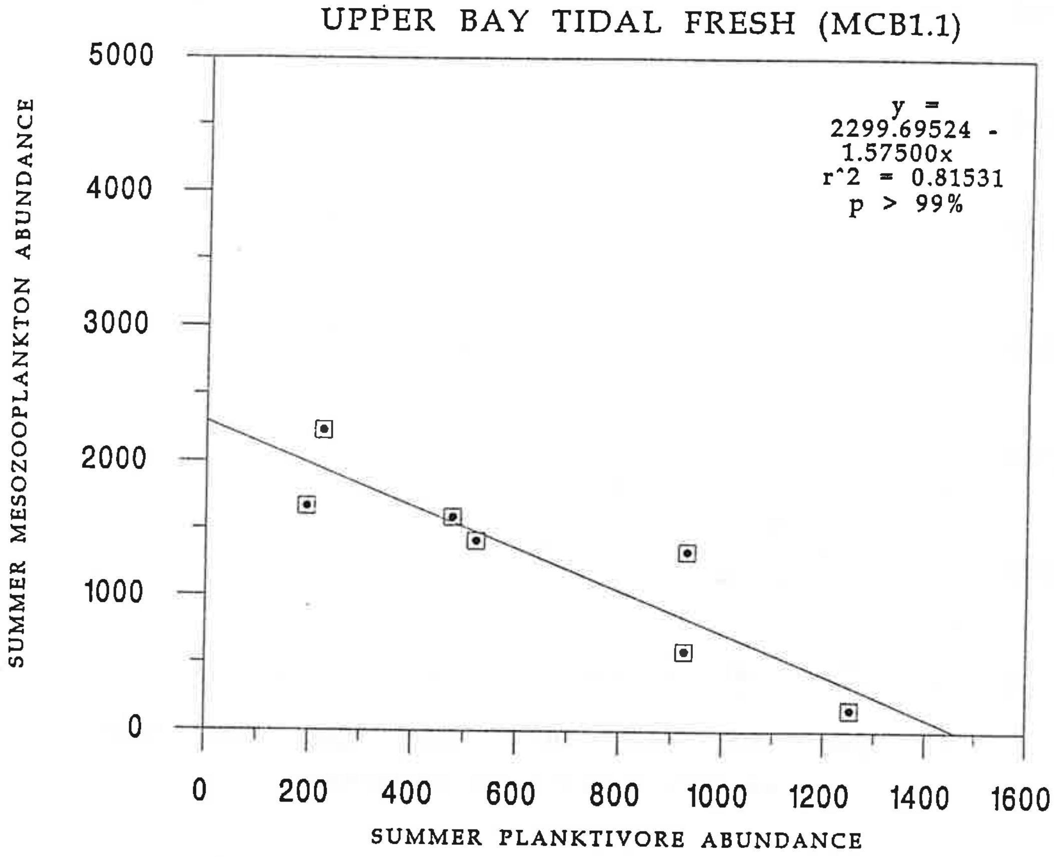
UPPER BAY TIDAL FRESH (MCB1.1)
y =
2299.69524 -
1.57500x
r^2 = 0.81531
p > 99%
SUMMER MESOZOOPLANKTON ABUNDANCE
0
1000
2000
3000
4000
5000
0
200
400
600
800
1000
1200
1400
1600
SUMMER PLANKTIVORE ABUNDANCE

Figure 26.

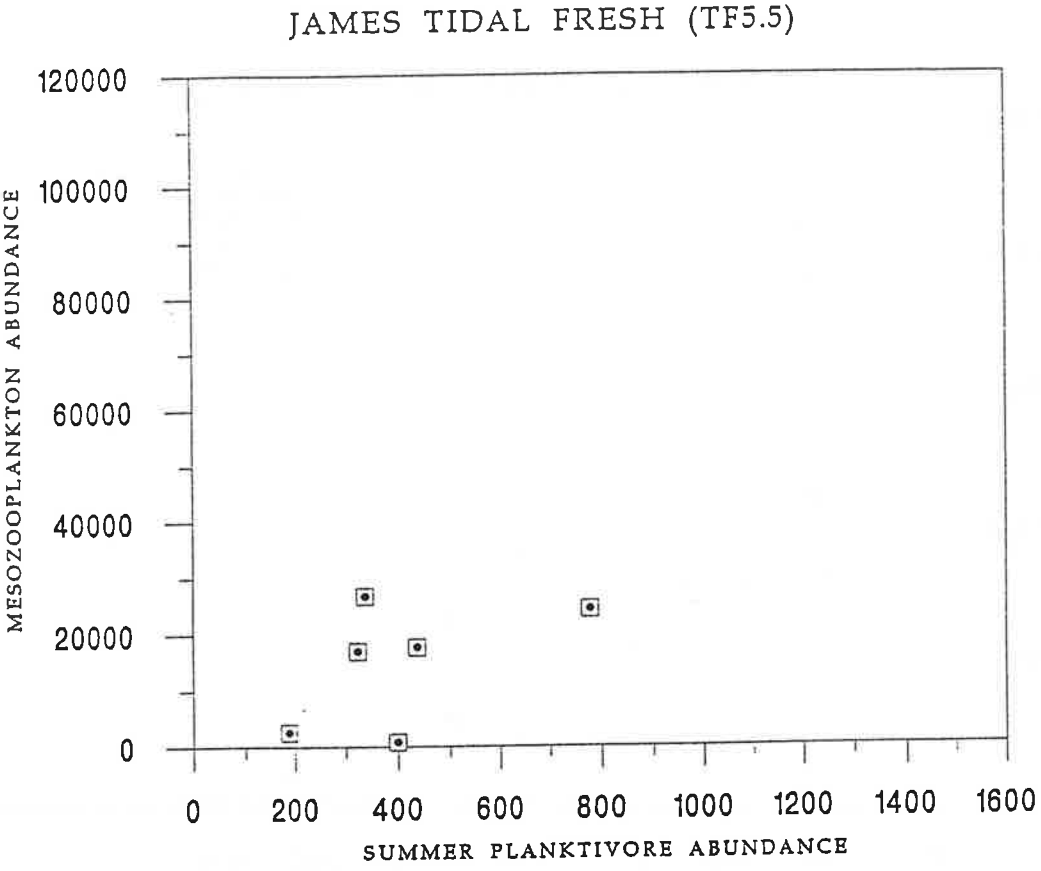
JAMES TIDAL FRESH (TF5.5)
120000
100000
80000
60000
40000
20000
0
MESOZOOPLANKTON ABUNDANCE
0
200
400
600
800
1000
1200
1400
1600
SUMMER PLANKTIVORE ABUNDANCE

Figure 27.

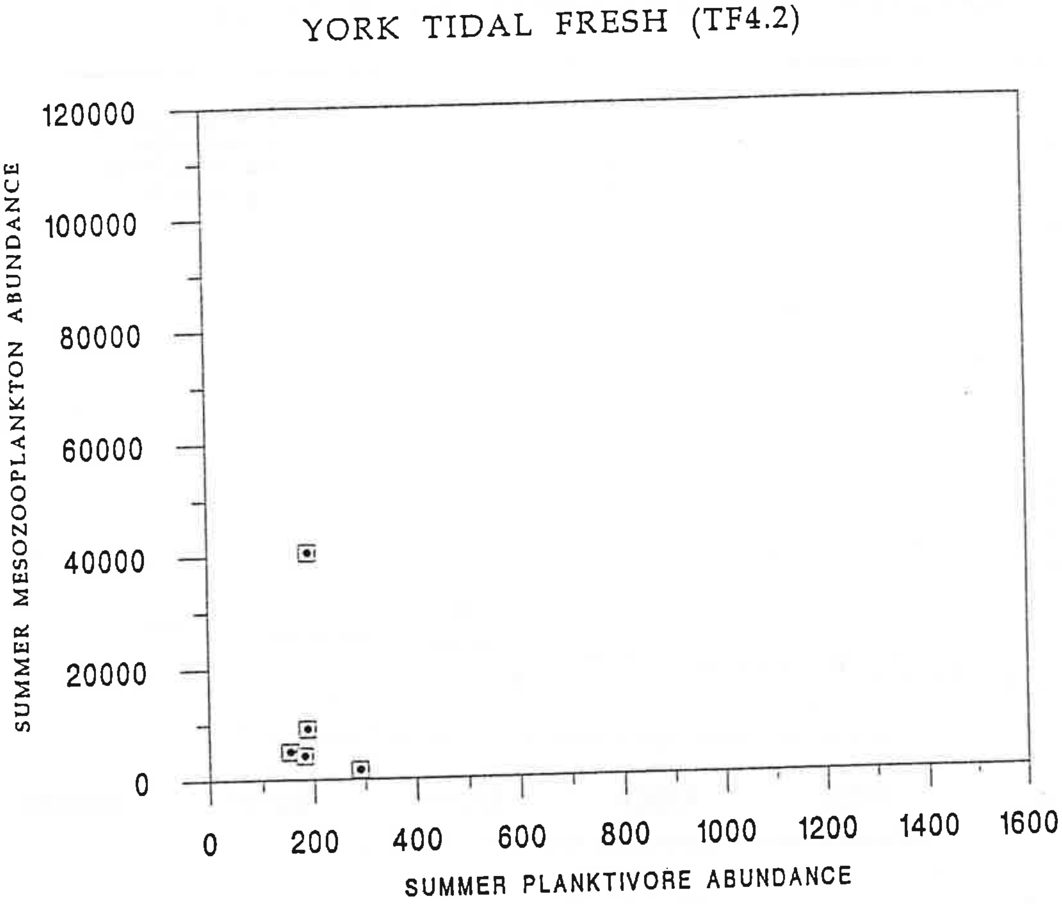
YORK TIDAL FRESH (TF4.2)
SUMMER MESOZOOPLANKTON ABUNDANCE
120000
100000
80000
60000
40000
20000
0
0
200
400
600
800
1000
1200
1400
1600
SUMMER PLANKTIVORE ABUNDANCE

Figure 28.

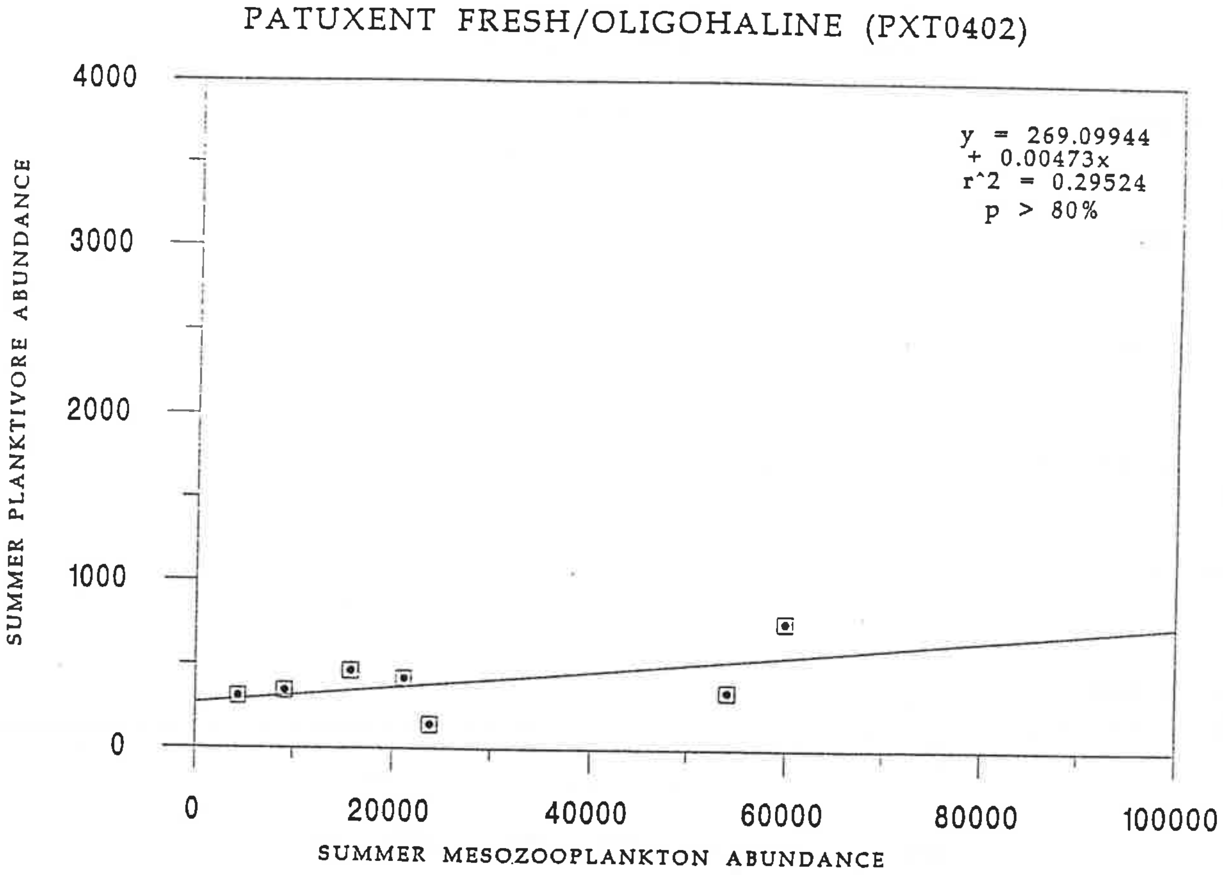
PATUXENT FRESH/OLIGOHALINE (PXT0402)
y = 269.09944
+ 0.00473x
r^2 = 0.29524
p > 80%
SUMMER PLANKTIVORE ABUNDANCE
4000
3000
2000
1000
0
0
20000
40000
60000
80000
100000
SUMMER MESOZOOPLANKTON ABUNDANCE

Figure 29.

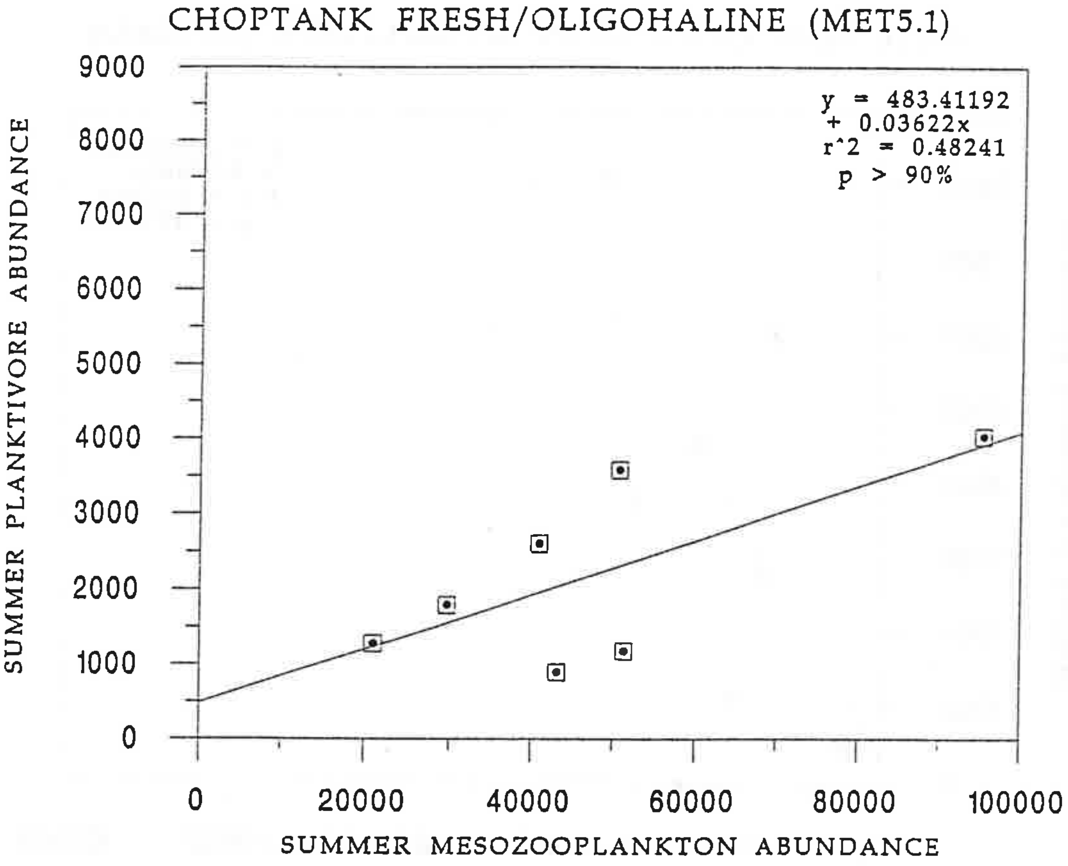
CHOPTANK FRESH/OLIGOHALINE (MET5.1)
y = 483.41192
+ 0.03622x
r^2 = 0.48241
p > 90%
SUMMER PLANKTIVORE ABUNDANCE
9000
8000
7000
6000
5000
4000
3000
2000
1000
0
0
20000
40000
60000
80000
100000
SUMMER MESOZOOPLANKTON ABUNDANCE

Figure 30.

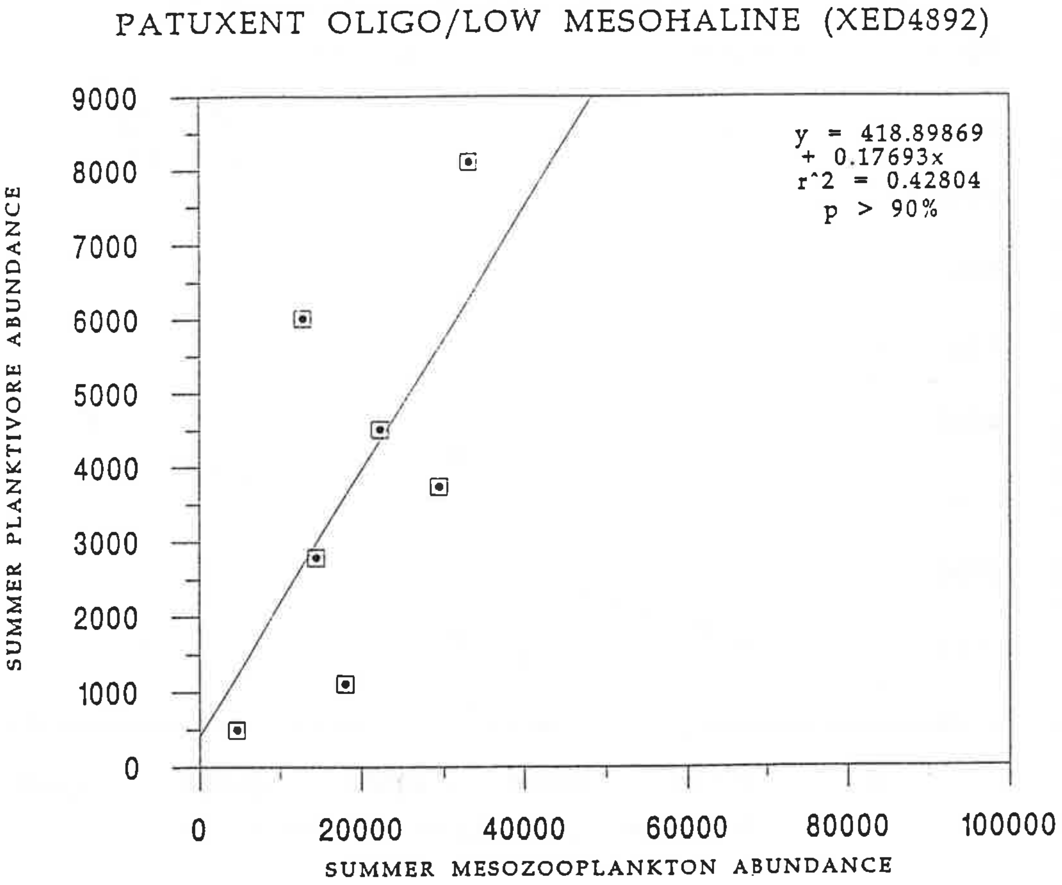
PATUXENT OLIGO/LOW MESOHALINE (XED4892)
y = 418.89869
+ 0.17693x
r^2 = 0.42804
p > 90%
SUMMER PLANKTIVORE ABUNDANCE
0
1000
2000
3000
4000
5000
6000
7000
8000
9000
SUMMER MESOZOOPLANKTON ABUNDANCE
0
20000
40000
60000
80000
100000

Figure 31.

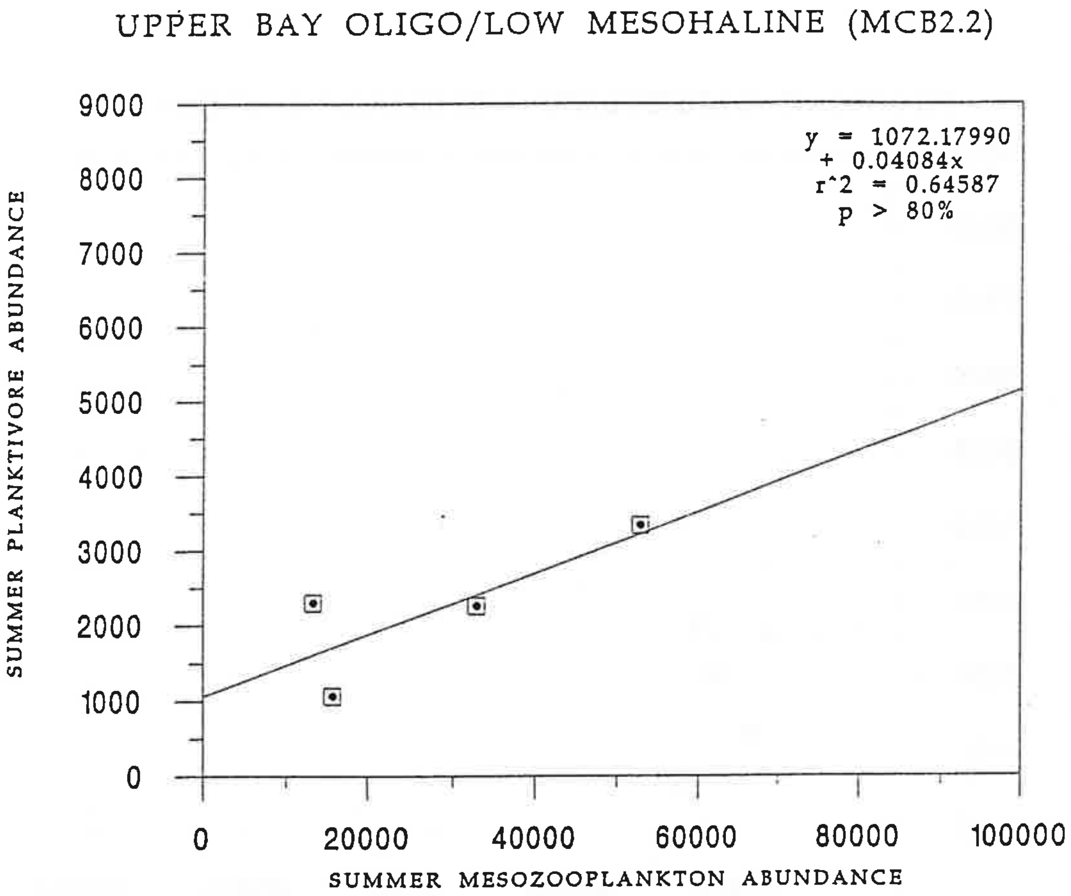
UPPER BAY OLIGO/LOW MESOHALINE (MCB2.2)
SUMMER PLANKTIVORE ABUNDANCE
9000
8000
7000
6000
5000
4000
3000
2000
1000
0
0
20000
40000
60000
80000
100000
SUMMER MESOZOOPLANKTON ABUNDANCE
y = 1072.17990
+ 0.04084x
r^2 = 0.64587
p > 80%

Figure 32.

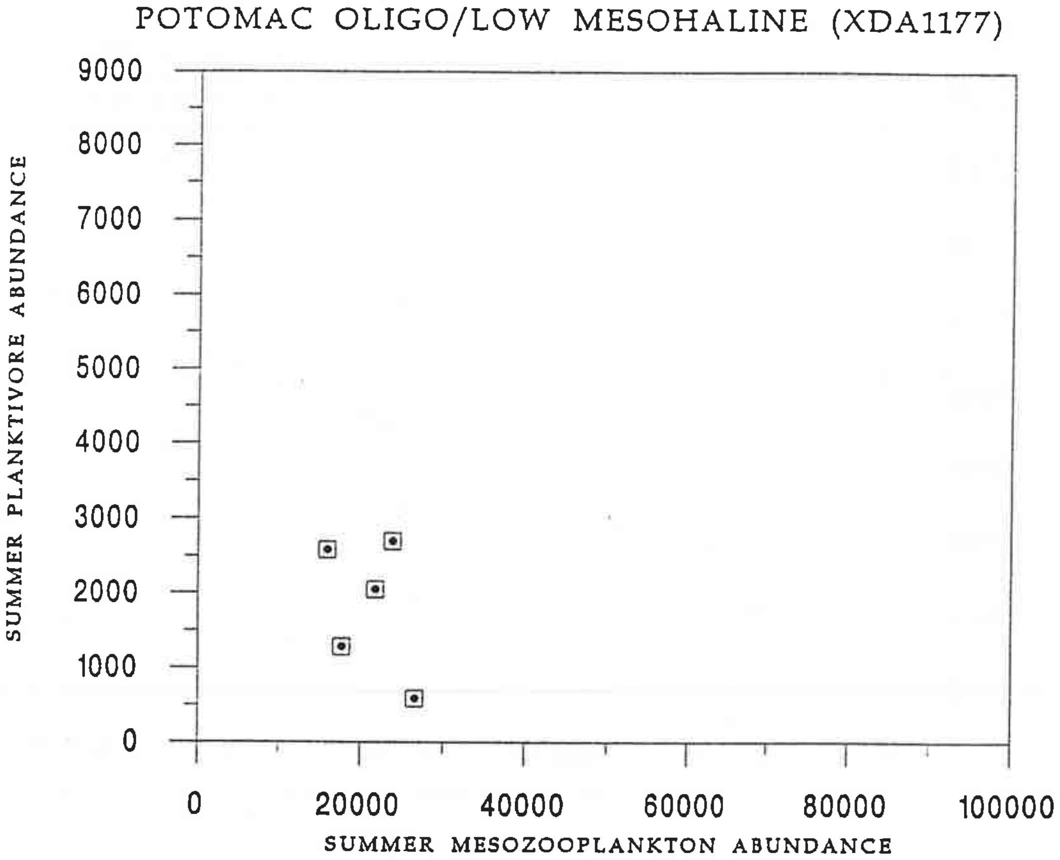
POTOMAC OLIGO/LOW MESOHALINE (XDA1177)
9000
8000
7000
6000
5000
4000
3000
2000
1000
0
SUMMER PLANKTIVORE ABUNDANCE
0
20000
40000
60000
80000
100000
SUMMER MESOZOOPLANKTON ABUNDANCE

Figure 33.

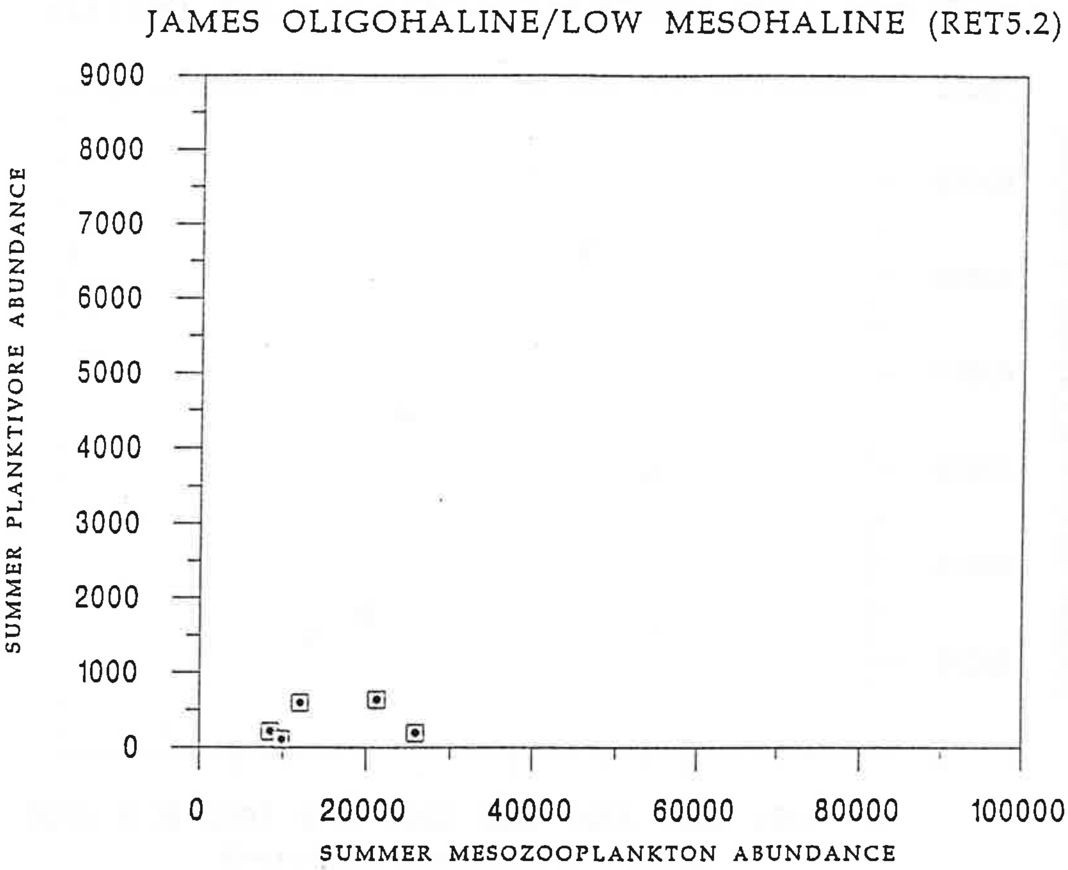
JAMES OLIGOHALINE/LOW MESOHALINE (RET5.2)
SUMMER PLANKTIVORE ABUNDANCE
9000
8000
7000
6000
5000
4000
3000
2000
1000
0
0
20000
40000
60000
80000
100000
SUMMER MESOZOOPLANKTON ABUNDANCE

Figure 34.

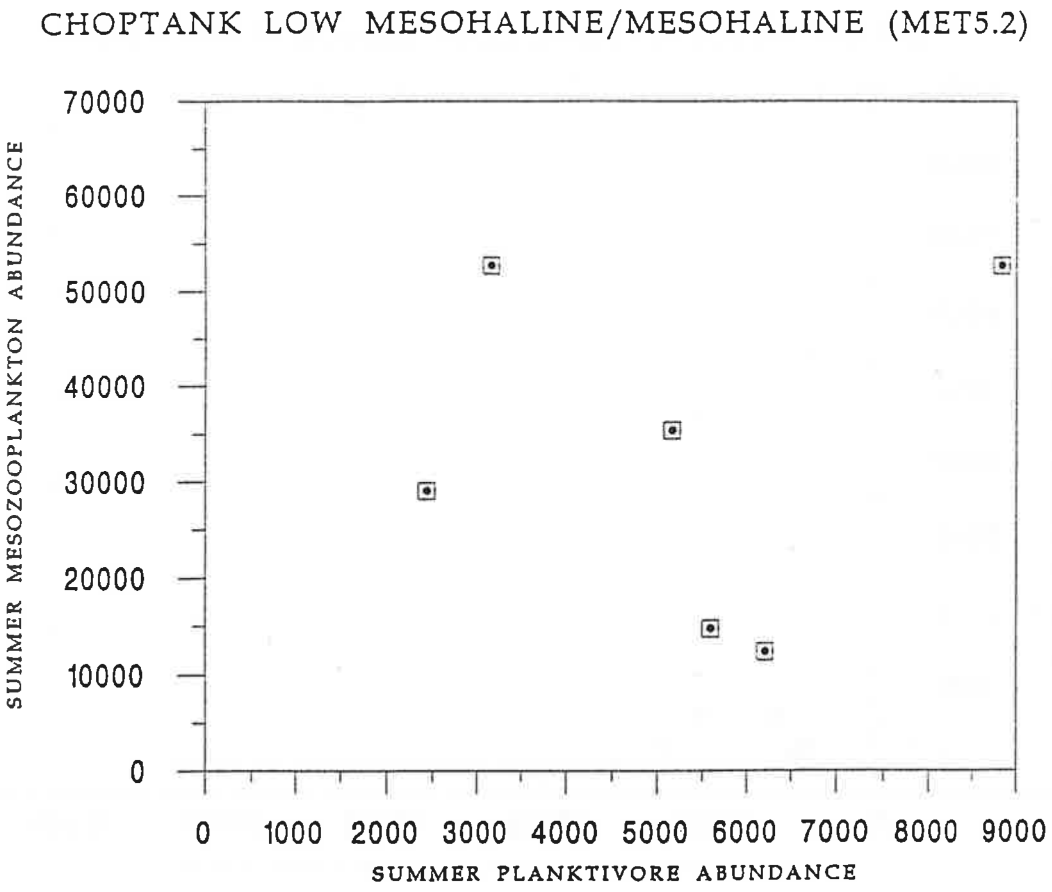
CHOPTANK LOW MESOHALINE/MESOHALINE (MET5.2)
SUMMER MESOZOOPLANKTON ABUNDANCE
70000
60000
50000
40000
30000
20000
10000
0
0
1000
2000
3000
4000
5000
6000
7000
8000
9000
SUMMER PLANKTIVORE ABUNDANCE

Figure 35.

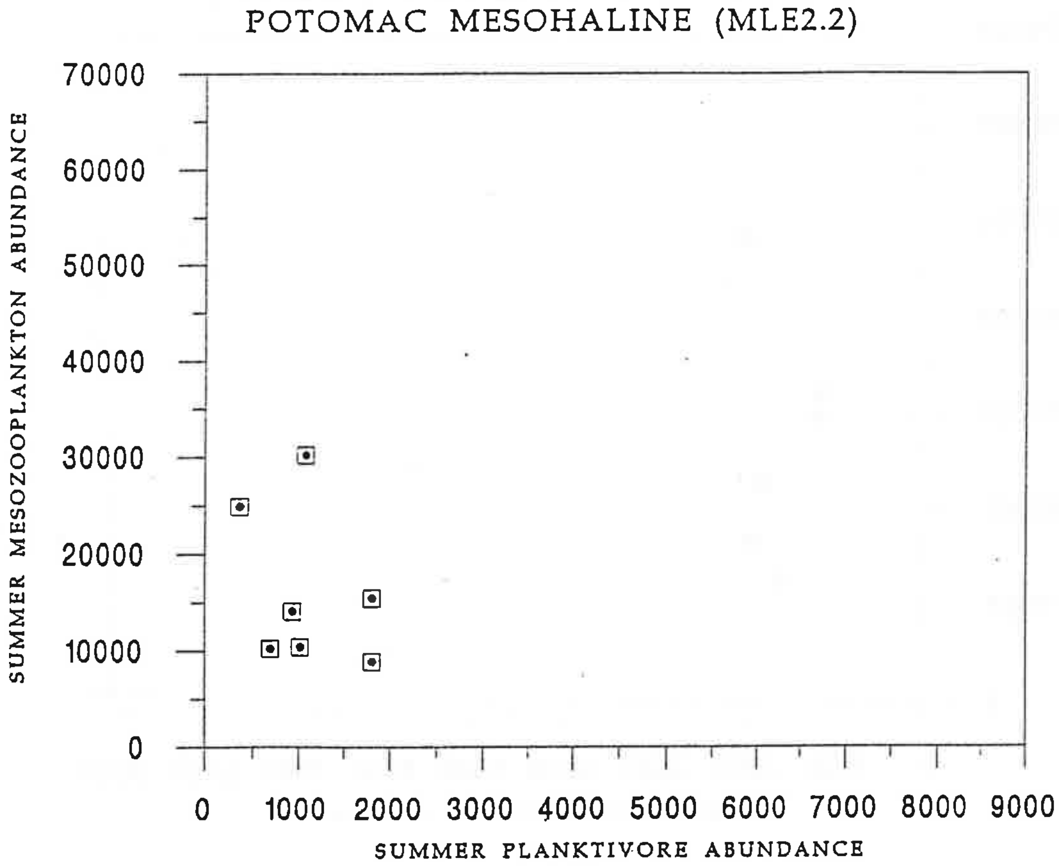
POTOMAC MESOHALINE (MLE2.2)
70000
60000
50000
40000
30000
20000
10000
0
SUMMER MESOZOOPLANKTON ABUNDANCE
0
1000
2000
3000
4000
5000
6000
7000
8000
9000
SUMMER PLANKTIVORE ABUNDANCE

Figure 36.

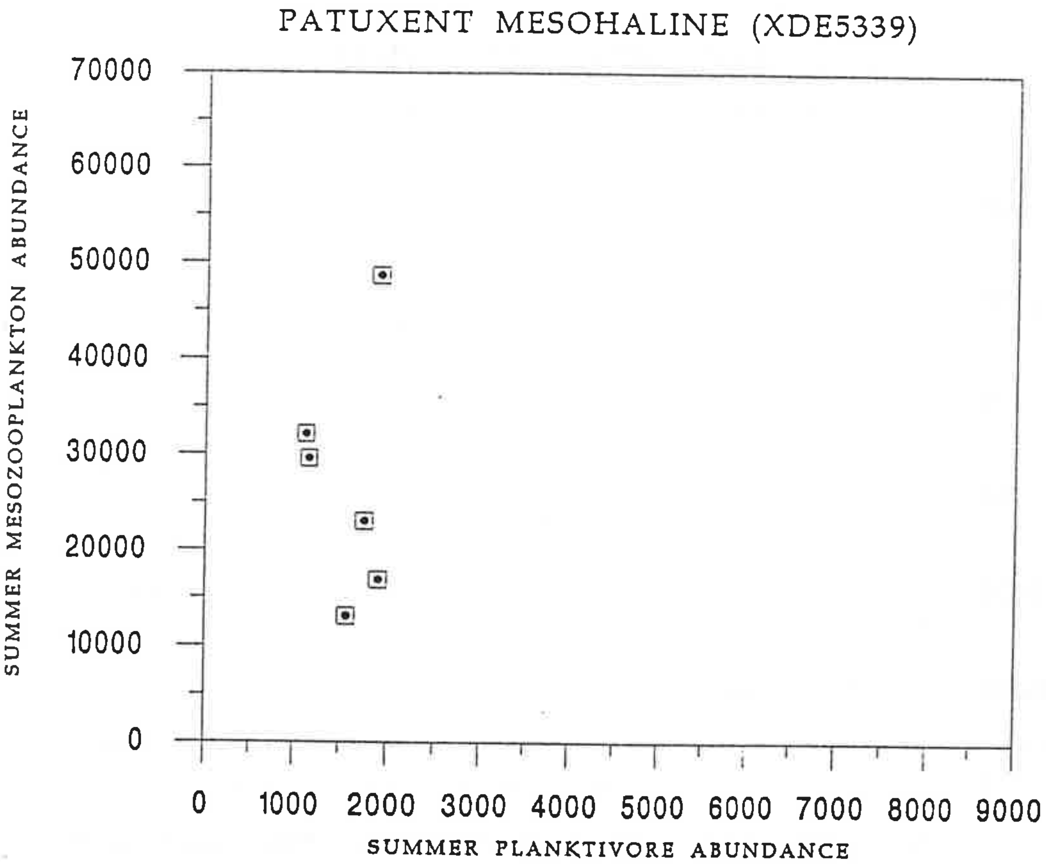
PATUXENT MESOHALINE (XDE5339)
SUMMER MESOZOOPLANKTON ABUNDANCE
70000
60000
50000
40000
30000
20000
10000
0
0
1000
2000
3000
4000
5000
6000
7000
8000
9000
SUMMER PLANKTIVORE ABUNDANCE

Figure 37.

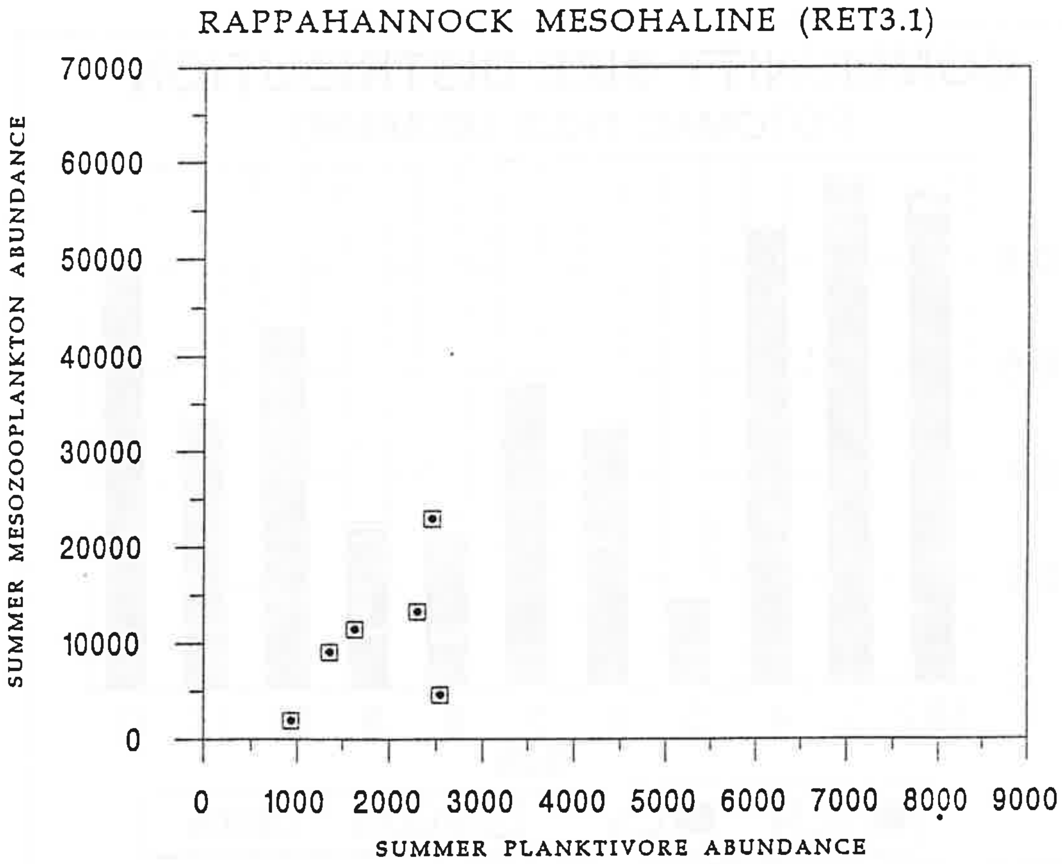
RAPPAHANNOCK MESOHALINE (RET3.1)
SUMMER MESOZOOPLANKTON ABUNDANCE
70000
60000
50000
40000
30000
20000
10000
0
0
1000
2000
3000
4000
5000
6000
7000
8000
9000
SUMMER PLANKTIVORE ABUNDANCE

Figure 38.

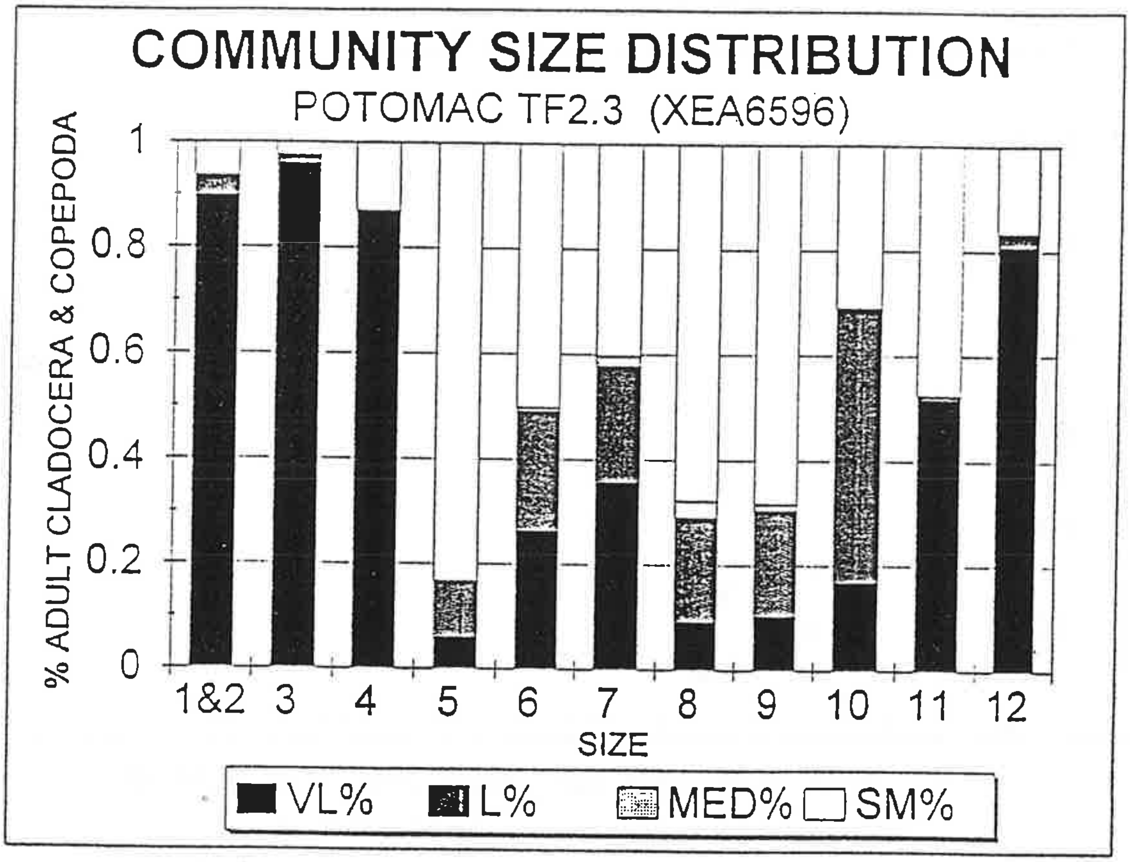
COMMUNITY SIZE DISTRIBUTION
POTOMAC TF2.3 (XEA6596)
% ADULT CLADOCERA & COPEPODA
1
0.8
0.6
0.4
0.2
0
1&2
3
4
5
6
7
8
9
10
11
12
SIZE
VL%
L%
MED%
SM%

Figure 39.

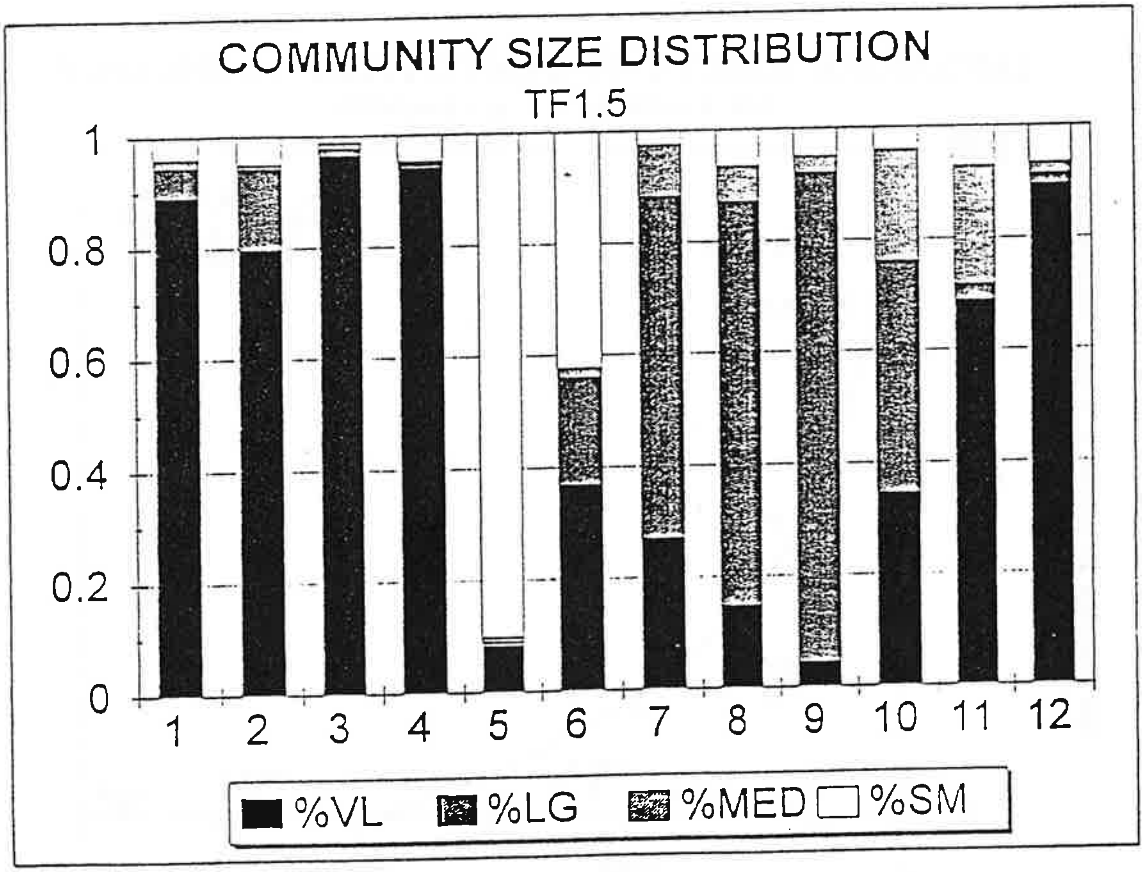
COMMUNITY SIZE DISTRIBUTION
TF1.5
1
0.8
0.6
0.4
0.2
0
1
2
3
4
5
6
7
8
9
10
11
12
%VL
%LG
%MED
%SM

Figure 40.

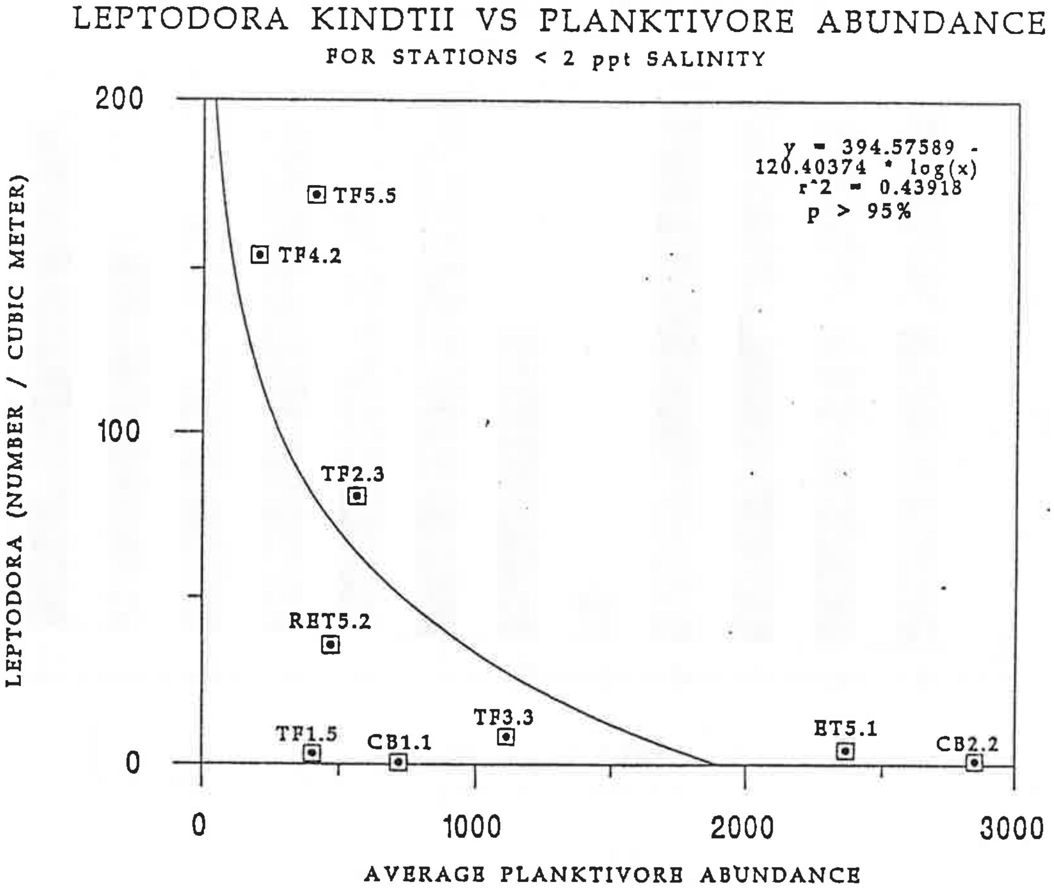
LEPTODORA KINDTII VS PLANKTIVORE ABUNDANCE
FOR STATIONS < 2 ppt SALINITY
y = 394.57589 - 120.40374 * log(x)
r^2 = 0.43918
p > 95%
TF5.5
TF4.2
TF2.3
RET5.2
TF1.5
CB1.1
TF3.3
ET5.1
CB2.2
200
100
0
0
1000
2000
3000
LEPTODORA (NUMBER / CUBIC METER)
AVERAGE PLANKTIVORE ABUNDANCE

Figure 41.

Index of Biotic Integrity

Summary

A preliminary assessment was conducted to determine the feasibility of developing an "index of biotic integrity" (IBI) system for zooplankton similar to IBIs which have been developed for fish communities and, more recently, for the Chesapeake Bay Benthic Restoration Goals. To allow the calculation of IBI scores, selected sites in each salinity regime were subjectively classified as environmentally "stressed" or "nonstressed". The IBI scores were calculated for four salinity regimes (tidal fresh, oligohaline, mesohaline, polyhaline) and two seasons (spring and summer) from data sets comprising various combinations of nine potential bioindicators: mesozooplankton abundance and biomass; microzooplankton abundance and biomass; mesozooplankton diversity (Shannon-Wiener); ratio of microzooplankton to mesozooplankton abundance;mesozooplankton variability;ratio of calanoid copepods to the sum of cyclopoid copepods plus cladocerans; and the percentage of the zooplankton community represented by *Bosmina spp.*.

Despite the limited time and resources available for the development and evaluation of the IBI system for the assessment and summarization of the health of Chesapeake Bay zooplankton communities, the results of the preliminary IBI effort did appear to hold promise for future development/refinement efforts. Canonical and classification discriminant analyses of the IBI scores for the bioindicators indicated significant discrimination between the "nonstressed" and "stressed" sites for the tidal fresh, mesohaline, and polyhaline salinity regimes. In fact, the stations with the most clearly documented "stressed" conditions (i.e. the known toxic conditions of Elizabeth River sites) had the most dramatic separation between these conditions. However, the findings of the assessment did indicate the need for future refinement of the zooplankton IBI approach for the Bay.

The number of misclassified samples from certain of the sites identified as "stressed" or "nonstressed" which were identified by the discriminant analyses has suggested that a more objective method for *a priori* classification of environmental conditions be developed in the future. Statistical approaches employing multivariate techniques have been recommended to delineate the sites that appear to have poor water quality from those that have better conditions. These groupings could be further assessed by discriminant analysis of the biological data to provide a means of identifying new potential bioindicators for future assessments. It is possible that certain of the salinity regimes in the Bay do not currently have clear-cut "stressed" and "nonstressed" regions, so other data sets (e.g. historical data from the Potomac River or data collected from other estuaries) may need to be explored to permit the full development and optimization of the IBI system. In addition, the effects of salinity and seasonality on the IBI scores within and between salinity regimes need to be explored further to determine whether there are alternate means of "correcting for" these non-anthropogenic factors statistically (e.g. alternate blocking schemes, regression-based adjustments, etc.) prior to the IBI calculations. Finally, it was recommended that, in future refinement efforts, the proposed and any new bioindicators be thoroughly assessed for ecological meaning and sensitivity to environmental stress, as well as for power and robustness in the measurement of changes and long term trends in water quality conditions and fish restorations.

A Preliminary "Index of Biotic Integrity" Approach To Summarizing Zooplankton Environmental Indicators

by Raymond W. Alden, III

Introduction

A variety of indices/metrics have been developed to aid managers in evaluating the environmental quality of aquatic ecosystems. Many of these indices focus upon the relative "health" of biological communities in the field. One approach being evaluated for application in a variety of ecosystems is the *Index of Biotic Integrity* (IBI). The IBI approach was developed for the assessment of the relative biotic integrity of fish communities in inland streams and rivers (Karr, 1981, 1987; Fausch *et al.*, 1984; Karr *et al.*, 1986; and many others). Biotic integrity has been defined as the ability of an ecosystem to support and maintain "*a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats of the region*" (Karr and Dudley, 1981); or, alternately, as a characteristic "*possessed by aquatic ecosystems in which composition, structure, and function have not been adversely impaired by human activities*" (Karr *et al.*, 1986). The approach has gained a considerable degree of acceptance among regulators and scientists: with applications in over two dozen states (Karr *et al.*, 1986); use by federal agencies such as the U.S. Army Corps of Engineers, the National Park Service, and the U.S. Environmental Protection Agency (Karr *et al.*, 1986; Hunsaker and Carpenter, 1990); use by other countries such as Canada and France (Hunsaker and Carpenter, 1990); as well as documented applications in over 30 journal publications (Hunsaker and Carpenter, 1990) and at least as many technical reports (Karr *et al.*, 1986).

Despite the promise of the IBI approach, there have been a number of potential problems identified with its use/misuse. In the first place, the IBI is not a universally applicable index. The approach was developed specifically for fish populations in Midwestern streams and lakes. Although it has been adapted for other regions, it has yet to be modified for use in the Southeast and Southwest, or for lakes and reservoirs (Hunsaker and Carpenter, 1990). Work on application of the IBI approach to fish communities in tidal fresh and estuarine ecosystems is only now being conducted (Steve Jordan, Maryland Department of Natural Resources, personal communications). The overall approach has only recently been considered for the assessment of biotic integrity of non-fish communities (*e.g.* for benthic communities, see below). Furthermore, the temporal and spatial variability of any IBI and the ecological meaningfulness of its components need to be well characterized prior to application for management purposes.

There have also been a number of philosophical and technical issues raised with the IBI approach. Suter (1993) presents a critique of the IBI. He identifies some of the major weaknesses of the IBI and similar approaches, which are summarized as follows:

1) **index ambiguity** - (*i.e.* with only the summary values, one cannot determine why the index may be high or low);

2) **eclipsing** - (*i.e.* a serious negative response may be hidden by other component measures that are not as severely affected);

3) unknown sensitivities of the results to the **computational scheme** used to aggregate

components;

4) unclear **relationships to "real" ecological properties**;

5) **unknown robustness** or universality of the metrics in their capacity to measure various types of "stress" or perturbations to environmental quality; and

6) the **oversimplification of multivariate responses** to a single monotonic vector.

To these potential problems can be added several other issues: the IBI approach requires a large data base with at least some unstressed (reference) habitats that must be identified *a priori* to define the threshold points for the classification scheme (see below); an independent data set should be tested to validate that the components of the IBI are meaningful in the discrimination of stressed from nonstressed conditions; it is unclear how the IBI "behaves" in terms of component sensitivity; and the power and robustness of IBIs in statistical analyses are generally unknown.

However, many of these potential problems can be alleviated by a thorough interpretation of the ecological meaning of the data (including the use of a variety of other analytical tools such as multivariate statistical techniques) prior to the use of the IBI approach for the purpose of data reduction/summarization for management use. The best professional judgement of the environmental experts must be used in the interpretation of the meaning of the underlying ecological patterns before an IBI ranking system can be considered to be valid. In other words, an IBI is not a panacea that can be used indiscriminately by the layman to reveal the environmental quality conditions of an ecosystem over time or space. Rather, it represents only one of a number of assessment techniques used by environmental scientists to identify and present patterns/trends of management interest.

With these caveats in mind, the principal investigators decided to explore an IBI-like approach to summarize the patterns observed for some of the zooplankton bioindicators being assessed by the present study. The specific approach taken was designed to parallel the efforts of benthic ecologists from VERSAR, Inc., the Virginia Institute of Marine Science, and Old Dominion University who are attempting to define *Chesapeake Bay Benthic Restoration Goals* (Dr. D. M. Dauer, Old Dominion University, personal communication). However, the findings reported in this section must be viewed as the results of a "first-cut" effort, with additional refinement/validation efforts being required before the approach can be used for management purposes. Some suggestions for the focus of refinement efforts will be presented in the **Recommendations** Section, while others will arise in the future, as more becomes known about the ecological meaning of the individual bioindicators which are the components of the IBI.

Calculation and Assessment of Zooplankton IBIs

The calculation of the components of the IBI followed the general approach employed by the benthic ecologists who worked on *Chesapeake Bay Benthic Restoration Goals* (CBBRG) for the Chesapeake Bay Program and the Maryland Department of Natural Resources (the report entitled *Chesapeake Bay Benthic Community Restoration Goals* by J. A. Ranasinghe, D. M. Dauer, and L. C. Schaffner is in draft form under review at the time of preparation of this report). For each of the zooplankton bioindicators, a classification system was developed based upon the characteristics of unstressed, reference (or minimally impacted) regions. Recognizing the potential influence of spatial and temporal patterns on all characteristics of zooplankton communities, the classification system was "blocked" by salinity regimes and "seasons" which were determined previously to represent ecologically significant time periods. Therefore, prior to classification, the data were divided according to salinity regime: tidal fresh (salinities < 0.5 ppt.); oligohaline (0.5 ppt ≤ salinities < 5 ppt.); mesohaline (5 ppt. ≤ salinities < 18 ppt.); and polyhaline (salinities ≥ 18 ppt.). The data were further partitioned by "season": Spring (March-May); Spring/Summer (June); Summer (July-September); Fall (October-November); and Winter (December-February). The CBBRGs were calculated with spatial (8 salinity-sediment type combinations) and temporal (only summer samples employed) blocking for similar reasons.

The salinity/season blocked data for each bioindicator was classified by the following scheme:

For Bioindicators Indicative of Nonstressed Conditions:

- Samples with bioindicator values that were less than the minimum value in the data set for reference conditions (site(s) selected *a priori* for the salinity regime) were rated with a value of '0';
- Samples with values that were greater than the minimum value and less than or equal to the 5-percentile value of the frequency distribution of the data for reference conditions were rated with a value of '1';
- Samples with values that were greater than the 5-percentile value and less than or equal to the median of the data for reference conditions were rated with a value of '3'; and
- Samples with values that were greater than the median of the data for reference conditions were rated with a value of '5'.

For Bioindicators Indicative of Stressed Conditions:

- Samples with bioindicator values that were greater than the maximum value in the data set for reference conditions were rated with a value of '0'.
- Samples with values that were less than or equal to the maximum value and greater than or equal to the 95-percentile value of the frequency distribution of the data for reference conditions were rated with a value of '1';
- Samples with values that were less than the 95-percentile value and greater than or equal to the median of the data for reference conditions were rated with a value of '3';
- Samples with values that were less than or equal to the median of the data for reference conditions were rated with a value of '5'.

All of the ranking values except for the zero rating parallelled the system adopted for the calculation of the CBBRGs. It was felt that the great variation of zooplankton community characteristics (*e.g.* "blooms" and "busts") would warrant a lower rating (0) for "outlier" values that may be particularly abnormal for nonstressed systems, but may be indicative of stressed conditions.

The IBI scores were calculated as the average of the component values that were applicable for each region (i.e. some bioindicators were not appriopriate, or data were not available, for certain regions; see below). This approach was taken for the CBBRGs for similar reasons.

Most of the bioindicators employed as components of the IBI have been discussed in the previous sections. These components generally cover many of the functional categories used in the fish IBIs and the CBBRGs. Table 5 summarizes the functional categories of the bioindicators employed, whether they are an indicator of stressed or unstressed conditions, and the regions for which they are applicable.

The only component that was not previously discussed was the Shannon-Wiener diversity index. This index was added only at the end of the project because problems were originally identified in the differences in the taxonomic sorting categories used in the Virginia and Maryland programs. These problems were partially resolved by blocking the data by state (as well as by salinity regime and season) prior to the calculation of the IBI component scores for diversity. It should be noted, however, that this additional blocking greatly reduces the numbers of observations from the reference data sets to very low levels, making this component of uncertain utility in certain regimes. The differences in taxonomic enumeration between the two programs should be explored and reconciled in the future to eliminate the need for the blocking protocol.

The validity/utility of the IBI (and its component bioindicators) as an indicator of stressed communities was explored in a preliminary fashion by calculating and comparing IBI scores for sites believed *a priori* to represent "stressed" and "nonstressed" habitats (Table 6). The scores from each of the bioindicator components of the IBI were also analyzed by canonical and classification discriminant analyses (SAS® **PROC CANDISC** and **PROC DISCRIM**) to determine whether the "stressed" and the "nonstressed" communities could be discriminated by the IBI component scores. This approach was similar to one used in the development of the CBBRGs. However, the zooplankton IBI assessment must be viewed as a much more preliminary effort, since time and resources were considerably more limited, and supplementary data sets and statistical assessments (e.g. the data sets and background statistical assessments from EMAP which were used in the development of the CBBRGs) were not available for the zooplankton.

Results and Discussion

The IBIs were calculated for three sets of data: Set 1 included Bay-wide data sets for tidal fresh, oligohaline, mesohaline, and polyhaline (only VA sites) regions employing IBI scores for mesozooplankton diversity, abundance, biomass and variability; Set 2 was comprised of Maryland data for tidal fresh, oligohaline, and mesohaline regions employing the Set 1 bioindicators plus IBIs for microzooplankton abundance, biomass, and the microzooplankton abundance to mesozooplankton abundance ratio; and Set 3 included Maryland data for the tidal fresh region only employing the Set 2 bioindicators plus IBIs for the ratio of calanoid copepods to the sum of cyclopoid copepods plus cladocerans. The separate sets of analyses were necessitated by the fact that the Virginia microzooplanton monitoring program began only in 1992. The additional two bioindicators employed in the Set 3 calculations were considered to be unique to the tidal fresh regime, so an additional analysis employing these metrics was considered to be warranted.

The threshold values for the IBI rating system are presented in Appendix . The values are for each salinity regime-season combination. For bioindicators which are considered to be indicative of "nonstressed" conditions, the minimum value of the reference data (MIN; the threshold between the ratings of 0 and 1), the 5-percentile of the reference data (5-P; the threshold between the ratings of 1 and 3), and the median of the reference data (the threshold between the ratings of 3 and 5) are presented. For bioindicators indicative of "stressed" conditions, the maximum value of the reference data (MAX; the threshold between the ratings of 0 and 1), the 95-percentile of the reference data (95-P; the threshold between the ratings of 1 and 3), and the median of the reference data (the threshold between the ratings of 3 and 5) are presented. The threshold values are given in the same units defined in Table 5.

Graphical Assessment of IBIs. The mean IBI scores and the IBI values of bioindicators for each of the sites are presented in Figures 42 - 46. A second series of IBI calculations were made for the summer season only (Figures 47 - 51). The mean IBI of a reference site should be between 3.5 and 4.0, assuming a normal distribution of values. If one assumes that a value of 3.5 is an approximate dividing point between "nonstressed" and "stressed" conditions, one can qualitatively evaluate the IBI figures.

For the tidal fresh regime, the "nonstressed" site TF5.5 is slightly below the expected mean for the year-round IBI calculation, while both the TF5.5 and the TF3.3-F sites are below expected values for the summer IBI calculations (Figure 46A and 51A). On the other hand, TF1.5-F and TF2.3-F are "stressed" sites for which the average IBI scores suggest "nonstressed" conditions (Figure 46A and 51A). The mean IBI scores for the "nonstressed" site ET5.1 and the "stressed" site CB1.1 appear to reflect their *a priori* classifications (Figures 46A and 51A).

For the oligohaline regime, the mean IBI scores for the "nonstressed" sites were as expected (Figure 46B and 51B). However, the "stressed" sites RET2.2 and TF1.5-O had IBIs more indicative of "nonstressed" conditions (Figures 46B and 51B). Therefore, there appears to be a problem with the IBI assessment for this region (see below).

For the mesohaline regime, the mean IBI scores for all of the "nonstressed" sites and the "stressed" site WT5.1 appear to have followed their classifications (Figure 46C and 51C). Unclassified sites ("unknowns") ET5.2 and LE1.1 appeared to have IBI scores that were more towards the "stressed" category, particularly in the summer assessment (Figure 46C and 51C). The remaining "unknown" mesohaline sites had higher IBIs, more representative of "nonstressed" conditions.

The pattern of mean IBI scores for the polyhaline region appeared to be most definitive of all. As expected, the "nonstressed" sites had higher IBIs, while the "stressed" sites from the Elizabeth River had the lowest IBIs of all sites (Figure 46D and 51D). It should be noted that the two Elizabeth River sites are the only ones for which a definitively stressed condition (due to toxics) could be supported by a substantial historical data base.

Statistical Assessments of IBIs. Table 7 presents a summary of the results of the canonical and classification discriminant analyses of the IBI scores. For each data set analyzed, the following types of information are summarized: 1) the overall statistical separation of "nonstressed" and "stressed" sites (CANDISC results); 2) results of univariate tests of differences between the IBI scores for each bioindicator for "nonstressed" and "stressed" conditions (CANDISC results); 3) the percent correct classification of the "nonstressed" and "stressed" samples (DISCRIM results); 4) a presentation of sites for which a large percentage misclassification occurred (DISCRIM results); and 5) the correlation between the IBI scores for the bioindicators[5]. The correlation information is useful in determining whether each bioindicator is "telling the same story" about the conditions indicated by samples taken from "nonstressed" and "stressed" sites. The discussion of the statistical results will follow the format of Table 7, examining the results of each set and salinity regime.

Set 1 - Tidal Fresh

The IBI components (mesozooplankton abundance, biomass, diversity, and variability) for the tidal fresh regime provided a highly significant ($p<0.0001$) discrimination between the "nonstressed" and "stressed" sites. However, only the mesozooplankton abundance and biomass IBIs were significantly different between the two conditions. Nearly 80% of the samples from the "stressed" sites were correctly classified by the discriminant analysis, but less than half (47%) of the "nonstressed" samples were correctly classified as to condition. The samples from site TF5.5 were misclassified as "stressed" most (88%) of the time, indicating that it should not have been used as a reference site for IBI score calculations. It is possible that high river flow conditions may produce an apparent stress on zooplankton communities at this site (see *Spatial and Temporal Limitations of Zooplankton Environmental Indicators* (above)). Alternately, the declining trends in water quality that were noted by Alden *et al.* (1991, 1992) may have placed the zooplankton communities at this site under stress. In the future, flow effects can be taken into account statistically prior to IBI calculations, if investigators decide to maintain TF5.5 as a reference site.

As one would expect, there was a strong correlation ($r=.84$) between the IBI scores for mesozooplankton abundance and biomass. However, IBI scores for mesozooplankton diversity was slightly negatively correlated ($r \approx -.2$) with both of these bioindicators. This relationship could be due to the observation that sites with high mesozooplankton densities (presumably a "good" characteristic according to the initial bioindicator scheme) are often dominated by a relatively few species, producing lower Shannon-Wiener diversities (presumably a "bad" characteristic according to the initial bioindicator scheme) due to lower evenness. Due to this relationship and the lack of apparent discrimination power by diversity IBI scores, future efforts should consider whether this bioindicator (as currently calculated[2]) is appropriate for the tidal fresh regime.

Set 1 - Oligohaline

For this set of analyses, which were based only upon mesozooplankton data, the "stressed" site (WT5.1) displayed IBI values that were significantly higher ($p<0.0001$ for overall discrimination; and for IBIs for mesozooplankton abundance, biomass and variability) than the "nonstressed" sites, although the mean values for all components from both categories were greater than 3 (see Figures 43A-43D; and 48A-48D). Thus, the differences between (and validity of) the *a priori* groupings used to calculate the IBIs do not appear to have been accurate (*i.e.* there may not be clear-cut "stressed" and "nonstressed" conditions for this salinity regime from the perspective of the

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[5] Since the "direction" of the values of the bioindicators relative to stress was accounted for in the calculation of the IBIs, the correlations, if any, should be positive.

[2] This index had to be calculated by blocking for state as well as salinity regime and season. Thus, relatively few observations were available for the IBI calculations, perhaps resulting in the observed low discrimination power and/or conflicting patterns.

mesozooplankton community).

Set 1 - Mesohaline

There was no significant discrimination between "nonstressed" and "stressed" samples for the Mesohaline regime. All sites were classified as "nonstressed" by the classification analysis. Therefore, the *a priori* categorization of sites did not appear to be accurate, at least from the perspective of the IBIs for the mesozooplankton bioindicators.

Set 1 - Polyhaline

There was a highly significant difference between the samples from the "nonstressed" and "stressed" sites ($p < 0.0001$, both overall and for univariate comparisons of the IBI values for each of the mesozooplankton bioindicators) from the polyhaline regime. Nearly all samples were correctly classified, as to whether they were collected from "nonstressed" (100% correct classification) or "stressed" (Elizabeth River; 98% correct classification) sites. Most of the IBI components were positively correlated to each other, indicating concordance.

Set 2 - Tidal Fresh (MD only)

There was a highly significant ($p < 0.0001$) difference between the IBI scores for the "nonstressed" and "stressed" sites in Maryland. The IBI scores for mesozooplankton abundance and biomass and the microzooplankton to mesozooplankton ratio were also highly significant ($p < 0.0001$) between the two categories. However, the IBIs for microzooplankton abundance and biomass from the "stressed" sites were significantly greater than for the "nonstressed" sites. Not surprisingly, the IBIs for mesozooplankton abundance and biomass, and the microzooplankton to mesozooplankton ratio were highly correlated with each other, but negatively correlated with the microzooplankton abundance and biomass IBIs. Therefore, it appears that microzooplankton abundance and biomass in the samples from this regime do not display the expected relationship. It is possible that, as with many biological communities, too little or too great densities for this community could signal stressful conditions (hence the bell-shaped curves displayed in most ecological text books discussing the biological effects of environmental gradients). In the tidal fresh regime, it appears that greater microzooplankton densities could be indicative of less stressful conditions. Nonetheless, the *relationship* between microzooplankton and mesozooplankton densities does appear to provide a significant indicator of stress in this salinity regime.

While most (96%) of the samples from the "stressed" sites were correctly classified, only half of the "nonstressed" samples were correctly classified. Since half of the samples from ET5.1 were classified as "stressed", the use of this site as a reference should be evaluated in the future. On the other hand, the high mean IBIs from some of the "stressed" sites (TF1.5-F, TF2.3-F; see Figures ??? and ???) could have confounded the classification process, as could the inclusion to two bioindicators (microzooplankton abundance and biomass) that did not perform as expected in the discrimination between the conditions. These effects will also have to be evaluated in the future in order to optimize the selection of an appropriate reference site(s) for the IBI calculations.

Set 2 - Oligohaline (MD only)

There was not a overall significant ($\alpha=0.001$) discrimination between samples from "nonstressed" and "stressed" sites based upon the mesozooplankton or microzooplankton IBIs. However, as discussed previously, this lack of discrimination capacity could be due to a lack of clear-cut stress conditions within this regime.

Set 2 - Mesohaline (MD only)

Unlike the Set 1 analysis for mesohaline sites, the overall discrimination for the Set 2 (Maryland sites) mesohaline region IBI assessment was highly significant ($p < 0.0001$). The univariate statistical comparisons were significant ($\alpha = 0.001$) for mesozooplankton (abundance & biomass), microzooplankton (abundance & biomass), as well as the microzooplankton to mesozooplankton abundance ratio. The IBIs for these bioindicators were significantly correlated with each other, indicating concordance. Mesozooplankton diversity and variability IBIs were not significantly different between "nonstressed" and "stressed" samples.

Classification analysis indicated that 76% of the "nonstressed" samples were correctly classified and 64% of "stressed" samples were correctly classified. Approximately one-quarter of samples from site CB4.3C were misclassified as "stressed" and one-third of samples from WT5.1 were misclassified as "nonstressed".

Set 3 - Tidal Fresh (MD only)

There was a highly significant ($p < 0.0001$) difference between the IBI scores for the "nonstressed" and "stressed" samples. The IBIs for mesozooplankton abundance, mesozooplankton biomass, microzooplankton to mesozooplankton abundance ratio, and the ratio of calanoid copepods to cyclopoid copepods plus cladocerans were all highly significantly ($p < 0.0001$) different between the two categories of sites. The IBIs for all of these bioindicators were positively correlated with each other, suggesting good concordance. The IBIs for mesozooplankton variability was only marginally significant ($p = 0.013$). The IBIs for mesozooplankton diversity and percent *Bosmina spp.* were not significantly different between the categories. As with the Set 2 assessment of data from the tidal fresh regime, the IBIs for microzooplankton densities (abundance and biomass) were higher in the "stressed" than in the "nonstressed" samples. This pattern again suggests that, for the tidal fresh regime, the **absolute** densities of microzooplankton may not be directly related to the degree of stress (in fact, as currently calculated, these bioindicators may be inversely related to the degree of stress), but that the **relative** relationship between microzooplankton and mesozooplankton densities may be indicative of stress.

The classification assessment of this data set indicated that 68% of "nonstressed" samples were correctly classified and 86% of "stressed" samples were correctly classified. Nearly one-third of samples from ET5.1 were misclassified as "stressed".

Summary and Conclusions

Despite the limited time and resources available for the development and evaluation of the IBI system for the assessment and summarization of the health of Chesapeake Bay zooplankton communities, the results of the preliminary IBI effort do appear to hold promise for future development/refinement efforts. The major conclusions can be summarized as follows:

Tidal Fresh Regime

- The discriminant analyses of IBIs for bioindicators generally provided a good separation between samples from sites determined *a priori* to be "stressed" (particularly CB1.1) and "nonstressed" (particularly ET5.1);
    - However, most of the samples from site TF5.5 were misclassified as "stressed";
        - ✓ Perhaps due to river flow effects;
        - ✓ Perhaps due to declining trends in water quality;
    - On the other hand, high IBI scores for TF1.5-F and TF2.3-F suggest that they may not have been as "stressed" as their *a priori* classification would presume;
- The IBIs were significantly different between the two condition categories for many of the bioindicators:
    - Mesozooplankton abundance & biomass;
    - Ratio of Microzooplankton to Mesozooplankton abundance;
    - Ratio of Calanoid copepods to Cyclopoid copepods + Cladocerans;
    - Mesozooplankton variability was marginally significant;
- The IBI values for Mesozooplankton Diversity were not significantly different between "nonstressed" and "stressed" sites; In fact, the IBIs for this variable were inversely correlated with the IBIs of bioindicators that were sigificant;
    - Could be due to low numbers of observations available for IBI calculations for this bioindicator due to blocking scheme needed to address different taxonomic enumeration protocols between the two states;
    - Could be due to inappropriate nature of this bioindicator for this salinity regime;
- The IBI values for Percent *Bosmina spp.* were not significantly different between the two categories of samples, but they were correlated with some of the indicators that were significant;
- The Microzooplankton abundance and biomass IBI scores were greater for "stressed" samples than for "nonstressed" samples; These bioindicators may not be appropriate for this salinity regime (or perhaps their direction of effect should be reconsidered);

Oligohaline Regime

- There was no significant discrimination between "nonstressed" and "stressed" sites by the IBIs for the bioindicators;
    - ■ Could be due to misclassification in *a priori* categories;
    - ■ Could be due to lack of clear-cut differences in stress in this regime:
        - ✓ All sites could be "nonstressed" or "stressed";
        - ✓ Sites could represent various degrees of stress;

Mesohaline Regime

- There was a significant discrimination between the two categories of conditions for the Maryland data, but this difference was not apparent when the entire data set was analyzed;
    - ■ Could be due to the fewer bioindicators available Bay-wide (*i.e.* microzooplankton bioindicators may be important discriminators for this regime);
    - ■ Could be due to less clear-cut differences in relative stress among Virginia sites or when the conditions are viewed Bay-wide (due to greater biological variability associated with the larger spatial scale);
- The IBIs were significantly different between the two condition categories for some of the bioindicators:
    - ■ Mesozooplankton abundance & biomass;
    - ■ Microzooplankton abundance & biomass;
    - ■ Ratio of Microzooplankton to Mesozooplankton abundance;
- The IBI values for Mesozooplankton Diversity and Variability were not significantly different between "nonstressed" and "stressed" sites;

Polyhaline Regime

- The greatest discrimination between "stressed" and "nonstressed" conditions were observed for this regime;
- The IBIs were significantly different between "nonstressed" and "stressed" samples for all bioindicators;
    - ■ Mesozooplankton abundance & biomass;
    - ■ Mesozooplankton Diversity;

- ■ Mesozooplankton Variability;

- The dramatic differences observed between condition categories for this regime could be due to the particular appropriateness of the bioindicators employed, or due to the fact that the Elizabeth River sites are the most clear-cut and best documented examples of stressed conditions available in the Bay data set.

For the assessments in which the IBI approach did not appear to provide good discrimination between the samples taken from "nonstressed" and "stressed" conditions, a number of factors could explain the failure of the approach. These factors can be divided into two categories: those involving the sites selected for reference and "stressed" conditions; and those involving the bioindicators themselves. Factors involving conditon classification include: 1) either reference or "stressed" sites are misclassified in the *a priori* selection process; 2) differences between reference and "stressed" conditions are not clear-cut for the particular salinity regime being assessed; 3) other physical factors (*e.g.* river flow, salinity shifts) may confound the perceived condition of reference sites; 4) the site(s) selected to represent "stressed" conditions may only be seasonally stressed ; and 5) the blocking of data by salinity regime, season and by state may provide an inadequate benchmark data base for IBI calculations when only a few sites are classified as "nonstressed" or "stressed" (as was suggested for Diversity). The biological factors include: 1) the bioindicator is not an appropriate indicator of stress; 2) the bioindicator is not appropriate for the salinity regime being assessed; 3) the bioindicator is not appropriate for all seasons; and 4) the response of the bioindicator to stress may not be monotonic (*i.e.* too great or too low a value for the bioindicator may indicate stress), and this relationship may be region or stress specific.

Recommendations

The IBI approach should be refined in the future, taking into account the findings of this preliminary assessment, the factors that may lead to poor discrimination, as well as new candidates for bioindicators which may arise in the future. Clearly, the classification of "nonstressed" and "stressed" sites needs to be re-evaluated in certain of the regimes (if, indeed, certain regimes monitored by the Bay program even have clear-cut examples of both conditions). Additional data sets from other ecosystems may need to be assessed to determine whether reference or "stressed" conditions from the Bay data set are adequate for the purpose of IBI calculation/confirmation. Such issues as seasonality of stress, the effects of non-anthropogenic factors (and mathematical means of "correcting" for these effects), and the comparability of data sets will have to be evaluated.

Likewise, the meaning, calculation procedures, and use of each of the bioindicators should be re-evaluated in light of the findings. It may be that certain bioindicators "work" perfectly well for certain salinity regime/season combinations and not for others. New bioindicators may need to be developed to make use of the large taxonomic data base that is available in order to characterize the functional biological roles of certain groups of the zooplankton (*e.g.* feeding guilds; trophic groups; opportunistic versus equilibrium species; age structure; *etc.* - information used by certain of the other IBI systems). In addition to refinement of individual bioindicators, the limitations of the IBI systems should be explored further by sensitivity, power and robustness analyses to determine how much of a change is necessary in the biological data to make a "nonstressed" site be classified as a "stressed" site, and which of the bioindicators (or combinations) are most sensitive to such a change. Further along, similar analyses should be conducted to determine the sensitivity of the IBI systems to detect long term trends in the biological condition of various regions of the Bay.

Literature Cited

[Unavailable at this time]

Table 5. Bioindicators employed in IBI calculations (units in parameters). The stressed/nonstressed column indicates whether increasing values for the bioindicator are believed to be indicative of stressed (S) or nonstressed (N) conditions. The regions column indicates the areas (salinity, regimes and state) for which the bioindicator is appropriate and the data are available.

Functional Category	Bioindicator	Stressed/ Nonstressed	Region
Zooplankton Abundance and Biomass	1) Mesozooplankton Abundance (#/m^3)	N	All
	2) Mesozooplankton biomass (g/m^3)	N	All
	3) Microzooplankton abundance (#/m^3)	S	MD only[1]
	4) Microzooplankton biomass (g/m^3)	S	MD only[1]
Zooplankton Diversity	5) Shannon-Wiener	N	All; By state[2]
Trophic Composition	6) Ratio of microzooplankton abundance to mesozooplankton abundance (decimal percent)	S	MD only[1]
Stability of Community (i.e. tendency towards "bloom and bust" cycles)	7) Deviation from median abundance for mesozooplankton (absolute value of: $\frac{\text{density - median density}}{\text{median density}}$, expressed as decimal percent)	S	All
Community Structure Indicators of Stress	8) Ratio of calanoid copepods to cyclopoid copepods plus cladocerans (decimal percent)	N	Tidal Fresh
	9) % of zooplankton abundance represented by *Bosmina spp.* (decimal percent)	S	Tidal Fresh

Footnotes:

1) VA began microzooplankton monitoring in 1992.

2) Data blocked by State as well as by salinity regime and season.

Table 6. Sites believed *a priori* to represent "stressed" and "nonstressed" conditions for IBI evaluation by discriminant analyses. The analyses were conducted on three data sets: Set 1 for both Virginia and Maryland data, using bioindicators common to both; Set 2 for Maryland data only; and Set 3 for Maryland tidal fresh water regimes only (see Table c. for details). (* = A suffix on the site designation following a hyphen indicates that the site fluctuates between salinity regimes: fresh (F), oligohaline (O), or mesohaline (M). The samples from the sites were divided according to their salinities.)

Set	Salinity Regime	Proposed Condition	Sites*	Comments
Set 1	Tidal Fresh	Nonstressed	ET5.1	
			TF3.3-F	
			TF5.5	Originally believed to be "nonstressed" but discriminant analysis indicated otherwise (Table c.). Perhaps if effects of riverflow were taken into account, apparent "stress" would diminish (see text).
		Stressed	CB1.1	Turbidity and possibly flow stressed.
			TF1.5-F	Eutrophic conditions suggested by phytoplankton data, but nearly half of samples were misclassified as "nonstressed" (Table c.).
			TF2.3-F	Eutrophic conditions and cyanobacteria blooms

Table 6. (cont.)

Set 1 (Cont.)	Mesohaline	Nonstressed	TF1.7	
			TF3.3-O	
			RET5.2	
		Stressed	RET2.2	Turbidity stress and cyanobacteria blooms
			TF1.5-O	Eutrophic conditions and cyanobacteria
	Polyhaline	Nonstressed	CB4.3C	
			LE3.6	
			RET3.1	
			TF3.3M	
		Stressed	WT5.1	Dinoflagellate blooms; eutrophic conditions
		Nonstressed	CB7.3E	
			CB7.4	
			LE5.5	
	Tidal Fresh	Stressed	SBE2	Stress of toxics
			SBE5	Stress of toxics
Set 2	Oligohaline	Nonstressed	ET5.1	
		Stressed	CB1.1	See above
			TF1.5-F	See above
			TF2.3-F	See above
	Mesohaline	Nonstressed	TF1.7	
		Stressed	RET2.2	See above
			TF1.5-O	See above
		Nonstressed	CB4.3C	
		Stressed	WT5.1	See above
Oligohaline				

Table 6. (cont.)

Set 3	Tidal Fresh	Nonstressed	ET5.1	
		Stressed	CB1.1	See above
			TF1.5-F	See above
			TF2.3-F	See above

Table 7. Summary of Results of IBI comparisons of "nonstressed" versus "stressed" sites by Canonical (CANDISC) and Classification (DISCRIM) discriminant analyses. The analyses were conducted on three sets of data: Set 1 was Maryland and Virginia data for tidal fresh, oligohaline, mesohaline, and polyhaline (VA only) regimes employing IBIs for four mesozooplankton bioindicators (mesozooplankton diversity, abundance, biomass, and variability); Set 2 was Maryland data for tidal fresh, oligohaline, and mesohaline regimes employing IBIs for the previous bioindicators plus IBIs for microzooplankton bioindicators (microzooplankton abundance and biomass; ratio of microzooplankton to mesozooplankton abundance); and Set 3 was Maryland data for tidal fresh sites employing the Set 2 data plus IBIs for the ratio of calanoid copepods to the sum of cyclopoid copepods and cladocerans, and the percent *Bosmina spp*. abundance of the total mesozooplankton counts.

Region	CANDISC Statistical Comparisons	DISCRIM Classification of Sites	Misclassified Sites in DISCRIM	Correlations Between IBI Scores of Components
SET 1:				
MD & VA Tidal Fresh	● Overall $p<0.0001$ ● For mesozooplankton abundance and biomass IBIs $p<0.0001$ ● IBIs for diversity and variability were not significant	● Only 47% of "non-stressed" correctly classified ● 78% of "stressed" correctly classified ● 69% of "unknowns" classified as "stressed"	● 88% of TF5.5 samples misclassified as "stressed" ● 46% of TF1.5 samples misclassified as "non-stressed"	● Mesozooplankton abundance IBI correlated with biomass ($r=0.84$) ● Diversity IBI slightly inversely correlated with IBIs for abundance and bio-mass ($r=-0.22$; -0.20)
MD & VA Oligohaline	● Overall $p<0.0001$ **BUT** ● IBIs for mesozooplankton abundance, biomass and variability were higher for "stressed" stations! ● IBI for diversity was not significant	● 83% of "nonstressed" correctly classified ● Only 47% of "stressed" correctly classified ● 85% of "unknowns" classified as "non-stressed" ● **BUT** considering CANDISC results, all classifications may be meaningless		● Mesozooplankton abundance IBI correlated with mesozooplankton biomass IBI ($r=0.79$) ● Variability IBI slightly inversely correlated with mesozooplankton biomass IBI ($r=-0.18$)

Table 7. (cont.)

SET 1: (Cont.)

MD & VA Mesohaline	• Not significant $p=0.41$ • None of the IBI components were significant	• All samples classified as "nonstressed" (no discrimination between "conditions")	N/A	• Mesozooplankton abundance IBI correlated with mesozooplankton biomass IBI ($r=0.66$) • Mesozooplankton variability IBI slightly inversely correlated with mesozooplankton abundance and biomass IBI ($r=-0.21$; -0.14)
VA only Polyhaline	• Overall $p<0.0001$ • For all IBI components (diversity, abundance, biomass; variability) $p<0.0001$	• 100% of "nonstressed" correctly classified • 97.7% of "stressed" correctly classified	N/A	• Mesozooplankton IBI correlated with meso-zooplankton biomass IBI ($r=0.72$) • Mesozooplankton variability IBI correlated with meso-zooplankton diversity, abundance, and biomass IBIs ($r=0.29$; 0.36; 0.32)

Table 7. (cont.)

SET 2:

MD only Tidal Fresh	• Overall $p<0.0001$ • For mesozooplankton abundance, biomass, and microzooplankton to mesozooplankton ratio IBIs $p<0.0001$ • IBI for diversity was not significant • For microzooplankton abundance and biomass IBIs, means of "stressed" sites were significantly higher than "nonstressed" sites ($p<0.0001$; 0.010)!	• Only 49% of non-stressed" correctly classified • 96% of "stressed" correctly classified • 85% of "unknowns" were classified as "stressed"	• 51% of ET5.1 samples misclassified as "stressed"	• Mesozooplankton abundance IBI correlated with meso-zooplankton biomass IBI ($r=0.85$) • Microzooplankton abundance IBI correlated with microzooplankton biomass IBI ($r=0.80$) • Mesozooplankton diversity IBI was inversely correlated with mesozooplankton abundance and biomass IBIs ($r=-0.25$; -0.23); as well as with the microzooplankton to mesozooplankton ratio IBI ($r=-0.24$) • IBIs for mesozooplankton abundance and biomass were inversely correlated with microzooplankton abundance and biomass IBIs (r values -0.23 to -0.29)

Table 7. (cont.)

<u>SET 2</u>: (Cont.)

MD only Tidal Fresh (Cont.)				• IBIs for mesozooplankton abundance and biomass were correlated with the microzooplankton to mesozooplankton ratio IBI (r=0.73; 0.69)
MD only Oligohaline	• Overall not significant @ 0.001 • For microzooplankton biomass IBI, means of "stressed" sites were significantly higher than "nonstressed" sites (p=0.0008)! • IBIs for other 6 components not significant	• Only 55% of "non-stressed" were correctly classified • 79% of "stressed" were correctly classified • 80% of "unknowns" were classified as "stressed" • **<u>BUT</u>** considering non-significant overall CANDISC statistics and reversal of patterns for micro-zooplankton biomass IBI, these results may be meaningless	• 45% of TF1.7 samples misclassified as "stressed"	• Mesozooplankton abundance IBI correlated with meso-zooplankton biomass IBI (r=0.72) • Microzooplankton abundance IBI correlated with microzooplankton biomass IBI (r=0.66) • Microzooplankton to mesozooplankton ratio IBI correlated to mesozooplankton abundance IBI (r=0.55), meso-zooplankton biomass IBI (r=0.48), micro-zooplankton abun- dance IBI (r=0.38), & microzooplankton biomass IBI (r=0.33); & inversely correlated

Table 7. (cont.)

SET 2: (Cont.)

MD only Oligohaline (Cont.)				with mesozooplankton variability IBI (r=-0.20) ● Mesozooplankton biomass IBI inversely correlated with mesozooplankton variability IBI (r=-0.23)
MD only Mesohaline	● Overall p<0.0001 ● For microzooplankton abundance and biomass IBIs p<0.0001 ● For microzooplankton to mesozooplankton ratio IBI p<0.0001 ● For mesozooplankton abundance IBI p=0.008 ● For mesozooplankton biomass IBI p=0.003 ● IBIs for mesozooplankton diversity and variability were not significant	● 76% of "nonstressed" correctly classified ● 64% of "stressed" correctly classified ● 61% of "unknowns" classified as "nonstressed"	● 24% of CB4.3C samples misclassified as "stressed" ● 36% of WT5.1 samples misclassified as "nonstressed"	● Mesozooplankton abundance IBI correlated with mesozooplankton biomass IBI (r=0.60) ● Microzooplankton abundance IBI correlated with microzooplankton biomass IBI (r=0.71) ● Microzooplankton to mesozooplankton ratio IBI correlated with mesozooplankton abundance and biomass IBIs (r=0.56; 0.45); and microzoo-plankton abundance and biomass IBIs (r=0.49; 0.34)

Table 7. (cont.)

<u>SET 3</u>:

MD only Tidal Fresh only	• Overall p<0.0001 • For mesozooplankton abundance & biomass, microzooplankton to mesozooplankton ratio IBI, ratio of calanoids to cyclopoids + cladocerans IBI p<0.0001 • For mesozooplankton variability IBI p=0.013 • IBIs for mesozooplankton diversity and percent *Bosmina* were not significant • For microzooplankton abundance & biomass IBIs, means of "stressed" sites were significantly higher than "nonstressed" sites (p=0.0002; 0.035)!	• 68% of "nonstressed" correctly classified • 86% of "stressed" correctly classified • 68% of "unknowns" classified as "stressed"	• 32% of ET5.1 samples misclassified as "stressed"	• See Set 2, "MD only Tidal Fresh" for all correlations except those involving the ratio of calanoids to cyclopoids + cladocerans and percent *Bosmina* IBIs. • Ratio of calanoids to cyclopoids + cladocerans IBI correlated with mesozooplankton abundance & biomass IBIs (r=0.46; 0.45), microzooplankton to mesozooplankton ratio IBI (r=0.42), and percent *Bosmina* IBI (r=0.58)

G) ratio of microzooplankton to mesozooplankton biomass.

Figure 50.	Mean summer IBI scores of samples from polyhaline sites for individual bioindicators: A) mesozooplankton abundance; B) mesozooplankton biomass; C) variability in mesozooplankton abundance; D) species diversity by Shannon-Wiener index.
Figure 51.	Mean summer total IBI scores of samples from major salinity regimes: A) tidal fresh sites; B) oligohaline sites; C) mesohaline sites; and D) polyhaline sites.

[The figures were unavailable at time of printing]

Utilization and future development of zooplankton indicators for Chesapeake Bay

by Steven E. Bieber, Fredrick Hoffman, and Kent Mountford

[THIS CHAPTER IS NOT AVAILABLE AT THIS TIME]

Appendices

This report includes three Appendices:

Appendix A: Literature Search

Appendix B: Zooplankton Species Codes Used in the Chesapeake Bay, Version 1.0

Appendix C: SAS Results for Multiple Regression Analyses

Appendices A and B are attached. Appendix C is available upon request. Contact Claire Buchanan at the Interstate Commission on the Potomac River Basin.

APPENDIX A: LITERATURE SEARCH

By:

K.G. Sellner, D. Nicholson and L. Baldwin

A search of the literature was undertaken through DIALOG, a electronic abstracting service. The main goal of the search was to find all relevant citations between zooplankton and water quality and between zooplankton and phytoplankton that might be related to water quality. One of the PI's (KGS) then sorted the citations into five topics:

1) General Zooplankton Ecology and Physiology
2) Bottom-Up Controls of Zooplankton
3) Top-Down Controls on Zooplankton
4) Microzooplankton-Water Quality Interactions
5) Mesozooplankton-Water Quality Interactions

All citations were distributed to all PI's involved in the Bioindicator Project. Citations for each of these topics is presented in the following appendix.

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APPENDIX B: ZOOPLANKTON SPECIES CODES USED IN THE CHESAPEAKE BAY, Version 1.0

By:

Claire Buchanan

ZOOPLANKTON SPECIES CODES USED IN THE CHESAPEAKE BAY, Version 1.0
February 1993

The species list in the ZOOCODES file was compiled by the Interstate Commission on the Potomac River Basin (ICPRB) over several years from different data sets, primarily from the Potomac estuary. This version was produced in February, 1993. The file contains the NOAA, Virginia and Maryland species codes for zooplankton found in Chesapeake Bay. The NOAA codes have been modified so as to reflect life stages of the organism. The suffix format for life stage outlined in the Chesapeake Bay Program Living Resources Data Management Plan was followed, and the suffixes are explained in the key below.

Key:

DB = database. Z refers to the zooplankton master list compiled by the Interstate Commission on the Potomac River Basin (ICPRB); F, fish master list; B, benthos master list.

NODC_CODE: Species codes being developed by the National Oceanographic Data Center of NOAA, modified by a suffix to indicate life stage. Species for which codes have not been completely developed have been given a temporary code, typically an alphanumeric (e.g. 61090701RECTANG for Alona rectangula). The suffixes are:

_11	nauplii
_12	copepodite
_98	adult
_00	viable egg
_97	larvae
_04	young-of-year
_17	cypris
_31	zoea
_33	megalopa
_30	protozoea

For a complete listing, refer to the Chesapeake Bay Program Living Resources Data Management Plan. The suffix "_S" indicates the species name has been superceded by another name which may or may not have the same NODC code.

VA_CODE: Virginia Chesapeake Bay Program zooplankton codes obtained from Dr. Ray Birdsong, Old Dominion University, February 1993.

MD_CODE: Maryland Chesapeake Bay Program zooplankton codes obtained from Mr. William Burton, Versar, Inc., February 1993.

Filename: ZOOCODES (Version 1.0)
(Compiled by the Interstate Comm. on the Potomac River Basin, February, 1993)

DB	SPECIES_NAME	NODC_CODE	VA_CODE	MD_CODE
Z	ACANTHOHAUSTORIUS_MILLSI	6169220602		
Z	ACARINA	5922		
Z	ACARTIA	61182901	260	T2360
Z	ACARTIA_(ADULT)	61182901_98		
Z	ACARTIA_(COPEPODITE)	61182901_12	297	
Z	ACARTIA_(NAUPLII)	61182901_11		
Z	ACARTIA_CLAUSI	6118290101		T704
Z	ACARTIA_HUDSONICA	6118290113	76	
Z	ACARTIA_HUDSONICA_(ADULT)	6118290113_98		
Z	ACARTIA_HUDSONICA_(COPEPODITE)	6118290113_12		
Z	ACARTIA_HUDSONICA_(NAUPLII)	6118290113_11		
Z	ACARTIA_LONGIREMUS	6118290103	77	
Z	ACARTIA_LONGIREMUS_(ADULT)	6118290103_98		
Z	ACARTIA_LONGIREMUS_(COPEPODITE)	6118290103_12		
Z	ACARTIA_TONSA	6118290104	78	T703
Z	ACARTIA_TONSA_(ADULT)	6118290104_98		
Z	ACARTIA_TONSA_(COPEPODITE)	6118290104_12		
Z	ACARTIA_TONSA_(NAUPLII)	6118290104_11		
Z	ACETES_AMERICANUS	6177020101		
Z	ACETES_AMERICANUS_(ZOEA)	6177020101_31	93	
Z	ACINETA	35340101		
Z	ACRICOTOPUS	65051029		
Z	AEGATHOA_MEDIALIS	6161060102	407	
Z	AEGATHOA_OCULATA	61610601OCULATA		
Z	AGABUS	63050602		
Z	AGLANTHA_DIGITALE	3711041101	16	
Z	ALONA	61090701	402	T608+T1160
Z	ALONA_AFFINIS	6109070103		T2368
Z	ALONA_COSTATA	6109070107		T2308
Z	ALONA_DIAPHANA	61090701DIAPHAN		
Z	ALONA_GUTTATA	61090701GUTTATA	401	
Z	ALONA_MONACANTHA	6109070111		
Z	ALONA_QUADRANGULARIS	61090701_QUADRA	403	
Z	ALONA_RECTANGULA	61090701RECTANG		
Z	ALONA_RUSTICA	61090701RUSTICA		
Z	ALONELLA	61090708	305	T609
Z	ALONELLA_ACUTIROSTRIS	61090708ACUTIRO		
Z	ALONELLA_ROSTRATA	6109070802	369	
Z	ALONOPSIS	61090717	351	
Z	ALOSA_(LARVAE)	87470101_97	374	
Z	ALOSA_(VIABLE_EGG)	87470101_00		
Z	ALOSA_AESTIVALIS_(LARVAE)	8747010102_97		
Z	ALOSA_AESTIVALIS_(VIABLE_EGG)	8747010102_00		
Z	ALOSA_MEDIOCRIS_(LARVAE)	8747010103_97	376	
Z	ALOSA_MEDIOCRIS_(VIABLE_EGG)	8747010103_00		
Z	ALOSA_PSEUDOHARENGUS_(LARVAE)	8747010105_97	377	
Z	ALOSA_PSEUDOHARENGUS_(VIABLE_EGG)	8747010105_00		
Z	ALOSA_SAPIDISSIMA_(LARVAE)	8747010101_97		
Z	ALOSA_SAPIDISSIMA_(VIABLE_EGG)	8747010101_00		
Z	ALPHEIDAE	617914		
Z	ALPHEIDAE_(ZOEA)	617914_31	99	
Z	ALPHEUS_HETEROCHAELIS	6179140101		
Z	ALPHEUS_HETEROCHAELIS_(ZOEA)	6179140101_31	293	
Z	ALPHEUS_NORMANNI	6179140102		
Z	ALPHEUS_NORMANNI_(ZOEA)	6179140102_31	194	
Z	ALTEUTHA_DEPRESSA	6119110102		T3093
Z	ALTEUTHA_DEPRESSA_(ADULT)	6119110102_98		
Z	ALTEUTHA_DEPRESSA_(COPEPODITE)	6119110102_12		
Z	ALTEUTHA_OBLONGA	6119110101	394	
Z	ALTEUTHA_OBLONGA_(ADULT)	6119110101_98		
Z	ALTEUTHA_OBLONGA_(COPEPODITE)	6119110101_12		
Z	AMMODYTES_AMERICANUS_(LARVAE)	8845010102_97	162	
Z	AMMODYTES_AMERICANUS_(VIABLE_EGG)	8845010102_00		
Z	AMPELISCA_ABDITA	6169020108		T1286
Z	AMPHARETIDAE	500167	26	
Z	AMPHIAGRON	AMPHIAGRON		
Z	AMPHIASCUS_PARVUS	6119280401		
Z	AMPHIASCUS_PARVUS_(ADULT)	6119280401_98		
Z	AMPHIASCUS_PARVUS_(COPEPODITE)	6119280401_12		
Z	AMPHINEMA_DINEMA	3703120701	9	

Z	AMPHIOXUS	85000103		T3080
Z	AMPHIPODA	6168	281	
Z	ANCHOA_(LARVAE)	87470202_97	197	
Z	ANCHOA_(VIABLE_EGG)	87470202_00		
Z	ANCHOA_HEPSETUS_(LARVAE)	8747020201_97	147	
Z	ANCHOA_HEPSETUS_(VIABLE_EGG)	8747020201_00	277	
Z	ANCHOA_MITCHILLI_(LARVAE)	8747020202_97	148	
Z	ANCHOA_MITCHILLI_(VIABLE_EGG)	8747020202_00	149	
Z	ANGUILLA_ROSTRATA_(ELVER)	8741010101_97		
Z	ANISOGAMMARUS	61692101		
Z	ANISOPTERA	6224		T2215
Z	ANNELIDA	50		
Z	ANNELIDA_(TROCHOPHORE_LARVAE)	50_TROCHOPHORE	45	
Z	ANTHOMEDUSAE	3703	225	
Z	ANURAEOPSIS	45060108		
Z	ANURAEOPSIS_FISSA	4506010801		
Z	APOCYCLOPS	61200811		
Z	APOCYCLOPS_(ADULT)	61200811_98		
Z	APOCYCLOPS_(COPEPODITE)	61200811_12		
Z	ARACHNIDA	59		T580
Z	ARACHNIDIUM_FIBROSUM	A_FIBROSUM		
Z	ARCELLA	34420201		
Z	ARCIDAE	550601	47	
Z	ARENAEUS_CRIBRARIUS_(MEGALOPA)	6189010101_33		
Z	ARENAEUS_CRIBRARIUS_(ZOEA)	6189010101_31	112	
Z	ARGULUS	61290101	311	T700
Z	ARGULUS_ALOSAE	6129010101		
Z	ASABELLIDES_OCULATA	5001670802	25	
Z	ASCIDACEAN	8401	146	
Z	ASCOMORPHA	45060801		
Z	ASCOMORPHA_OVALIS	4506080101		
Z	ASCOMORPHA_SULTANS	4506080102		
Z	ASPLANCHNA	45061201		
Z	ASPLANCHNA_BRIGHTWELLI	4506120101		
Z	ASPLANCHNA_HERRICKI	4506120102		
Z	ASTEROIDEA	8100	140	
Z	ATHERINIDAE_(LARVAE)	880502_97	175	
Z	AURELIA_AURITA	3734030201	20	
Z	AUTOLYTUS	50012301	273	
Z	BAETIDAE	621602		
Z	BAETIS	62160215		
Z	BAIRDIELLA_CHRYSOURA_(EGG)	8835440301_00	155	
Z	BAIRDIELLA_CHRYSOURA_(LARVAE)	8835440301_97	176	
Z	BALANIDAE	613402	88	
Z	BALANIDAE_(CYPRIS)	613402_17	196	
Z	BALANIDAE_(NAUPLII)	613402_11	89	T2069
Z	BALANUS_(CYPRIS)	61340201_17		
Z	BALANUS_(NAUPLII)	61340201_11		
Z	BALANUS_AMPHITRITE	6134020112		
Z	BALANUS_AMPHITRITE_(CYPRIS)	6134020112_17		
Z	BALANUS_AMPHITRITE_(NAUPLII)	6134020112_11		
Z	BALANUS_BALANOIDES	6134020101		
Z	BALANUS_BALANOIDES_(CYPRIS)	6134020101_17		
Z	BALANUS_BALANOIDES_(NAUPLII)	6134020101_11		
Z	BALANUS_EBURNEUS	6134020113		
Z	BALANUS_EBURNEUS_(CYPRIS)	6134020113_17		
Z	BALANUS_EBURNEUS_(NAUPLII)	6134020113_11		
Z	BALANUS_IMPROVISUS	6134020114		
Z	BALANUS_IMPROVISUS_(CYPRIS)	6134020114_17		
Z	BALANUS_IMPROVISUS_(NAUPLII)	6134020114_11		
Z	BARENTSIA	BARENTSIA		
Z	BDELLOIDA	4504		
Z	BELOSTOMA	62720502		
Z	BEROE_OVATA	3807010103	22	
Z	BEZZIA	65050401		
Z	BIPINNARIA_(LARVAE)	BIPINNARIA_97	284	
Z	BITHYNIA	51031703		
Z	BITTACOMORPHA	65040201		
Z	BIVALVIA	55	177	
Z	BIVALVIA_(GLOCHIDIUM)	55_GLOCHIDIUM	410	
Z	BIVALVIA_(STRAIGHT-HINGE_VELIGER_LARVAE)	55_STRAIGHT-HIN		
Z	BIVALVIA_(TROCHOPHORE_LARVAE)	55_TROCHOPHORE		
Z	BIVALVIA_(UMBO_VELIGER_LARVAE)	55_UMBO_VELIGER		
Z	BOSMINA	61090301	235	T776
Z	BOSMINA_COREGONI_MARITIMA	6109030202MARIT	292	
Z	BOSMINA_LONGIROSTRIS	6109030101	301	T618

Z BOTHIDAE_(LARVAE)	885703_97		
Z BOUGAINVILLIA_RUGOSA	3703010707	7	
Z BOWMANIELLA_DISSIMILIS	6153012607	253	
Z BRACHIOLARIA_(LARVAE)	BRACHIOLARIA_97	283	
Z BRACHIONUS	45060104		
Z BRACHIONUS_ANGULARIS	45060104_ANGULA		
Z BRACHIONUS_BIDENTATA	4506010407		
Z BRACHIONUS_BUDAPESTINENSIS	4506010408		
Z BRACHIONUS_CALYCIFLORUS	4506010402	364	
Z BRACHIONUS_CAUDATUS	4506010409		
Z BRACHIONUS_DIVERSICORNIS	45060104DIVERSI		
Z BRACHIONUS_HAVANENSIS	4506010403		
Z BRACHIONUS_PLERODINOIDES	4506010404		
Z BRACHIONUS_PLICATILIS	4506010401		
Z BRACHIONUS_QUADRIDENTATA	4506010411		
Z BRACHIONUS_RUBENS	4506010412		
Z BRACHIONUS_URCELARIS	4506010405		
Z BRACHIONUS_VARIABILIS	4506010413		
Z BRACHYURA_(MEGALOPA)	6184_33		T2022
Z BRACHYURA_(ZOEA)	6184_31		T2023
Z BRANCHIOSTOMA_CARIBAEUM	8500010101	249	
Z BREVOORTIA_(LARVAE)	87470104_97		
Z BREVOORTIA_(VIABLE_EGG)	87470104_00		
Z BREVOORTIA_TYRANNUS_(LARVAE)	8747010401_97		
Z BREVOORTIA_TYRANNUS_(VIABLE_EGG)	8747010401_00	390	
Z BRILLIA	65050823		
Z BRYCHIUS	63050405		
Z BRYOCAMPTUS	61192901		
Z BRYOCAMPTUS_(ADULT)	61192901_98		
Z BRYOCAMPTUS_(COPEPODITE)	61192901_12		
Z BRYOCAMPTUS_ZSCHOKKEI	6119290103	330	
Z BRYOCAMPTUS_ZSCHOKKEI_(ADULT)	6119290103_98		
Z BRYOCAMPTUS_ZSCHOKKEI_(COPEPODITE)	6119290103_12		
Z CAENIS	62180202		
Z CALANOIDA	6118	264	T701
Z CALANOIDA_(ADULT)	6118_98		
Z CALANOIDA_(COPEPODITE)	6118_12		
Z CALANOIDA_(NAUPLII)	6118_11		
Z CALANUS	611801CALANUS	236	T3169
Z CALANUS_FINMARCHICUS	6118010208	396	
Z CALANUS_FINMARCHICUS_(ADULT)	6118010208_98		
Z CALANUS_FINMARCHICUS_(COPEPODITE)	6118010208_12		
Z CALANUS_HELGOLANDICUS	6118010210	356	
Z CALANUS_HELGOLANDICUS_(ADULT)	6118010210_98		
Z CALANUS_HELGOLANDICUS_(COPEPODITE)	6118010210_12		
Z CALIGUS	61230101	300	
Z CALLIANASSA	61830402		
Z CALLIANASSA_(ZOEA)	61830402_31	103	
Z CALLIANASSA_ATLANTICA	6183040205		
Z CALLIANASSA_ATLANTICA_(ZOEA)	6183040205_31		
Z CALLIANASSA_BIFORMIS	6183040209		
Z CALLIANASSA_BIFORMIS_(ZOEA)	6183040209_31		
Z CALLINECTES_SAPIDUS_(MEGALOPA)	6189010301_33	206	
Z CALLINECTES_SAPIDUS_(ZOEA)	6189010301_31	113	
Z CALYPTRAEIDAE	800829	55	
Z CAMPTOCERCUS_RECTIROSTRIS	6109071001		T620
Z CANCER_IRRORATUS_(MEGALOPA)	6188030108_33	186	
Z CANCER_IRRORATUS_(ZOEA)	6188030108_31	118	
Z CANTHOCAMPTUS	61192915	316	
Z CANTHOCAMPTUS_(ADULT)	61192915_98		
Z CANTHOCAMPTUS_(COPEPODITE)	61192915_12		
Z CANUELLA_ELONGATA	6119050201_S	241	
Z CANUELLA_ELONGATA_(ADULT)	6119050201_98_S		
Z CANUELLA_ELONGATA_(COPEPODITE)	6119050201_12_S		
Z CAPRELLA_GEOMETRICA	61710107GEOMETR	406	
Z CARINOGAMMARUS_MUCRONATUS	61692107CARINOG		
Z CARPIODES_CYPRINUS_(LARVAE)	8776040201_97		
Z CATOSTOMUS_COMMERSONI_(LARVAE)	8776040102_97		
Z CENTRARCHIDAE_(LARVAE)	883516_97		
Z CENTROPAGES	61181701	261	
Z CENTROPAGES_(ADULT)	61181701_98		
Z CENTROPAGES_(COPEPODITE)	61181701_12		
Z CENTROPAGES_FURCATUS	6118170104	70	T2620
Z CENTROPAGES_FURCATUS_(ADULT)	6118170104_98		
Z CENTROPAGES_FURCATUS_(COPEPODITE)	6118170104_12		
Z CENTROPAGES_HAMATUS	6118170105	71	T705

Z	CENTROPAGES_HAMATUS_(ADULT)	6118170105_98		
Z	CENTROPAGES_HAMATUS_(COPEPODITE)	6118170105_12		
Z	CENTROPAGES_HAMATUS_(NAUPLII)	6118170105_11		
Z	CENTROPAGES_TYPICUS	6118170106	72	T2207
Z	CENTROPAGES_TYPICUS_(ADULT)	6118170106_98		
Z	CENTROPAGES_TYPICUS_(COPEPODITE)	6118170106_12		
Z	CENTROPYXIDAE	344203		
Z	CENTROPYXIS_ACULEATA	3442030101		
Z	CEPHALODELLA	45060403		
Z	CERATOPOGONIDAE	CERATOPOGONIDAE		
Z	CERIODAPHNIA	61090204		
Z	CERIODAPHNIA_DUBIA	6109020407		
Z	CERIODAPHNIA_LACUSTRIS	6109020404		T622
Z	CERIODAPHNIA_PULCHELLA	61090204PULCHEL		
Z	CERIODAPHNIA_QUADRANGULA	6109020402		T2318
Z	CERIODAPHNIA_RETICULATA	6109020401	397	
Z	CERIODAPHNIA_ROTUNDA	61090204_ROTUND		
Z	CHAETODIPTERUS_FABER_(LARVAE)	8835520101_97	159	
Z	CHAETOGASTER	50090307		
Z	CHAETOGASTER_DIAPHANUS	5009030701		
Z	CHAETOGASTER_DIASTROPHUS	5009030703		
Z	CHAETOGNATHA	83	360	
Z	CHAMA_LACTUCA	5515510103		
Z	CHAOBORUS	65050301		T292
Z	CHAOBORUS_PUNCTIPENNIS	6505030102	334	
Z	CHASMODES_BOSQUIANUS_(LARVAE)	8842010301_97	172	
Z	CHEUMATOPSYCHE	64180402		
Z	CHIONE_CYPRIA	5515471514		
Z	CHIRIDOTEA_ALMYRA	6162020501		
Z	CHIRONOMIDAE	650508		
Z	CHIRONOMID_(LARVAE)	650508_97	244	T787
Z	CHIRONOMUS	65050833		
Z	CHIRONOMUS_ATTENUATUS	6505083302		
Z	CHIRONOMUS_STAEGERI	6505083318		
Z	CHLOROPERLIDAE	625404		
Z	CHRYSAORA_QUINQUECIRRHA	3734010203	18	
Z	CHRYSOPETALIDAE	500108	28	
Z	CHTHAMALUS_FRAGILIS	6134010102	87	
Z	CHTHAMALUS_FRAGILIS_(CYPRIS)	6134010102_17		
Z	CHTHAMALUS_FRAGILIS_(NAUPLII)	6134010102_11		
Z	CHYDORIDAE	610907		
Z	CHYDORUS	61090702	328	T629
Z	CHYDORUS_BICORNUTUS	61090702_BICORN		
Z	CHYDORUS_PIGER	61090702_PIGER		
Z	CHYDORUS_SPHAERICUS	6109070201		
Z	CILIOPHORA	3512		
Z	CIRRIPEDIA	6134		
Z	CLADOCERA	6109	279	
Z	CLETODES_LONGICAUDATUS	6119270101		
Z	CLETODES_LONGICAUDATUS_(ADULT)	6119270101_98		
Z	CLETODES_LONGICAUDATUS_(COPEPODITE)	6119270101_12		
Z	CLIMACOCYLIS	35400504		
Z	CLINOTANYPUS	65050801		
Z	CLIONE_LIMACINA	5125060101	184	
Z	CLUPEA_HARENGUS_HARENGUS_(LARVAE)	874701020102_97		
Z	CLUPEA_HARENGUS_HARENGUS_(VIABLE_EGG)	874701020102_00		
Z	CLUPEIDAE_(LARVAE)	874701_97	337	
Z	CLUPEIDAE_(VIABLE_EGG)	874701_00	375	
Z	CLYTEMNESTRA_ROSTRATA	6119120101	252	
Z	CLYTEMNESTRA_ROSTRATA_(ADULT)	6119120101_98		
Z	CLYTEMNESTRA_ROSTRATA_(COPEPODITE)	6119120101_12		
Z	CLYTIA_EDWARDSI	3704010502		
Z	CLYTIA_HEMISPHAERICA	3704010503		
Z	CLYTIA_LONGICYATHA	3704010509		
Z	CLYTIA_PAULENSIS	3704010505		
Z	CNIDARIA	37		
Z	CNIDARIA_ANEMONE	37ANEMONE	333	
Z	CNIDARIA_EPHYRA	37EPHYRA	282	
Z	CNIDARIA_MEDUSAE	37MEDUSAE	285	
Z	CNIDARIA_PLANULA_LARVAE	37PLANULA_LARVA		
Z	COELOTANYPUS	65050802		
Z	COELOTANYPUS_CONCINNUS	6505080201		
Z	COLEOPTERA	6302		T833
Z	COLLEMBOLA	6208		T489
Z	COLLOTHECA	45080101		
Z	COLLOTHECACEA	4508		

Z	COLLOTHECA_MUTABILIS	4508010101		
Z	COLLOTHECA_PELAGICA	4508010102		
Z	COLURELLA	45060103		
Z	CONCHAPELOPIA	65051003		
Z	CONOCHILOIDES	45070401		
Z	CONOCHILOIDES_DOSSUARIUS	4507040102		
Z	CONOCHILOIDES_NATANS	4507040103		
Z	CONOCHILUS	45070402		
Z	CONOCHILUS_HIPPOCREPIS	4507040201		
Z	CONOCHILUS_UNICORNIS	4507040202		
Z	COPEPODA	6117		
Z	COPEPOD_(ADULT)	6117_98		
Z	COPEPOD_(COPEPODITE)	6117_12	262	
Z	COPEPOD_(NAUPLII)	6117_11	318	T749
Z	CORBICULA_FLUMINEA	551450201		
Z	CORIXIDAE	627201		T834
Z	COROPHIUM	61691502	307	T634
Z	COROPHIUM_LACUSTRE	6169150205	144	T635
Z	CORYCAEUS	61200401	267	
Z	CORYCAEUS_(ADULT)	61200401_98		
Z	CORYCAEUS_(COPEPODITE)	61200401_12		
Z	CORYCAEUS_AMAZONICUS	6120040103	191	
Z	CORYCAEUS_AMAZONICUS_(ADULT)	6120040103_98		
Z	CORYCAEUS_AMAZONICUS_(COPEPODITE)	6120040103_12		
Z	CORYCAEUS_ELONGATUS	6120040104	223	
Z	CORYCAEUS_ELONGATUS_(ADULT)	6120040104_98		
Z	CORYCAEUS_ELONGATUS_(COPEPODITE)	6120040104_12		
Z	CORYCAEUS_SPECIOSUS	6120040109	171	
Z	CORYCAEUS_SPECIOSUS_(ADULT)	6120040109_98		
Z	CORYCAEUS_SPECIOSUS_(COPEPODITE)	6120040109_12		
Z	CORYCAEUS_VENUSTUS	6120040105	85	
Z	CORYCAEUS_VENUSTUS_(ADULT)	6120040105_98		
Z	CORYCAEUS_VENUSTUS_(COPEPODITE)	6120040105_12		
Z	CRANGON_SEPTEMSPINOSA	6179220103		
Z	CRANGON_SEPTEMSPINOSA_(ZOEA)	6179220103_31	102	
Z	CRASSOSTREA_VIRGINICA	5510020102	50	
Z	CRICOTOPUS	65050826		
Z	CRICOTOPUS_BICINCTUS	6505082604		
Z	CRUSTACEA	61	61	
Z	CRUSTACEAN_(EGG)	61_00	288	
Z	CRUSTACEAN_(MEGALOPA)	61_33	139	
Z	CRUSTACEAN_(NAUPLII)	61_11	344	
Z	CRUSTACEAN_(PROTOZOEA)	61_30	275	
Z	CRUSTACEAN_(ZOEA)	61_31		
Z	CRYPTOCHIRONOMUS	65050836		
Z	CRYPTOCHIRONOMUS_FULVUS	6505083602		
Z	CRYPTOPHORIDA	3517		
Z	CTENOPHORA	38	254	
Z	CUMACEA	6154	195	T2024
Z	CUNINA_OCTONARIA	3712030302	17	
Z	CYANEA_CAPILLATA	3734020101	19	
Z	CYATHURA_POLITA	6160010201		T497
Z	CYCLOPOIDA	6120	248	T176
Z	CYCLOPOIDA_(ADULT)	6120_98		
Z	CYCLOPOIDA_(COPEPODITE)	6120_12		
Z	CYCLOPOIDA_(NAUPLII)	6120_11		
Z	CYCLOPS	61200802	303	T3182
Z	CYCLOPS_(ADULT)	61200802_98		
Z	CYCLOPS_(COPEPODITE)	61200802_12		
Z	CYCLOPS_BICOLOR	61200802_BICOLO		
Z	CYCLOPS_BICOLOR_(ADULT)	61200802_BIC_98		
Z	CYCLOPS_BICOLOR_(COPEPODITE)	61200802_BIC_12		
Z	CYCLOPS_BICUSPIDATUS	6120080203		T730
Z	CYCLOPS_BICUSPIDATUS_(ADULT)	6120080203_98		
Z	CYCLOPS_BICUSPIDATUS_(COPEPODITE)	6120080203_12		
Z	CYCLOPS_BICUSPIDATUS_(NAUPLII)	6120080203_11		
Z	CYCLOPS_VARICANS	6120080204		T3183+T2320
Z	CYCLOPS_VARICANS_(ADULT)	6120080204_98		
Z	CYCLOPS_VARICANS_(COPEPODITE)	6120080204_12		
Z	CYCLOPS_VARICANS_RUBELLUS	612008020401		
Z	CYCLOPS_VARICANS_RUBELLUS_(ADULT)	612008020401_98		
Z	CYCLOPS_VARICANS_RUBELLUS_(COPEPODITE)	612008020401_12		
Z	CYCLOPS_VERNALIS	6120080201		T726
Z	CYCLOPS_VERNALIS_(ADULT)	6120080201_98		
Z	CYCLOPS_VERNALIS_(COPEPODITE)	6120080201_12		
Z	CYCLOPS_VERNALIS_(NAUPLII)	6120080201_11		

Z	CYMADUSA_COMPTA	6169040201		
Z	CYNOSCION_NEBULOSUS_(LARVAE)	8835440102_97	221	
Z	CYNOSCION_REGALIS_(EGG)	8835440104_00	380	
Z	CYNOSCION_REGALIS_(LARVAE)	8835440104_97	156	
Z	CYPHODERIA	34450401		
Z	CYPHODERIIDAE	344504		
Z	CYPRIA	61130302		
Z	CYPRIDOPSIS	61133001		
Z	CYPRINIDAE_(LARVAE)	877601_97	84	
Z	CYPRINUS_CARPIO_(LARVAE)	8776010101_97		
Z	CYRNELLUS	64181802		
Z	DAPHNIA	61090201	381	T643
Z	DAPHNIA_CATAWBA	6109020111		
Z	DAPHNIA_DUBIA	6109020119		
Z	DAPHNIA_LAEVIS	6109020114		
Z	DAPHNIA_LONGISPINA	6109020102	315	
Z	DAPHNIA_PARVULA	61090201PARVULA		
Z	DAPHNIA_PULEX	6109020101	185	
Z	DAPHNIA_RETROCURVA	6109020104		T642
Z	DECAPODA	6175	343	
Z	DECAPODA_(MEGALOPA)	6175_33	187	
Z	DECAPODA_(ZOEA)	6175_31		
Z	DEMICRYPTOCHIRONOMUS	65051079		
Z	DIAPHANOSOMA	61090102		T645
Z	DIAPHANOSOMA_BRACHYURUM	6109010201	340	
Z	DIAPHANOSOMA_LEUCHTENBERGIANUM	6109010202		T2038
Z	DIAPTOMUS	61181801	366	T715
Z	DIAPTOMUS_(ADULT)	61181801_98		
Z	DIAPTOMUS_(COPEPODITE)	61181801_12		
Z	DIAPTOMUS_(NAUPLII)	61181801_11		
Z	DIAPTOMUS_MINUTUS	6118180103		
Z	DIAPTOMUS_MINUTUS_(ADULT)	6118180103_98		
Z	DIAPTOMUS_MINUTUS_(COPEPODITE)	6118180103_12		
Z	DIAPTOMUS_PALLIDUS	6118180113		
Z	DIAPTOMUS_PALLIDUS_(ADULT)	6118180113_98		
Z	DIAPTOMUS_PALLIDUS_(COPEPODITE)	6118180113_12		
Z	DIAPTOMUS_SICILIS	6118180114		
Z	DIAPTOMUS_SICILIS_(ADULT)	6118180114_98		
Z	DIAPTOMUS_SICILIS_(COPEPODITE)	6118180114_12		
Z	DIAPTOMUS_SICILOIDES	6118180115		
Z	DIAPTOMUS_SICILOIDES_(ADULT)	6118180115_98		
Z	DIAPTOMUS_SICILOIDES_(COPEPODITE)	6118180115_12		
Z	DICROTENDIPES	65050845		
Z	DICROTENDIPES_FUMIDUS	6505084504		
Z	DICROTENDIPES_NERVOSUS	6505084511		
Z	DIDINIUM	35160101		
Z	DIFFLUGIA	34420101		
Z	DIFFLUGIIDAE	344201		
Z	DIOSACCUS_TENUICORNIS	6119280203	278	
Z	DIOSACCUS_TENUICORNIS_(ADULT)	6119280203_98		
Z	DIOSACCUS_TENUICORNIS_(COPEPODITE)	6119280203_12		
Z	DIPTERA_(LARVAE)	6501_97	382	T782
Z	DIPURENA_STRANGULATA	3703060202	3	
Z	DOROSOMA_CEPEDIANUM_(LARVAE)	8747010501_97		
Z	DOROSOMA_CEPEDIANUM_(VIABLE_EGG)	8747010501_00		
Z	DULICHIOPSIS_CYCLOPS	6169440703		
Z	ECHINODERMATA	81	250	
Z	ECTINOSOMA	61190902		
Z	ECTINOSOMA_(ADULT)	61190902_98		
Z	ECTINOSOMA_(COPEPODITE)	61190902_12		
Z	ECTINOSOMA_(NAUPLII)	61190902_11		
Z	ECTINOSOMA_CURTICORNE	6119090201_S	183	
Z	ECTINOSOMA_CURTICORNE_(ADULT)	6119090201_98_S		
Z	ECTINOSOMA_CURTICORNE_(COPEPODITE)	6119090201_12_S		
Z	ECTINOSOMA_CURTICORNE_(NAUPLII)	6119090201_11_S		
Z	ECTOCYCLOPS PHALERATUS	ECTOCYCLOPS_PHA		T3016
Z	ECTOPLEURA_DUMORTIERI	3703030501	1	
Z	EDOTEA	61620207	308	
Z	EDOTEA_TRILOBA	6162020703		
Z	EMERITA_TALPOIDA_(MEGALOPA)	6183140101_33		
Z	EMERITA_TALPOIDA_(ZOEA)	6183140101_31	110	
Z	ENALLAGMA_CIVILE	6229040618		
Z	ENDOCHIRONOMUS	65050888		
Z	EOGAMMARUS	61692119		
Z	EOSPHORA	45060406		
Z	EPHELOTA	35340401		

Z	EPHEMERELLA	62170201		
Z	EPHEMERELLA_VERRUCA	6217020187		
Z	EPHEMERELLIDAE	621702		T181
Z	EPHEMEROPTERA	6215		T237
Z	EPHYDRA	65380102		
Z	EPIPHANES	45060109		
Z	ERGASILIS_CHATAUQUAENSIS	61200501CHATAUQ		
Z	ERGASILIS_CHATAUQUAENSIS_(ADULT)	61200501CHAT_98		
Z	ERGASILIS_CHATAUQUAENSIS_(COPEPODITE)	61200501CHAT_12		
Z	ERGASILUS	61200501		T734
Z	ERGASILUS_(ADULT)	61200501_98		
Z	ERGASILUS_(COPEPODITE)	61200501_12		
Z	ERGASILUS_VERSILCOLOR	6120050105	355	
Z	ERGASILUS_VERSILCOLOR_(ADULT)	6120050105_98		
Z	ERGASILUS_VERSILCOLOR_(COPEPODITE)	6120050105_12		
Z	ETHEOSTOMA_OLMSTEDI_(LARVAE)	8835200102_97		
Z	ETHEOSTOMA_OLMSTEDI_(VIABLE_EGG)	8835200102_00		
Z	EUBOSMINA	61090302		
Z	EUBOSMINA_COREGONI	6109030202		T653
Z	EUBOSMINA_LONGISPINA	6109030201		T3074
Z	EUCALANUS_PILIEATUS	6118030105	182	
Z	EUCALANUS_PILIEATUS_(ADULT)	6118030105_98		
Z	EUCALANUS_PILIEATUS_(COPEPODITE)	6118030105_12		
Z	EUCERAMUS_PRAELONGUS_(MEGALOPA)	6183120301_33	174	T2266
Z	EUCERAMUS_PRAELONGUS_(ZOEA)	6183120301_31	105	
Z	EUCHEILOTA_VENTRICULARIS	3704110201	12	
Z	EUCLANIS	45060110		
Z	EUCLANIS_DILATATA	4506011001		
Z	EUCRICOTOPUS	65050826		
Z	EUCYCLOPS	61200804		T2329
Z	EUCYCLOPS_(ADULT)	61200804_98		
Z	EUCYCLOPS_(COPEPODITE)	61200804_12		
Z	EUCYCLOPS_AGILIS	6120080401		T654
Z	EUCYCLOPS_AGILIS_(ADULT)	6120080401_98		
Z	EUCYCLOPS_AGILIS_(COPEPODITE)	6120080401_12		
Z	EUCYCLOPS_AGILIS_(NAUPLII)	6120080401_11		
Z	EUCYCLOPS_SPERATUS	6120080403		T2328
Z	EUCYCLOPS_SPERATUS_(ADULT)	6120080403_98		
Z	EUCYCLOPS_SPERATUS_(COPEPODITE)	6120080403_12		
Z	EUGLYPHA	34450201		
Z	EUPLOTIDAE	354501		
Z	EURYCERCUS_LAMELLATUS	6109070301	404	T2327
Z	EURYPANOPEUS_DEPRESSUS_(MEGALOPA)	6189020501_33	389	
Z	EURYPANOPEUS_DEPRESSUS_(ZOEA)	6189020501_31	119	
Z	EURYTEMORA	61182002	239	
Z	EURYTEMORA_(COPEPODITE)	61182002_12		
Z	EURYTEMORA_(NAUPLII)	61182002_11		
Z	EURYTEMORA_AFFINIS	6118200201	238	T702
Z	EURYTEMORA_AFFINIS_(ADULT)	6118200201_98		
Z	EURYTEMORA_AFFINIS_(COPEPODITE)	6118200201_12		
Z	EURYTEMORA_AFFINIS_(NAUPLII)	6118200201_11		
Z	EURYTEMORA_AMERICANA	6118200202	69	
Z	EURYTEMORA_AMERICANA_(ADULT)	6118200202_98		
Z	EURYTEMORA_AMERICANA_(COPEPODITE)	6118200202_12		
Z	EURYTEMORA_COMPOSITA	61182002COMPOSI		
Z	EURYTEMORA_COMPOSITA_(ADULT)	61182002COMP_98		
Z	EURYTEMORA_COMPOSITA_(COPEPODITE)	61182002COMP_12		
Z	EURYTEMORA_HIRUNDOIDES	6118200204	237	
Z	EURYTEMORA_HIRUNDOIDES_(ADULT)	6118200204_98		
Z	EURYTEMORA_HIRUNDOIDES_(COPEPODITE)	6118200204_12		
Z	EUTERPINA_ACUTIFRONS	6119140101	79	
Z	EUTERPINA_ACUTIFRONS_(ADULT)	6119140101_98		
Z	EUTERPINA_ACUTIFRONS_(COPEPODITE)	6119140101_12		
Z	EUTINTINNUS	35401301		
Z	EVADNE	61090501	353	T2723
Z	EVADNE_NORDMANNI	6109050101	63	
Z	EVADNE_SPINIFERA	6109050103	65	
Z	EVADNE_TERGESTINA	6109050102	64	
Z	FARRANULA_GRACILIS	6120040201	86	
Z	FARRANULA_GRACILIS_(ADULT)	6120040201_98		
Z	FARRANULA_GRACILIS_(COPEPODITE)	6120040201_12		
Z	FAVELLA	35400701		
Z	FILINIA	45070501		
Z	FILINIA_BRACHIATA	4507050102		
Z	FILINIA_LONGISETA	4507050101		
Z	FILINIA_TERMINALIS	4507050103		

Z	FISH_(DEAD_EGG)	87_90		
Z	FISH_(LARVAE)	87_97	220	T585
Z	FISH_(VIABLE_EGG)	87_00	218	T584
Z	FLOSCULARIACEA	4507		
Z	FORAMINIFERIDA	3448		
Z	GAMMARIDAE	616921	295	T242+T464
Z	GAMMARUS	61692107	294	
Z	GAMMARUS_FASCIATUS	6169210706	312	T1037
Z	GAMMARUS_MUCRONATUS	6169210709		
Z	GASTROPODA	51	201	T543
Z	GASTROPODA_(VELIGER_LARVAE)	51_VELIGER		T871
Z	GASTROPUS	45060802		
Z	GASTROPUS_MINOR	4506080201		
Z	GASTROTRICHA	44		
Z	GLYPTOTENDIPES	65050839		
Z	GLYPTOTENDIPES_LOBIFERUS	6505083909		
Z	GOBIESOX_STRUMOSUS_(LARVAE)	8784010102_97	150	
Z	GOBIIDAE_(LARVAE)	884701_97	180	
Z	GOBIONELLUS_BOLEOSOMA_(LARVAE)	8847010501_97		
Z	GOBIONELLUS_HASTATUS_(LARVAE)	8847010502_97		
Z	GOBIOSOMA_(LARVAE)	88470106_97	361	
Z	GOBIOSOMA_BOSCI_(LARVAE)	8847010601_97	161	
Z	GOBIOSOMA_GINSBURGI_(LARVAE)	8847010602_97	357	
Z	GOELDICHIRONOMUS	65051014		
Z	GOLFINGIA_GLAUCA	7200020136		
Z	GONIADIDAE	500128	29	
Z	GRAMMARIA_STENTOR	3704020401		
Z	GYRINUS	63050801		
Z	HALECTINOSOMA	61190904		
Z	HALECTINOSOMA_(ADULT)	61190904_98		
Z	HALECTINOSOMA_(COPEPODITE)	61190904_12		
Z	HALECTINOSOMA_(NAUPLII)	61190904_11		
Z	HALECTINOSOMA_CURTICORNE	61190904CURT		
Z	HALECTINOSOMA_CURTICORNE_(ADULT)	61190904CURT_98		
Z	HALECTINOSOMA_CURTICORNE_(COPEPODITE)	61190904CURT_12		
Z	HALECTINOSOMA_CURTICORNE_(NAUPLII)	61190904CURT_11		
Z	HALICELLA_HALONA	6169400602		
Z	HALICYCLOPS	61200801	327	T729
Z	HALICYCLOPS_(ADULT)	61200801_98		
Z	HALICYCLOPS_(COPEPODITE)	61200801_12		
Z	HALICYCLOPS_(NAUPLII)	61200801_11		
Z	HALICYCLOPS_FOSTERI	6120080101		
Z	HALICYCLOPS_FOSTERI_(ADULT)	6120080101_98		
Z	HALICYCLOPS_FOSTERI_(COPEPODITE)	6120080101_12		
Z	HALICYCLOPS_FOSTERI_(NAUPLII)	6120080101_11		
Z	HALICYCLOPS_MAGNICEPS	61200801MAGN		T2206
Z	HALICYCLOPS_MAGNICEPS_(ADULT)	61200801MAGN_98		
Z	HALICYCLOPS_MAGNICEPS_(COPEPODITE)	61200801MAGN_12		
Z	HAPTORIDA	3516		
Z	HARPACTICOIDA	6119	228	T177+T3157
Z	HARPACTICOIDA_(ADULT)	6119_98		
Z	HARPACTICOIDA_(COPEPODITE)	6119_12		
Z	HARPACTICOIDA_(NAUPLII)	6119_11		
Z	HARPACTICUS	61191002	229	
Z	HARPACTICUS_(ADULT)	61191002_98		
Z	HARPACTICUS_(COPEPODITE)	61191002_12		
Z	HARPACTICUS_CHELIFER	6119100101	395	
Z	HARPACTICUS_CHELIFER_(ADULT)	6119100101_98		
Z	HARPACTICUS_CHELIFER_(COPEPODITE)	6119100101_12		
Z	HARPACTICUS_GRACILIS	6119100103	80	
Z	HARPACTICUS_GRACILIS_(ADULT)	6119100103_98		
Z	HARPACTICUS_GRACILIS_(COPEPODITE)	6119100103_12		
Z	HAUSTORIIDAE	616922	335	
Z	HELICHUS	63160102		
Z	HEMICYCLOPS	61202601		
Z	HEMICYCLOPS_(ADULT)	61202601_98		
Z	HEMICYCLOPS_(COPEPODITE)	61202601_12		
Z	HEMICYCLOPS_(NAUPLII)	61202601_11		
Z	HEMICYCLOPS_ADHAERENS	61202601ADHA		
Z	HEMICYCLOPS_ADHAERENS_(ADULT)	61202601ADHA_98		
Z	HEMICYCLOPS_ADHAERENS_(COPEPODITE)	61202601ADHA_12		
Z	HEMICYCLOPS_ADHAERENS_(NAUPLII)	61202601ADHA_11		
Z	HEMIPTERA	6271		T1133
Z	HESIONIDAE	500121	30	
Z	HETEROMYSIS_FORMOSA	6153010802	92	
Z	HETEROTRICHINA	3537		

Z	HEXAPANOPEUS_ANGUSTIFRONS_(MEGALOPA)	6189020601_33	121	
Z	HEXAPANOPEUS_ANGUSTIFRONS_(ZOEA)	6189020601_31	120	
Z	HEXARTHRA	45070201		
Z	HIPPOCAMPUS_ERECTUS_(LARVAE)	8820020201_97	153	
Z	HIPPOLYTE_PLEURACANTHA	6179160103		
Z	HIPPOLYTE_PLEURACANTHA_(ZOEA)	6179160103_31	326	
Z	HIPPOLYTIDAE	617916		
Z	HIPPOLYTIDAE_(ZOEA)	617916_31	101	
Z	HIRUDINEA	5012	251	T662
Z	HOLOPEDIUM	61090401	83	
Z	HOLOPEDIUM_GIBBERUM	6109040101		
Z	HOMARUS_AMERICANUS_(LARVAE)	6181010201_97		
Z	HOMARUS_GAMMARUS_(LARVAE)	6181010202_97		
Z	HYALOSPHENIIDAE	344205		
Z	HYBOGNATHUS_NUCHALIS_(LARVAE)	8776010502_97		
Z	HYBOGNATHUS_NUCHALIS_(VIABLE_EGG)	8776010502_00		
Z	HYDRACARINA	61HYDRACARINA	411	T267
Z	HYDROMEDUSAE	3701	255	
Z	HYDROPORUS	63050642		
Z	HYPERIA_GALBA	6170010101		
Z	HYPERIIDAE	617001	268	
Z	HYPEROCHE_MEDUSARUM	6170010702		
Z	HYPORHAMPHUS_UNIFASCIATUS_(LARVAE)	8803010301_97		
Z	HYPOTRICHIDA	3543		
Z	HYPSOBLENNIUS_HENTZI_(LARVAE)	8842010201_97	160	
Z	IANIRELLA_POLYCHAETA	6163060308		
Z	IDOTEA_BALTHICA	6162020308		
Z	ILYOCRYPTUS	61090805		T778
Z	ILYOCRYPTUS_ACUTIFRONS	61090805ACUTIFR		
Z	ILYOCRYPTUS_SORDIDUS	61090805SORDIDU		
Z	ILYOCRYPTUS_SPINIFER	6109080501	368	T671
Z	INSECTA	62		
Z	ISCHNURA	62290401		
Z	ISCHNURA_VERTICALIS	6229040101		
Z	ISOPODA	6158	258	T500
Z	KELLICOTIA_BOSTONIENSIS	45060105_BOSTON		
Z	KELLICOTTIA	45060105		
Z	KELLICOTTIA_LONGISPINA	4506010501		
Z	KERATELLA	45060101		
Z	KERATELLA_COCHLEARIS	4506010103		
Z	KERATELLA_COCHLEARIS_COCHLEARIS	450601010302		
Z	KERATELLA_COCHLEARIS_HISPIDA	450601010303		
Z	KERATELLA_COCHLEARIS_MICRACANTHA	450601010304		
Z	KERATELLA_COCHLEARIS_ROBUSTA	450601010305		
Z	KERATELLA_COCHLEARIS_TECTA	450601010306		
Z	KERATELLA_CRASSA	4506010104		
Z	KERATELLA_EARLINAE	4506010105		
Z	KERATELLA_QUADRATA	4506010102		
Z	KERATELLA_VALGA	4506010106		
Z	KRENOSMITTIA	65051040		
Z	LABIDOCERA_AESTIVA	6118270205	73	T3151
Z	LABIDOCERA_AESTIVA_(ADULT)	6118270205_98		
Z	LABIDOCERA_AESTIVA_(COPEPODITE)	6118270205_12	373	
Z	LABIDOCERA_WOLLASTONI	6118270204	74	
Z	LABIDOCERA_WOLLASTONI_(ADULT)	6118270204_98		
Z	LABIDOCERA_WOLLASTONI_(COPEPODITE)	6118270204_12		
Z	LAEONEREIS_GLAUCA	5001240802		
Z	LATHONURA_RECTIROSTRIS	L_RECTIROSTRIS		T3001
Z	LATONA_SETIFERA	6109010401		T676
Z	LATONOPSIS_FASCICULATA	6109010602	346	
Z	LECANE	45060201		
Z	LEIOSTOMUS_XANTHURUS_(LARVAE)	8835440401_97	179	
Z	LEPADELLA	45060107		
Z	LEPADELLA_PATELLA	4506010704		
Z	LEPIDOPA_WEBSTERI_(MEGALOPA)	6183130101_33		
Z	LEPIDOPA_WEBSTERI_(ZOEA)	6183130101_31	111	
Z	LEPOMIS_(LARVAE)	88351605_97		
Z	LEPOMIS_AURITUS_(LARVAE)	8835160501_97		
Z	LEPOMIS_CYANELLUS_(LARVAE)	8835160502_97		
Z	LEPOMIS_GIBBOSUS_(LARVAE)	8835160505_97		
Z	LEPOMIS_GULOSUS_(LARVAE)	8835160503_97		
Z	LEPOMIS_HUMILIS_(LARVAE)	8835160506_97		
Z	LEPOMIS_MACROCHIRUS_(LARVAE)	8835160504_97		
Z	LEPOMIS_MEGALOTIS_(LARVAE)	8835160508_97		
Z	LEPOMIS_MICROLOPHUS_(LARVAE)	8835160509_97		
Z	LEPOMIS_PUNCTATUS_(LARVAE)	8835160510_97		

Z	LEPTOCHEIRUS_PLUMULOSUS	6169060701		T466
Z	LEPTODORA_KINDTII	6109060101	331	T678
Z	LEPTOMEDUSAE	3704	230	
Z	LESQUEREUSIA_GIBBOSA	3442010201		
Z	LEUCON_AMERICANUS	6154040110		T481
Z	LEYDIGIA	61090705		
Z	LEYDIGIA_ACANTHOCERCOIDES	6109070502		
Z	LEYDIGIA_QUADRANGULARIS	6109070501		T779
Z	LIBELLULA	62260125		
Z	LIBINIA_(MEGALOPA)	61870109_33	193	
Z	LIBINIA_(ZOEA)	61870109_31	136	
Z	LIBINIA_DUBIA_(MEGALOPA)	6187010901_33		
Z	LIBINIA_DUBIA_(ZOEA)	6187010901_31		
Z	LIBINIA_EMARGINATA_(MEGALOPA)	6187010902_33	137	
Z	LIBINIA_EMARGINATA_(ZOEA)	6187010902_31		
Z	LIMNOCALANUS	61181702		
Z	LIMNOCALANUS_(ADULT)	61181702_98		
Z	LIMNOCALANUS_(COPEPODITE)	61181702_12		
Z	LIMNOCALANUS_MACRURUS	6118170202		
Z	LIMNOCALANUS_MACRURUS_(ADULT)	6118170202_98		
Z	LIMNOCALANUS_MACRURUS_(COPEPODITE)	6118170202_12		
Z	LINVILLEA_AGASSIZI	3703060301	4	
Z	LIRIOPE_TETRAPHYLLA	3711030201	15	
Z	LIRONECA_OVALIS	6161060301		
Z	LOLIGUNCULA_BREVIS	5706010201	59	
Z	LOPHIUS_AMERICANUS_(LARVAE)	8786010101_97	151	
Z	LOPHOCHARIS	45060111		
Z	LOPHOCHARIS_SALPINA	4506011101		
Z	LOVENELLA_GRACILIS	3704110101	13	
Z	LUCIFER_FAXONI	6177020201		T2621
Z	LUCIFER_FAXONI_(ZOEA)	6177020201_31	94	
Z	LUMPENUS_SAGITTA	8842120902		
Z	LYSMATA_WURDEMANNI	6179161101		
Z	LYSMATA_WURDEMANNI_(ZOEA)	6179161101_31	313	
Z	MACROBRACHIUM_OHIONE	6179110201		
Z	MACROBRACHIUM_OHIONE_(ZOEA)	6179110201_31	97	
Z	MACROCHAETUS	45060112		
Z	MACROCYCLOPS	61200806		
Z	MACROCYCLOPS_(ADULT)	61200806_98		
Z	MACROCYCLOPS_(COPEPODITE)	61200806_12		
Z	MACROCYCLOPS_ALBIDUS	6120080601		T2335
Z	MACROCYCLOPS_ALBIDUS_(ADULT)	6120080601_98		
Z	MACROCYCLOPS_ALBIDUS_(COPEPODITE)	6120080601_12		
Z	MACROCYCLOPS_FUSCUS	61200806FUSC		
Z	MACROCYCLOPS_FUSCUS_(ADULT)	61200806FUSC_98		
Z	MACROCYCLOPS_FUSCUS_(COPEPODITE)	61200806FUSC_12		
Z	MACROTHRIX	61090801		
Z	MACROTHRIX_LATICORNIS	6109080103		T681
Z	MACROTRACHELA	45040203		
Z	MAGELLONIDAE	500144	31	
Z	MARENZELLERIA_VIRIDIS	50014323		
Z	MARINOGAMMARUS	61692117		
Z	MEGACYCLOPS	61200808		
Z	MEGACYCLOPS_(ADULT)	61200808_98		
Z	MEGACYCLOPS_(COPEPODITE)	61200808_12		
Z	MELITA	61692110		T682
Z	MELITA_NITIDA	6169211006		
Z	MEMBRAS_(LARVAE)	88050202_97	321	
Z	MEMBRAS_MARTINICA_(LARVAE)	8805020201_97	270	
Z	MENIDIA_(LARVAE)	88050203_97		
Z	MENIDIA_BERYLLINA_(LARVAE)	8805020301_97	354	
Z	MENIDIA_BERYLLINA_(VIABLE_EGG)	8805020301_00		
Z	MENIDIA_MENIDIA_(LARVAE)	8805020302_97	152	
Z	MENIDIA_MENIDIA_(VIABLE_EGG)	8805020302_00		
Z	MENTICIRRHUS_(LARVAE)	88354406_97	298	
Z	MENTICIRRHUS_SAXATILIS_(LARVAE)	8835440603_97	157	
Z	MERCENARIA_MERCENARIA	5515471101	52	
Z	MEROPLANKTON_(LARVAE)	MEROPLANKTON		
Z	MESOCRICOTOPUS	65051039		
Z	MESOCYCLOPS	61200803	306	
Z	MESOCYCLOPS_(ADULT)	61200803_98		
Z	MESOCYCLOPS_(COPEPODITE)	61200803_12		
Z	MESOCYCLOPS_(NAUPLII)	61200803_11		
Z	MESOCYCLOPS_EDAX	6120080301		T728
Z	MESOCYCLOPS_EDAX_(ADULT)	6120080301_98		
Z	MESOCYCLOPS_EDAX_(COPEPODITE)	6120080301_12		

Z MESOCYCLOPS_EDAX_(NAUPLII)	6120080301_11		
Z MESOCYCLOPS_OBSOLETUS	61200803OBSULET		
Z MESOCYCLOPS_OBSOLETUS	61200803OBSOLET	317	
Z MESOCYCLOPS_OBSOLETUS_(ADULT)	61200803OBSU_98		
Z MESOCYCLOPS_OBSOLETUS_(COPEPODITE)	61200803OBSU_12		
Z METACYLIS	35400501		
Z METAMYSIDOPSIS	61530314	243	
Z METIS	61191601	399	
Z METIS_(ADULT)	61191601_98		
Z METIS_(COPEPODITE)	61191601_12		
Z METIS_HOLOTHURIAE	6119160102		
Z METIS_HOLOTHURIAE_(ADULT)	6119160102_98		
Z METIS_HOLOTHURIAE_(COPEPODITE)	6119160102_12		
Z METRIDIA	61181602	372	
Z METRIDIA_(ADULT)	61181602_98		
Z METRIDIA_(COPEPODITE)	61181602_12		
Z MIATHYRIA_MARCELLA	6226013801		
Z MICROARTHRIDION_LITTORALE	6119140201		
Z MICROARTHRIDION_LITTORALE_(ADULT)	6119140201_98		
Z MICROARTHRIDION_LITTORALE_(COPEPODITE)	6119140201_12		
Z MICROCHIRONOMUS	65051072		
Z MICROCRICOTOPUS	65051038		
Z MICROCYCLOPS	61200810		
Z MICROCYCLOPS_(ADULT)	61200810_98		
Z MICROCYCLOPS_(COPEPODITE)	61200810_12		
Z MICROCYCLOPS_VARICANS	6120081001		
Z MICROCYCLOPS_VARICANS_(ADULT)	6120081001_98		
Z MICROCYCLOPS_VARICANS_(COPEPODITE)	6120081001_12		
Z MICROGOBIOUS_GULOSUS_(LARVAE)	8847010701_97		
Z MICROGOBIUS_THALASSINUS_(LARVAE)	8847010702_97	271	
Z MICROPOGON_UNDULATUS_(LARVAE)	8835440701_97	272	
Z MICROTENDIPES	65050828		
Z MITRELLA_LUNATA	5105030207	58	
Z MNEMIOPSIS_BRACHEI	38030201BRACHEI	370	
Z MNEMIOPSIS_LEIDYI	3803020101	21	
Z MOINA	61090901		T2190
Z MOINA_AFFINIS	6109090103		
Z MOINA_BRANCHIATA	61090901BRANCHI	142	
Z MOINA_MACROCOPA	6109090102		
Z MOINA_MICRURA	6109090101		T683
Z MOINODAPHNIA	61090902		
Z MOLLUSCA	5085		T236+T978
Z MONOCULODES	61693708	309	T684
Z MONOCULODES_EDWARDSI	6169370820		T468
Z MONOMMATA	45060405		
Z MONOSPILUS_DISPAR	6109070701		T2040
Z MONOSTYLA	45060202		
Z MONOSTYLA_BULLA	4506020201		
Z MONOSTYLA_CLOSTEROCERCA	4506020202		
Z MONOSTYLA_LUNARIS	45060202_LUNARI		
Z MONOSTYLA_QUADRIDENTATA	4506020203		
Z MONOSTYLA_STENROOSI	45060202_STENRO		
Z MORONE_(LARVAE)	88350201_97	82	
Z MORONE_AMERICANA_(LARVAE)	8835020101_97		
Z MORONE_AMERICANA_(VIABLE_EGG)	8835020101_00		
Z MORONE_SAXATILIS_(LARVAE)	8835020102_97	336	
Z MORONE_SAXATILIS_(VIABLE_EGG)	8835020102_00		
Z MUCROGAMMARUS MUCRONATUS	MUCROGAMMARUS_M		T463
Z MULINA_LATERALIS	5515250301	53	
Z MYCOLIA_MAJOR	MYCOLIA_MAJOR		
Z MYROPHIS_PUNCTATUS_LEPTOCEPHALUS_(LARVAE	8741130802LE_97		
Z MYSELLA	55151001	51	
Z MYSIDACEA	6153		
Z MYSIDAE	615301	314	T820
Z MYSIDOPSIS_BIGELOWI	6153012101	91	
Z MYTILIDAE	550701	49	
Z MYTILINA	45060113		
Z MYTILUS_EDULIS	5507010101	48	
Z NAIDIDAE	500903	371	
Z NANNOCALANUS	611801NANNOCALA	202	
Z NATICIDAE	510376	57	
Z NAUSHONIA_CRANGONOIDES	6183030101		
Z NAUSHONIA_CRANGONOIDES_(ZOEA)	6183030101_31	217	
Z NEMATODA	47	23	
Z NEMOPSIS_BACHEI	3703011401	8	
Z NEOMYSIS_AMERICANA	6153011508	90	

Z NEOPANOPE_TEXANA_SAYI_(MEGALOPA)	618902070101_33	123	
Z NEOPANOPE_TEXANA_SAYI_(ZOEA)	618902070101_31	122	
Z NEPHYTIDAE	500125	32	
Z NEREIDAE	500124	34	
Z NEREIS_SUCCINEA	5001240410	33	
Z NETAMELITA	61692123		
Z NILOTANYPUS	65051016		
Z NITOCRA	61191801		
Z NITOCRA_(ADULT)	61191801_98		
Z NITOCRA_(COPEPODITE)	61191801_12		
Z NITOCRA_(NAUPLII)	61191801_11		
Z NITOCRA_LACUSTRIS	61191801_LACUST		
Z NITOCRA_LACUSTRIS_(ADULT)	61191801_LAC_98		
Z NITOCRA_LACUSTRIS_(COPEPODITE)	61191801_LAC_12		
Z NITOCRELLA_POLYCHAETA	6119181829		
Z NITOCRELLA_POLYCHAETA_(ADULT)	6119181829_98		
Z NITOCRELLA_POLYCHAETA_(COPEPODITE)	6119181829_12		
Z NOTHOLCA	45060102		
Z NOTHOLCA_ACUMINATA	4506010203		
Z NOTOMMATA	45060404		
Z NOTROPIS_(LARVAE)	87760111_97		
Z NOTROPIS_HUDSONICUS_(LARVAE)	8776011106_97		
Z NUDIBRANCHIA	5127	173	
Z OBELIA	37040102	11	
Z OCHTERIDAE	627208		
Z OCYPODIDAE_(MEGALOPA)	618909_33		
Z OCYPODIDAE_(ZOEA)	618909_31	362	
Z ODONATA	6223		
Z OECETIS	64181204		
Z OGYRIDAE	622902		
Z OGYRIDAE_(ZOEA)	622902_31	100	
Z OGYRIDES	61791501		
Z OGYRIDES_(ZOEA)	61791501_31	190	
Z OITHONA	61200901	265	
Z OITHONA_(ADULT)	61200901_98		
Z OITHONA_(COPEPODITE)	61200901_12		
Z OITHONA_(NAUPLII)	61200901_11		
Z OITHONA_COLCARVA	6120090109	266	T721
Z OITHONA_COLCARVA_(ADULT)	6120090109_98		
Z OITHONA_COLCARVA_(COPEPODITE)	6120090109_12		
Z OITHONA_COLCARVA_(NAUPLII)	6120090109_11		
Z OITHONA_SIMILIS	6120090103		
Z OITHONA_SIMILIS_(ADULT)	6120090103_3		
Z OITHONA_SIMILIS_(COPEPODITE)	6120090103_12		
Z OITHONA_SIMILIS_(NAUPLII)	6120090103_11		
Z OLENCIRA_PRAEGUSTATOR	6161060401		
Z OLIGOCHAETA	5003	296	T974
Z OLIGOTRICH	3539		
Z ONCAEA	61200103	192	
Z ONCAEA_(ADULT)	61200103_98		
Z ONCAEA_(COPEPODITE)	61200103_12		
Z ONCAEA_VENUSTA	6120010312	339	
Z ONCAEA_VENUSTA_(ADULT)	6120010312_98		
Z ONCAEA_VENUSTA_(COPEPODITE)	6120010312_12		
Z OPHELIIDAE	500158	35	
Z OPHIUROIDEA	8120	141	
Z OPHRYOXUS_GRACILIS	6109080401	365	
Z ORTHOCYCLOPS	61200809		
Z ORTHOCYCLOPS_(ADULT)	61200809_98		
Z ORTHOCYCLOPS_(COPEPODITE)	61200809_12		
Z OSTRACODA	6110	310	T981
Z OVALIPES_OCELLATUS_(MEGALOPA)	6189010502_33	188	
Z OVALIPES_OCELLATUS_(ZOEA)	6189010502_31	114	
Z OXYUROSTYLIS_SMITHI	6154050801		T482
Z PAGURIDAE	618306	209	
Z PAGURIDAE_(MEGALOPA)	618306_33	200	
Z PAGURIDAE_(ZOEA)	618306_31		
Z PAGURUS_(MEGALOPA)	61830602_33		
Z PAGURUS_(ZOEA)	61830602_31	231	
Z PAGURUS_(ZOEA)	61830602_31	109	
Z PAGURUS_LONGICARPUS_(MEGALOPA)	6183060230_33		
Z PAGURUS_LONGICARPUS_(ZOEA)	6183060230_31	107	
Z PAGURUS_POLLICARUS_(MEGALOPA)	6183060232_33		
Z PAGURUS_POLLICARUS_(ZOEA)	6183060232_31	108	
Z PALAEMONETES	61791103		T106
Z PALAEMONETES_(ZOEA)	61791103_31	98	

Z	PALAEMONETES_INTERMEDIUS	6179110301		
Z	PALAEMONETES_INTERMEDIUS_(ZOEA)	6179110301_31		
Z	PALAEMONETES_PUGIO	6179110303		
Z	PALAEMONETES_PUGIO_(ZOEA)	6179110303_31	359	
Z	PALAEMONETES_VULGARIS	6179110304		
Z	PALAEMONETES_VULGARIS_(ZOEA)	6179110304_31		
Z	PALAEMONIDAE	617911		
Z	PALAEMONIDAE_(ZOEA)	617911_31		
Z	PALAENOTUS_HETEROSETA	5001080103	27	
Z	PANOPEUS_HERBSTII_(MEGALOPA)	6189020801_33	324	
Z	PANOPEUS_HERBSTII_(ZOEA)	6189020801_31	169	
Z	PARACALANUS	61180401	66	T3017
Z	PARACALANUS_(ADULT)	61180401_98		
Z	PARACALANUS_(COPEPODITE)	61180401_12		
Z	PARACALANUS_CRASSIROSTRIS	6118040102	67	T707
Z	PARACALANUS_CRASSIROSTRIS_(ADULT)	6118040102_98		
Z	PARACALANUS_CRASSIROSTRIS_(COPEPODITE)	6118040102_12		
Z	PARACALANUS_CRASSIROSTRIS_(NAUPLII)	6118040102_11		
Z	PARACALANUS_FIMBRIATUS	61180401FIMBRIA	350	
Z	PARACALANUS_FIMBRIATUS_(ADULT)	61180401FIMB_98		
Z	PARACALANUS_FIMBRIATUS_(COPEPODITE)	61180401FIMB_12		
Z	PARACALANUS_INDICUS	6118040103	274	
Z	PARACALANUS_INDICUS_(ADULT)	6118040103_98		
Z	PARACALANUS_INDICUS_(COPEPODITE)	6118040103_12		
Z	PARACAPRELLA_TENUIS	6171010901		T2808
Z	PARACHIRONOMUS	65050854		
Z	PARACLADOPELMA	65051061		
Z	PARACRICOTOPUS	65050850		
Z	PARACYCLOPS	61200805	338	T2803
Z	PARACYCLOPS_(ADULT)	61200805_98		
Z	PARACYCLOPS_(COPEPODITE)	61200805_12		
Z	PARACYCLOPS_FIMBRIATUS	6120080501		T3102
Z	PARACYCLOPS_FIMBRIATUS_(ADULT)	6120080501_98		
Z	PARACYCLOPS_FIMBRIATUS_(COPEPODITE)	6120080501_12		
Z	PARACYCLOPS_FIMBRIATUS_POPPEI	612008050101		T2041
Z	PARACYCLOPS_FIMBRIATUS_POPPEI_(ADULT)	612008050101_98		
Z	PARACYCLOPS_FIMBRIATUS_POPPEI_(COPEPODIT	612008050101_12		
Z	PARALAOPHONTE_BREVIROSTRIS	6119150101	400	
Z	PARALAOPHONTE_BREVIROSTRIS_(ADULT)	6119150101_98		
Z	PARALAOPHONTE_BREVIROSTRIS_(COPEPODITE)	6119150101_12		
Z	PARALAOPHONTE_CONGENERA	6119150102		
Z	PARALAOPHONTE_CONGENERA_(ADULT)	6119150102_98		
Z	PARALAOPHONTE_CONGENERA_(COPEPODITE)	6119150102_12		
Z	PARANAITIS_SPECIOSA	5001130801	37	
Z	PARAPRIONOSPIO_PINNATA	5001431701	41	
Z	PARAQUADRULIDAE	344204		
Z	PARATANYTARSUS	65050894		
Z	PARATHEMISTO_COMPRESSA	6170011008		T2617
Z	PARATHEMISTO_LIBELLULA	6170011002		
Z	PARONYCHOCAMPTUS	61191528		
Z	PARONYCHOCAMPTUS_(ADULT)	61191528_98		
Z	PARONYCHOCAMPTUS_(COPEPODITE)	61191528_12		
Z	PECTINARIA_GOULDII	5001660302	36	
Z	PEDALIA	PEDALIA		
Z	PELAGIA_NOCTILUCA	3734010301		
Z	PENAEIDAE	617701		
Z	PENAEIDAE_(ZOEA)	617701_31	208	
Z	PENAEUS_AZTECUS	6177010101		
Z	PENAEUS_AZTECUS_(ZOEA)	6177010101_31	95	
Z	PENAEUS_AZTECUS_AZTECUS	617701010101		
Z	PENAEUS_AZTECUS_AZTECUS_(ZOEA)	617701010101_31		
Z	PENAEUS_DUORARUM_DUORARUM	617701010201		
Z	PENAEUS_DUORARUM_DUORARUM_(ZOEA)	617701010201_31		
Z	PENILIA_AVIROSTRIS	6109010101	214	
Z	PENNARIA_TIARELLA	3703040307	2	
Z	PENTANEURA_MONILIS	6505085604	385	
Z	PEPRILUS_PARU_(LARVAE)	8851030102_97	170	
Z	PERCA_FLAVESCENS_(LARVAE)	8835200201_97	145	
Z	PERCA_FLAVESCENS_(VIABLE_EGG)	8835200201_00		
Z	PHERUSA_ERUCA	5001540309		
Z	PHIALUCIUM_CAROLINAE	3704180101	14	
Z	PHORONIS_ARCHITECTA	7700010203	24	
Z	PHYLLODOCIDAE	500113	38	
Z	PINNIXIA_(MEGALOPA)	61890604_33	199	
Z	PINNIXIA_(ZOEA)	61890604_31	129	
Z	PINNIXIA_CHAETOPTERANA_(MEGALOPA)	6189060405_33	126	

Z PINNIXIA_CHAETOPTERANA_(ZOEA)	6189060405_31	125	
Z PINNIXIA_CYLINDRICA_(MEGALOPA)	6189060406_33		
Z PINNIXIA_CYLINDRICA_(ZOEA)	6189060406_31	127	
Z PINNIXIA_SAYANNA_(MEGALOPA)	6189060408_33		
Z PINNIXIA_SAYANNA_(ZOEA)	6189060408_31	128	
Z PINNOTHERES_MACULATUS_(MEGALOPA)	6189060201_33		
Z PINNOTHERES_MACULATUS_(ZOEA)	6189060201_31	130	
Z PINNOTHERES_OSTREUM_(MEGALOPA)	6189060202_33	131	
Z PINNOTHERES_OSTREUM_(ZOEA)	6189060202_31	203	
Z PINNOTHERIDAE_(MEGALOPA)	618906_33		
Z PINNOTHERIDAE_(ZOEA)	618906_31	388	
Z PIROMIS_ERUCA	5001540501		
Z PISCICOLA	50140104		
Z PISCICOLIDAE	501401		T1579
Z PLATYHELMINTHES	39		T788
Z PLATYIAS	45060114		
Z PLATYIAS_PATULUS	4506011401		
Z PLATYIAS_QUADRICORNIS	4506011402		
Z PLEUROXUS	61090709		
Z PLEUROXUS_DENTICULATUS	6109070901		
Z PLEUROXUS_HAMULATUS	61090709HAMULAT		
Z PLIOMA	4506		
Z PLOESOMA	45061304		
Z PLOESOMA_HUDSONI	4506130401		
Z PLOESOMA_TRUNCATUM	4506130402		
Z PODOCORYNE_MINIMA	3703180206	6	
Z PODON_INTERMEDIUS	6109050201	257	
Z PODON_POLYPHEMOIDES	6109050203	62	T775
Z POGONIAS_CROMIS_(VIABLE_EGG)	8835440801_00	158	
Z POLINICES_DUPLICATUS	5103760407	56	
Z POLYARTHRA	45061303		
Z POLYARTHRA_DISSIMULANS	4506130302		
Z POLYARTHRA_DOLICHOPTERA	4506130303		
Z POLYARTHRA_EURYPTERA	4506130304		
Z POLYARTHRA_MAJOR	4506130305		
Z POLYARTHRA_REMATA	4506130306		
Z POLYARTHRA_VULGARIS	4506130307		
Z POLYCHAETA	5001	181	
Z POLYCHAETA_(METATROCHOPHORE)	5001METATROCHOP		
Z POLYCHAETA_(POSTTROCHOPHORE)	5001POSTTROCHOP		
Z POLYCHAETA_(TROCHOPHORE)	5001TROCHOPHORE		
Z POLYCHAETE_LARVAE	5001_97		T760
Z POLYDORA_LIGNI	5001430411	40	
Z POLYNOIDIDAE	500102	39	
Z POLYONYX_GIBBESI_(MEGALOPA)	6183120401_33	178	
Z POLYONYX_GIBBESI_(ZOEA)	6183120401_31	106	
Z POLYPEDILUM	65050837		
Z POLYPEDILUM_SCALAENUM	6505083738		
Z POLYPHEMUS_PEDICULUS	6109050301		
Z POMOXIS_(LARVAE)	88351607_97		
Z PONTELLA_MEADI	6118270302	322	
Z PONTELLA_MEADI_(ADULT)	6118270302_98		
Z PONTELLA_MEADI_(COPEPODITE)	6118270302_12		
Z PONTELLA_PENNATA	6118270301	75	
Z PONTELLA_PENNATA_(ADULT)	6118270301_98		
Z PONTELLA_PENNATA_(COPEPODITE)	6118270301_12		
Z PONTELLIDAE	611827	393	
Z PORCELLANIDAE_(MEGALOPA)	618312_33	207	
Z PORCELLANIDAE_(ZOEA)	618312_31	205	
Z PORTUNIDAE_(MEGALOPA)	618901_33		
Z PORTUNIDAE_(ZOEA)	618901_31	117	
Z PORTUNUS_(MEGALOPA)	61890106_33		
Z PORTUNUS_(ZOEA)	61890106_31		
Z PORTUNUS_GIBBESII_(MEGALOPA)	6189010601_33		
Z PORTUNUS_GIBBESII_(ZOEA)	6189010601_31	115	
Z PORTUNUS_SPINICARPUS_(MEGALOPA)	6189010603_33	216	
Z PORTUNUS_SPINICARPUS_(ZOEA)	6189010603_31		
Z PORTUNUS_SPINIMANUS_(MEGALOPA)	6189010604_33		
Z PORTUNUS_SPINIMANUS_(ZOEA)	6189010604_31	116	
Z PROBOSCIDACTYLA_ORNATA	3705060201	10	
Z PROCLADIUS	65050804		
Z PROCLADIUS_BELLUS	6505080401		
Z PROCLADIUS_CULICIFORMIS	6505080402		
Z PROSMITTIA	65050870		
Z PROTANYPUS	65050871		
Z PSECTROCLADIUS	65051025		

Z PSECTROTANYPUS	65050872		
Z PSEUDECTINOSOMA	61190915		
Z PSEUDECTINOSOMA_(ADULT)	61190915_98		
Z PSEUDECTINOSOMA_(COPEPODITE)	61190915_12		
Z PSEUDOBRADYA_OLIGOCHAETA	6119090320		
Z PSEUDOBRADYA_OLIGOCHAETA_(ADULT)	6119090320_98		
Z PSEUDOBRADYA_OLIGOCHAETA_(COPEPODITE)	6119090320_12		
Z PSEUDOCALANUS_MINUTUS	6118050502		T709
Z PSEUDOCALANUS_MINUTUS_(ADULT)	6118050502_98		
Z PSEUDOCALANUS_MINUTUS_(COPEPODITE)	6118050502_12		
Z PSEUDOCALANUS_MINUTUS_(NAUPLII)	6118050502_11		
Z PSEUDOCHIRONOMUS	65050838		
Z PSEUDOCHYDORUS	61090706		
Z PSEUDOCHYDORUS_GLOBOSUS	61090706GLOBOSU		
Z PSEUDOCYCLOPS	61181501		
Z PSEUDOCYCLOPS_(ADULT)	61181501_98		
Z PSEUDOCYCLOPS_(COPEPODITE)	61181501_12		
Z PSEUDODIAPTOMUS	61181902		
Z PSEUDODIAPTOMUS_(ADULT)	61181902_98		
Z PSEUDODIAPTOMUS_(COPEPODITE)	61181902_12		
Z PSEUDODIAPTOMUS_(NAUPLII)	61181902_11		
Z PSEUDODIAPTOMUS_CORONATUS	6118190201	240	T706
Z PSEUDODIAPTOMUS_CORONATUS_(ADULT)	6118190201_98		
Z PSEUDODIAPTOMUS_CORONATUS_(COPEPODITE)	6118190201_12		
Z PSEUDODIAPTOMUS_CORONATUS_(NAUPLII)	6118190201_11		
Z PSEUDODIFFLUGIA	34420804		
Z PSEUDODIFFLUGIIDAE	344208		
Z PSEUDONYCHOCAMPTUS_PROXIMUS	P_PROXIMUS		
Z PSEUDOPLEURONECTES_AMERICANUS_(LARVAE)	8857042001_97		
Z PSEUDOSIDA_BIDENTATA	6109010501	383	
Z PSEUDOSMITTIA	65050875		
Z PSILOTANYPUS	65051023		
Z PSYCHODA	65040101		
Z PSYCHOMYIA	64180301		
Z PTERONARCELLA	62520102		
Z PTEROSAGITTA	83030201		
Z PYCNOGONIDA	60	60	
Z PYRALIDAE	646302		
Z QUADRULELLA	34420402		
Z RAMELLOGAMMARUS	61692146		
Z RATHKEA_OCTOPUNCTATA	3703190203		
Z RHEOCRICOTOPUS	65051019		
Z RHEOSMITTIA	65051078		
Z RHEOTANYTARSUS	65050849		
Z RHITHROPANOPEUS_HARRISII_(ZOEA)	6189020901_31	290	
Z RHITHROPANOPEUS_HARRISSI_(MEGALOPA)	6189020901_33	358	
Z RHITHROPANOPEUS_HERMANDII_(MEGALOPA)	61890209HERM_31		
Z RHITHROPANOPEUS_HERMANDII_(ZOEA)	61890209HERM_31		
Z RHOPILEMA_VERRILLI	3737040201		
Z RHYNCHOTALONA	61090713		
Z RIVULOGAMMARUS	616921RIVULOGAM		
Z ROBERTGURNEYA_OLIGOCHAETA	6119280611		
Z ROBERTGURNEYA_OLIGOCHAETA_(ADULT)	6119280611_98		
Z ROBERTGURNEYA_OLIGOCHAETA_(COPEPODITE)	6119280611_12		
Z ROBERTSONIA_CHESAPEAKENSIS	61192815CHESAPE	387	
Z ROBERTSONIA_CHESAPEAKENSIS_(ADULT)	61192815CHES_98		
Z ROBERTSONIA_CHESAPEAKENSIS_(COPEPODITE)	61192815CHES_12		
Z ROTARIA	45040201		
Z ROTARIA_CITRINUS	4504020103		
Z ROTARIA_NEPTUNIA	4504020104		
Z ROTIFERA	45	386	
Z SACCULINA	61410203		
Z SACCULINA_(CYPRIS)	61410203_17		
Z SACCULINA_(NAUPLII)	61410203_11		
Z SAGITTA	83030101	198	T2217
Z SAGITTA_ELEGANS	8303010103		
Z SAGITTA_ENFLATA	8303010104	143	
Z SAPHIRELLA	61200602	245	T22F1
Z SAPHIRELLA_(ADULT)	61200602_98		
Z SAPHIRELLA_(COPEPODITE)	61200602_12		
Z SAPPHIRINA	61201001	289	
Z SAPPHIRINA_(ADULT)	61201001_98		
Z SAPPHIRINA_(COPEPODITE)	61201001_12		
Z SAPPHONCAEA	61200105		
Z SAPPHONCAEA_(ADULT)	61200105_98	0	
Z SAPPHONCAEA_(COPEPODITE)	61200105_12		

Z SARCODINA	3438		
Z SARSIA_TUBULOSA	3703060902		
Z SCAPHOLEBERIS	61090205		
Z SCAPHOLEBERIS_KINGI	6109020502		T2039
Z SCIAENIDAE_(LARVAE)	883544_97	213	
Z SCIAENIDAE_(VIABLE_EGG)	883544_00	210	
Z SCOLECOLEPIDES_VIRIDIS	5001430602		
Z SCOPTHALMUS_AQUOSUS_(LARVAE)	8857030401_97	163	
Z SCOPTHALMUS_AQUOSUS_(VIABLE_EGG)	8857030401_00	164	
Z SCOTTOLANA	61190502		
Z SCOTTOLANA_(ADULT)	61190502_98		
Z SCOTTOLANA_(COPEPODITE)	61190502_12		
Z SCOTTOLANA_(NAUPLII)	61190502_11		
Z SCOTTOLANA_CANADENSIS	6119050201		
Z SCOTTOLANA_CANADENSIS_(ADULT)	6119050201_98		
Z SCOTTOLANA_CANADENSIS_(COPEPODITE)	6119050201_12		
Z SCOTTOLANA_CANADENSIS_(NAUPLII)	6119050201_11		
Z SCYPHOZOA	3730	291	
Z SEMAEOSTOMAE	3734	232	
Z SEMIBALANUS_BALANOIDES	6134050301		
Z SEMIBALANUS_BALANOIDES_(CYPRIS)	6134050301_17		
Z SEMIBALANUS_BALANOIDES_(NAUPLII)	6134050301_11		
Z SESARAMA_(ZOEA)	61890702_31	276	
Z SESARMA_(EGG)	61890702_00		
Z SESARMA_(MEGALOPA)	61890702_33		
Z SESSILINA	3531		
Z SIALIS	64060101		
Z SIDA_CRYSTALLINA	6109010301	332	T570
Z SIGARA	62720103		
Z SIPHONOPHORA	3715	379	
Z SKAPTOPUS_BRYCHIUS	6169540101		
Z SMITTIA	65050848		
Z SOLENIDAE	551529	54	
Z SPHOEROIDES_(LARVAE)	88610102_97		
Z SPHOEROIDES_(VIABLE_EGG)	88610102_00		
Z SPHOEROIDES_MACULATUS_(LARVAE)	8861010201_97	168	
Z SPHOEROIDES_MACULATUS_(VIABLE_EGG)	8861010201_00		
Z SPINULOGAMMARUS	61692120		
Z SPIONIDAE	500143	43	
Z SPIOPHANES_BOMBYX	5001431001	42	
Z SPIROLOCULINA_CADUCA	3452100105		
Z SQUILLA_EMPUSA_(MEGALOPA)	6191010101_33		
Z SQUILLA_EMPUSA_(ZOEA)	6191010101_31	138	
Z STAUROPHRYA	35340307		
Z STEMPELLINA_BAUSEI	6505106401		
Z STENDSMELLA	35400301		
Z STENELMIS	63160401		
Z STENOCHIRONOMUS	65050830		
Z STENOTHIDAE	616948	269	
Z STENTOR	35390201		
Z STICTOCHIRONOMUS	65050841		
Z STROMBIDIDAE	353903		
Z STROMBIDIUM	35390301		
Z STYLARIOIDES_ERUCA	5001540309		
Z STYLICAUDA_PLATENSIS	3540050501		
Z STYLICLETODES_LONGICAUDATUS	6119273501		
Z STYLICLETODES_LONGICAUDATUS_(ADULT)	6119273501_98		
Z STYLICLETODES_LONGICAUDATUS_(COPEPODITE)	6119273501_12		
Z SUCTORIA	3533		
Z SYLLIDAE	500123	44	
Z SYMPHURUS_PLAGIUSA_(LARVAE)	8858020101_97	167	
Z SYNCHAETA	45061302		
Z SYNCHAETA_OBLONGA	4506130206		
Z SYNCHAETA_PECTINATA	4506130204		
Z SYNCHAETA_STYLATA	4506130207		
Z SYNGNATHUS_FUSCUS_(LARVAE)	8820020103_97	154	
Z SYNGNATHUS_FUSCUS_(LARVAE)	820020103_97		
Z TACHIDIUS_DISCIPES	6119140302		
Z TACHIDIUS_DISCIPES_(ADULT)	6119140302_98		
Z TACHIDIUS_DISCIPES_(COPEPODITE)	6119140302_12		
Z TANYPUS	65050834		
Z TARDIGRADA	75	204	
Z TEMORA	61182003	345	
Z TEMORA_(ADULT)	61182003_98		
Z TEMORA_(COPEPODITE)	61182003_12		
Z TEMORA_LONGICORNIS	6118200303	68	

Z	TEMORA_LONGICORNIS_(ADULT)	6118200303_98		
Z	TEMORA_LONGICORNIS_(COPEPODITE)	6118200303_12		
Z	TEMORA_LONGICORNIS_(NAUPLII)	6118200303_11		
Z	TEMORA_STYLIFERA	6118200301	259	
Z	TEMORA_STYLIFERA_(ADULT)	6118200301_98		
Z	TEMORA_STYLIFERA_(COPEPODITE)	6118200301_12		
Z	TEMORA_TURBINATA	6118200304	391	T2619
Z	TEMORA_TURBINATA_(ADULT)	6118200304_98		
Z	TEMORA_TURBINATA_(COPEPODITE)	6118200304_12		
Z	TEREBELLIDAE	500168	247	
Z	TESTUDINELLA	45070101		
Z	TESTUDINELLA_PATINA	4507010101		
Z	THALAICEAN	8407	222	
Z	THALASSOSMITTIA	65050853		
Z	THEMISTO_LIBELLULA	6170011401		
Z	THORACICA_(NAUPLII)	6131_11		
Z	TINTINNIDIUM	35400101		
Z	TINTINNINA	3540		
Z	TINTINNOPSIS	35400201		
Z	TINTINNOPSIS_DADAYI	3540020105		
Z	TINTINNOPSIS_FIMBRIATA	3540020123		
Z	TINTINNOPSIS_KARAJACENSIS	3540020137		
Z	TINTINNOPSIS_MEUNIERI	3540020136		
Z	TINTINNOPSIS_NITIDA	3540020138		
Z	TINTINNOPSIS_RADIX	3540020129		
Z	TINTINNOPSIS_SUBACUTA-HUGE	3540020133		
Z	TISBE	61191301	233	
Z	TISBE_(ADULT)	61191301_98		
Z	TISBE_(COPEPODITE)	61191301_12		
Z	TISBE_FURCATA	6119130101	81	
Z	TISBE_FURCATA_(ADULT)	6119130101_98		
Z	TISBE_FURCATA_(COPEPODITE)	6119130101_12		
Z	TONTONIA	35390302		
Z	TORTANUS_DISCAUDATUS	6118300101	392	
Z	TORTANUS_DISCAUDATUS_(ADULT)	6118300101_98		
Z	TORTANUS_DISCAUDATUS_(COPEPODITE)	6118300101_12		
Z	TRACHELOCERCIDAE	351602		
Z	TRACHYPENAEUS	61770102		
Z	TRACHYPENAEUS_(ZOEA)	61770102_31	325	
Z	TRACHYPENAEUS_CONSTRICTUS	6177010201		
Z	TRACHYPENAEUS_CONSTRICTUS_(ZOEA)	6177010201_31	96	
Z	TRICHOCERCA	45060701		
Z	TRICHOCERCA_CYLINDRICA	4506070102		
Z	TRICHOCERCA_LONGISETA	4506070103		
Z	TRICHOCERCA_SIMILIS	4506070105		
Z	TRICHOPTERA	6418		T576
Z	TRICHOTANYPUS	65050809		
Z	TRICHOTRIA	45060115		
Z	TRICHOTRIA_TETRACTIS	4506011501		
Z	TRINECTES_MACULATUS_(LARVAE)	8858010101_97	165	
Z	TRINECTES_MACULATUS_(VIABLE_EGG)	8858010101_00	166	
Z	TROPOCYCLOPS	61200807		
Z	TROPOCYCLOPS_(ADULT)	61200807_98		
Z	TROPOCYCLOPS_(COPEPODITE)	61200807_12		
Z	TROPOCYCLOPS_PRASINUS	6120080701		T2345
Z	TROPOCYCLOPS_PRASINUS_(ADULT)	6120080701_98		
Z	TROPOCYCLOPS_PRASINUS_(COPEPODITE)	6120080701_12		
Z	TURBELLARIA	3901	256	T578
Z	TURRITOPSIS_NUTRICOLA	3703020401	5	
Z	UCA_(MEGALOPA)	61890902_33	135	
Z	UCA_(ZOEA)	61890902_31	133	
Z	UCA_MINAX_(MEGALOPA)	6189090201_33		
Z	UCA_MINAX_(ZOEA)	6189090201_31	363	
Z	UCA_PUGILATOR_(MEGALOPA)	6189090203_33		
Z	UCA_PUGILATOR_(ZOEA)	6189090203_31		
Z	UCA_PUGNAX_(MEGALOPA)	6189090202_33		
Z	UCA_PUGNAX_(ZOEA)	6189090202_31		
Z	UPOGEBIA_AFFINIS	6183040102		
Z	UPOGEBIA_AFFINIS_(ZOEA)	6183040102_31	104	
Z	URANOTAENIA_SAPPHIRINA	6504031401		
Z	UROTAENIA_SAPPHIRINA	6505030501		
Z	VERRUCA	61330101		
Z	XANTHIDAE_(MEGALOPA)	618902_33		
Z	XANTHIDAE_(ZOEA)	618902_31	124	
Z	XENOCHIRONOMUS_XENOLABIS	6505089205		
Z	YOLDIA_LIMATULA	5502040511	46	

Z	ZANCLEA	37031401		
Z	ZAUSODES_ARENICOLUS	6119100401	134	
Z	ZAUSODES_ARENICOLUS_(ADULT)	6119100401_98		
Z	ZAUSODES_ARENICOLUS_(COPEPODITE)	6119100401_12		
F	ANCHOA MITCHILLI	8747020202		T25
F	ANGUILLA_ROSTRATA	8741010101	405	
F	BOTHIDAE	885703	414	
F	BREVOORTIA_TYRANNUS	8747010401	341	
F	CLUPEIDAE	874701		T11
F	GASTEROSTEUS_ACULEATUS	8818010101	408	
F	GOBIOSOMA BOSCI	8847010601		T90
F	GOBIOSOMA BOSCI(JUV)	8847010601_04		T135
F	HYPORAMPHUS_UNIFASCIATUS	8803010301	319	
F	LEPOMIS_MACROCHIRUS	8835160504	224	
F	MENIDIA	88050203		T1975
F	MICROPOGON UNDULATUS	8835440701		T84
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F	MORONE SAXATILIS	8835020102		T1
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F	MYROPHIS_PUNCTATUS_LEPTOCEPHALUS	8741130802LEPTO	280	
F	PARALICHTHYS_DENTATUS	8857030301	413	
F	POMATOMUS_SALTATRIX	8835250101	415	
F	PSEUDOPLEURONECTES_AMERICANUS	8857042001	378	T97
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