Lecture Notes on
Coastal and Estuarine
Studies

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23 (1987)

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Ecological Studies in the Middle Reach of Chesapeake Bay

Calvert Cliffs

Springer-Verlag
Berlin Heidelberg New York London Paris Tokyo

SUMMARY AND CONCLUSIONS

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The principal use of long-term environmental data records is to help formulate hypotheses about how ecological systems work. Long-term records are especially useful when a phenomenon under study is intermittent or has a relatively long periodicity. Long-term records also provide a standard against which the effects of experimental manipulations may be evaluated (Likens, 1983; Coull, 1985). The data derived from the long-term studies near Calvert Cliffs are first summarized and then the hypotheses about the ecological interactions indicated by these data are discussed. The effects of the construction and operation of the CCNPP on the Bay environment are evaluated and, finally, suggestions are made for future monitoring studies.

Ecosystem Structure and Function

Benthic-Pelagic Coupling

Consumer populations in the study area are largely dependent upon planktonic production, since macrophytes are nearly nonexistent near Calvert Cliffs (Orth and Moore, 1983). In this plankton-dominated system, zooplankton grazing does not control populations of primary producers (Sellner, 1983; Sellner and Horwitz, 1983; Sellner and Olson, in press), but instead, as in other temperate estuaries (Riley, 1956; Hargrave, 1980; Malone et al., 1983), large quantities of ungrazed phytoplankton cells are found in bottom waters. During annual spring blooms diatoms reach large population sizes and deplete water column silica to limiting concentrations, after which elevated densities of diatoms are found in bottom waters, presumably from sedimentation. Given the rates of production that occur during summer and fall, ambient dissolved inorganic nitrogen (DIN) appears limiting in the water column. However, water column regeneration processes may be sufficient to provide enough ammonium nitrogen to meet calculated requirements in nearly every month.

The vertical distribution of nutrients, oxygen and phytoplankton suggests frequent mixing of the water column in the study area, in contrast to the stratification seen in the deeper main channel of the Bay (Taft et al., 1980; Officer et al., 1984). Dissolved oxygen concentration and water density changes at the study sites indicate the frequent intrusion of low DO, high-density bottom water. The upwelling of bottom water produced as westerly winds move surface water away from shore (Carter et al., 1978), enriches the western shelf in the vicinity of Calvert Cliffs with elevated nutrient concentrations typical of hypoxic/anoxic bottom waters of the Bay.

Blue-green algal abundance in samples from 1980-1981 has increased greatly over abundances seen in the period 1975-1979. This increase could be due to changes in preservation techniques or better identifications, although it may signal a real shift in phytoplankton species composition, perhaps as a response to increasing nutrient enrichment of the Bay system as shown by Flemer et al. (1983). However, there has been no noticeable increase in cell density or chlorophyll *a* concentration that would signal high levels of nutrient enrichment.

Zooplankton abundance is not clearly related to estimates of phytoplankton abundance, which is consistent with the observation that much phytoplankton biomass is unconsumed in the water column. There is some evidence to suggest that zooplankton standing stock has been reduced since Tropical Storm Agnes in 1972 and this reduction may account for the relatively low grazing rates observed. Abundance estimates of potential predators such as ctenophores and plankton-feeding fish were not often inversely related to zooplankton abundance, as had been expected. Moreover, in cases where correlations were significant, these two variables were positively correlated. These facts suggest that predators do not control zooplankton populations. It is possible, however, that we have not sampled at short enough time intervals to observe decreases in zooplankton density in response to predation. Because food does not appear to limit zooplankton populations in the study area, predation seems likely to have a role in regulating zooplankton populations, even though there is little evidence to support this statement.

Estimates of filter-feeding rates by benthic molluscs suggest that they may be important consumers of phytoplankton in the study area, possibly consuming, according to the season, anywhere from 1% to 74% of phytoplankton cells. The major summer decline observed in benthic populations has previously been attributed to the effects of predation, especially by bottom-feeding fish and blue crabs, or to low dissolved oxygen (Holland et al., 1980). Significant negative correlations between blue crab abundance and polychaete and bivalve biomass are consistent with the notion of predator control, although no significant relationship exists between bottom-feeding fish abundance and infaunal biomass or abundance.

An alternative explanation for the summer decline in benthic populations is suggested by the reduced phytoplankton populations in summer after large spring diatom blooms; namely, that food limitation caused by reduced phytoplankton sedimentation is responsible for the summer decline in benthos. The fall increase in benthic abundance also coincides with the increased abundance of phytoplankton biomass following the late summer blooms. In this hypothesized sequence of benthic-water column coupling, the depletion of silica by spring diatom populations begins the cycle of boom and bust of both plankton and benthic populations with rates of silica regeneration (benthic regeneration rates yield 1.1 $\mu g/at \cdot L^{-1} \cdot d^{-1}$ in June (D'Elia et al., 1983) limiting diatom production in the spring. This leads to a decrease in benthic populations and nutrient regeneration from the sediments due to reduced bioturbation (Aller, 1980, 1983; Kristensen, 1984) and, therefore, nutrient supply for primary producers in surface waters.

While Chesapeake Bay harvests of oysters have declined markedly over the past 50 years (Flemer et al., 1983), there is little evidence of decline in oyster populations near Calvert Cliffs during the study period. Instead, even though sufficient food seems available for oyster growth, poor recruitment and lack of hard substrate have consistently limited oyster populations. Only in 1980 and 1981, after completion of a program to add cultch to soft bottom in the Calvert Cliffs area, was there spatfall sufficient that commercially harvestable densities could appear in the area.

Soft-shell clam populations in the study area have never been large enough to support commercial harvesting. Over the years, some large recruitments have been observed, but subsequent mortality precluded large adult populations. The relatively low average density of soft-shell clams near Calvert Cliffs is likely due to high predation rates, sediment instability, low summer DO concentrations, food limitation or some combination of these factors.

Blue crab populations in the Calvert Cliffs area have shown major fluctuations in abundance that were significantly correlated with those shown by commercial catch records. Over the study period, there is no evidence that crab mortality caused by low DO is occurring more frequently at the sampling depths of 2.5 to 4.6 m. This suggests that the well-mixed shallow waters of the shelf in the Calvert Cliffs area have not been experiencing increasingly frequent occurrences of very low oxygen concentrations as reported for the Bay's main channel (Taft et al., 1980) and as seen at greater depths in the study area.

An unexplained trend is the significant decline of blue crab males from 66 to 49 percent of the catch over the period 1968-1983. There is no obvious reason for this, such as persistent salinity increases at the study sites, or the selective release of captured females. The financial consequences for the industry may be serious, however, since during the crabbing season the value of male crabs may be two to three times that of females. But even if the trend continues, the resulting female-biased sex ratios pose no threat to population persistence.

During the study period, fish populations showed large fluctuations and changes in species composition, many of which are readily interpretable. For example, several species characteristically associated with submerged vegetation (e.g., *Apeltes quadracus*) declined during the study, mirroring the decline in submerged vegetation in this part of the Bay (Orth and Moore, 1983). Striped bass (*Morone saxatilis*) declined markedly during the early 1970s after a strong year class in 1970 that was not followed in subsequent years by significant recruitment events. Abundance of menhaden (*Brevoortia tyrannus*) increased over the period, probably in response to lowered fishing pressure in offshore waters (Grosslein and Azarovitz, 1982).

Measurements of DO during fish trawling indicated declining levels at 9 m and 12 m during August throughout the study period, and also in June and July 1980 and 1981. These data are consistent with the proposal that anoxia and hypoxia are increasing in Cheasapeake Bay (Flemer et al., 1983; Officer et al., 1984). It has been suggested that anadromous fish stocks have been declining while coastal-spawning stocks may be increasing in the Bay

(Rothschild et al., 1981), and Officer et al. (1984) have hypothesized that these changes may be due to increasing environmental stress, especially in the form of low DO concentrations. Our data do not completely support this, since one anadromous species (the blueback herring, *Alosa aestivalis*) has increased in abundance in 1980 and 1981, and another (the alewife, *A. pseudoharengus*) showed high abundance in 1978. Moreover, there is a good reason to believe that increases in ocean-spawned menhaden may be due to reduced fishing pressure.

The Environmental Protection Agency has recently summarized the results of its massive $27 million program to assess the health of the Chesapeake Bay (Flemer et al., 1983). Data from the Calvert Cliffs studies lend support to the general conclusions of Flemer et al. (1983) and Officer et al. (1984), that eutrophication is increasing throughout the Chesapeake Bay. The increasing presence of blue-green algae in plankton samples in recent years, and the increasing frequency of low DO concentrations at depths of 6 m and 9 m are consistent with accelerating rates of eutrophication in the study area. However, conclusions regarding the fin fishes, especially regarding the decline of anadromous species, are not supported by the Calvert Cliffs data.

Impact of the Calvert Cliffs Nuclear Power Plant

Our studies have shown significant changes in the density and composition of phyto- and zooplankton as a result of entrainment in the CCNPP cooling water system. Microflagellates and copepods are most susceptible to damage. Both inhibition and stimulation of production were observed as entrainment effects upon phytoplankton, but monitoring studies found few differences between plankton assemblages in the vicinity of the CCNPP discharge and those at reference stations.

Survival studies showed that from 19% to 98% of impinged fish survive, depending on the species. Based on these data, losses of commercially harvested fishes from operation of the CCNPP were less than 0.1% of commercial landings. When recreational landings of fish are included, the percent loss from CCNPP operation may be further reduced by more than 50%. Survival studies also showed greater than 99% survival of blue crabs. Taken together, these data indicate that the plant's harvest of commercial species is a very small fraction of mortality from all sources.

Thermal effects are minor, due to relatively low rise in temperature (5.5 to 6.7°C) during plant transit, the short time in transit, and the very large volume of receiving waters available for dilution. There is some evidence of stimulated growth in oysters near the discharge, but no evidence of decreased growth or mortality. Winter fish kills during plant shutdown have never been observed, presumably because the heated effluent does not cover a large enough area to induce fish to overwinter near the discharge. The velocity of the discharge has removed fine-grained sediments from a 17-hectare area that now supports an assemblage of epifaunal species instead of its former infauna of mud-dwellers. The installation of titanium and stainless steel condenser tubing in 1979, and the absence of chlorine reduces the likelihood of any impacts due to chemical additions from the plant. Radioisotopes of silver and cobalt have been detected in the Bay, but at very low levels, and the State of Maryland has concluded that there is no health risk from the presence of these materials.

In addition to the studies described in this volume, other studies to detect impact resulting from operations of the CCNPP have been funded by the federal and state governments, as well as by the Baltimore Gas & Electric Company. Enormous effort has gone into these studies, and the entire body of work at Calvert Cliffs was reviewed in 1980 (Martin Marietta Corporation, 1980). The conclusion is that detectable impacts from plant operation are surprisingly small. Subsequently, after reviewing a compilation of impact assessment studies at CCNPP (ANSP, 1981), the State of Maryland determined that Calvert Cliffs Nuclear Power Plant was in compliance with state water quality regulations.

Impact Assessment Techniques

In attempting quantitative analyses of impact in an open system such as Chesapeake Bay, one of the first and most difficult tasks is to define the population about which inferences are to be drawn. This difficulty can be shown by an example at Calvert Cliffs where it was not clear how to define the population of spot (*Leiostomus xanthurus*) so that an estimate could be made of the percent of the spot population cropped by the power plant. Should all spot in the middle portion of the Bay be considered the population? The spot in the entire Bay? Or all spot along all of the Atlantic coast? Without direct knowledge of spot breeding habits, it is very difficult to decide what constitutes the population; it is more difficult to estimate how power plant-induced mortality might affect the population. Because of uncertainties in predicting population, consequences of power plant-induced mortality, and the controversies surrounding specific models, management and regulatory decisions might alternatively based on comparisons of power plant effects with magnitudes of inputs and losses from other sources rather than on detailed, but unverifiable, models (cf. Barnthouse et al., 1984).

The detection of differences in abundance or biomass among stations is a function of the number of stations sampled, the number of replicates taken, and the power of the statistical test. A well-developed statistical theory allows the investigator to determine the desired levels of difference to be detected (cf. Cochran, 1963). A sampling scheme can be designed so that mean differences of, for example, 10% can be detected in a variable among sampling stations. The problem in applying this statistical theory to data from biological monitoring programs is that the biological data usually violate the assumptions of the underlying model. Of particular importance is the fact that biological samples are serially correlated and not independent (Eberhardt, 1976; Heck and Horwitz, 1984). Thus, even though it is important to consider and estimate minimum detectable differences, one must be cautious in applying these methods to biological field data.

To illustrate, consider the process of arriving at minimum detectable differences among stations by evaluating the results of fish trawling studies for a 5-year period (1969-1973). If we assume a one-way ANOVA model in which samples are considered to be randomly taken at each of the three fish trawling stations during each month from 1969-1973, then the tabulation can be interpreted in the following manner. The A Table shows the sample size needed to detect a given percent difference in abundance with power of 0.5, 0.7, or 0.9 between the Plant Site and the two reference sites (Kenwood Beach and Rocky Point) at both 0.05 and 0.01 levels. (This table was constructed assuming that both reference sites have greater abundances than the Plant Site). If we wish to detect a 30% decrease in log-transformed abundance at the Plant Site and find it statistically significant at the 0.05 level, a sample size of 41 is needed for a power of 0.5, a sample size of 62 for a power of 0.7, and a sample size greater than 100 for a power of 0.9. Now, the actual number of samples was 59 and the section B of the table lists the probability of detecting certain differences given our actual sample sizes. For example, a 30% difference in log-transformed abundance can be detected between plant and reference stations 70% of the time at the 0.5 level and 47% of the time at the 0.01 level.

"Power" estimate for logarithms of fish trawl abundances, 1969-1973. One-way model, log transformed variables, based on the assumption that reference sites contain greater abundance than the Plant Site. (A) Sample sizes required to detect specified decreases in log-transformed abundance between reference and impact stations, under preset levels of alpha and beta. (B) Probability of detecting specified difference in log-transformed fish abundance between reference and impact stations, with a sample size n=59, and levels of 0.05 and 0.01.

	% Decrease													
		10	15	20	25		30		40		50		60	
					.05	.01	.05	.01	.05	.01	.05	.01	.05	.01
Power (1-β)														
A.	0.5	>	>	96	56	88	41	70	25	42	20	34	17	26
	0.7	>	>	>	80		62	100	38	60	29	44	23	39
	0.9	>	>	>	>	>	>	>	60	87	48	65	39	57
Alpha Level														
B.	0.05	0.14	0.26	0.37	0.54		0.70		0.88		0.96		0.99	
	0.01	0.05	0.11	0.18	0.33		0.47		0.72		0.81		0.95	

> = Sample size greater than 100.

These examples should not be taken literally because many caveats must be considered in interpreting such information, and there are other models that could also be used. Nevertheless, the examples discussed here indicate general trends and suggest that monitoring programs, even those lasting 5 years, are unlikely to detect relatively small (in the 10% to 15% range) biomass or abundance reductions. In fact, the results of these calculations led to increased sampling effort in the Calvert Cliffs fish trawling studies.

It is likely in other studies, as in the fish example, that levels of detectability of impact will be relatively high and as a result there may be uncertainty concerning the magnitude of power plant impact. In our sampling effort, there could be from 0 to as much as 30% reduction in the logarithm of fish abundance and we would detect this only 70% of the time at the 0.05 level.

Power calculations have not been carried out for all our programs because it was determined that strict reliance on these estimates was unwarranted, due to violations of assumptions in the underlying ANOVA model. Nevertheless, these calculations point out the potential uncertainties that may exist in impact estimates in the Calvert Cliffs studies, as well as in other impact assessment studies.

Should future impact-assessment programs employ the strategy used in the Calvert Cliffs studies, or has our experience produced results that suggest an alternative approach? The rationale for the Calvert Cliffs studies -- to monitor as many trophic groups as possible and to measure the important physico-chemical variables to which the biota would respond -- intuitively seems correct and difficult to criticize. And this strategy has characterized most large impact assessment studies done in the United States during the past two decades. Yet, as was recognized early on in the analysis of impact assessment studies, the difficulties in defining populations and in carrying out sufficiently intensive sampling programs to detect less than catastrophic changes in ecosystem structure (Eberhardt, 1976; Thomas, 1977), suggest a reevaluation of this approach.

An alternative strategy accepts the limitations of our ability to predict the future consequences of man's activities on biological populations and aims to focus monitoring efforts on those organisms most likely to show detectable changes, or on those organisms of special importance to the system under study. Studies can then be designed to approach specific questions that are relevant to the biology of the target group and that can be answered with feasible levels of effort. For example, comparisons of abundances of organisms between impact and reference sites, or between pre-operational and post-operational periods should be concentrated on groups with population structure and sampling properties allowing statistically powerful comparisons.

However, organisms such as keystone predators or dominant primary producers may be chosen for study even if their population fluctuations and the required sampling program are large. The purpose of these studies should be to establish magnitudes of direct impacts (e.g., entrainment, impingement, thermal mortalities), magnitudes of standing crop, production and/or other losses with which power plant impacts may be compared, and factors of the life history which may affect the organisms' vulnerability to power plant impacts. Because of the importance of these groups in the system, results will also be useful in assessing changes in other groups. But, because one of the major goals of monitoring is to develop testable hypotheses, the testing of these hypotheses should also form an essential component of impact assessment studies. Hypotheses may be tested through experimentation in both field and laboratory. Reciprocal transplants of artificial substrates might be made between impact and reference sites to evaluate the effects of pollution on the associated fouling species. Laboratory ecosystems can also be developed and subjected to varying treatments that simulate impact. For some larger species, experimentation is impossible or unethical and clever sampling designs are needed to evaluate properly posed alternative hypotheses.

The results of the Calvert Cliffs studies suggest that future impact assessment studies can be improved by directing attention to sensitive species, especially those that do not usually experience very large population fluctuations. By focusing on a subset of taxa a higher level of resolution of impact effects may result. The challenge facing future environmental impact studies is to combine field observations and experimental results so skillfully that the results withstand careful scrutiny and alternative hypotheses can be rigorously evaluated.

Literature Cited

Academy of Natural Sciences of Philadelphia (ANSP) (1981) Assessment of thermal, entrainment and impingement impacts on the Chesapeake Bay in the vicinity of the Calvert Cliffs Nuclear Power Plant. Report No 81-10: PA: Acad Nat Sci Phila: 297 pp.

Aller RC (1980) Diagenetic processes near the sediment-water interface of Long Island Sound. II: Fe and Mn. Adv in Geophysics 22:351-415

__________ (1983) The importance of the diffusive permeability of animal burrow linings in determining marine sediment chemistry. J Mar Res 41:299-322

Barnthouse LW, Boreman J, Christensen SW, Goodyear C, Van Winkle W and Vaughan DS (1984) Population biology in the courtroom: The Hudson River controversy. Bioscience 34:14-19

Carter HH, Regier RJ, Schiemer EW and Michael JA (1978) The summertime vertical distribution of dissolved oxygen at the Calvert Cliffs Generating Station: a physical interpretation. Spec. Rpt. 60. Ref. 78-1, CBI. Baltimore, MD: Johns Hopkins: 95 pp.

Cochran WG (1963) *Sampling techniques*. New York: Wiley and Sons. 273 pp.

Coull BC (1985) The use of long-term biological data to generate testable hypotheses. Estuaries 8:84-92

D'Elia CF, Nelson DM and Boynton WR (1983) Chesapeake Bay nutrient and plankton dynamics: III: The annual cycle of dissolved silicon. Geochim Cosmochim Acta 47:1945-1955

Eberhardt LL (1976) Quantitative ecology and impact assessment. J Environ Manage 4:27-60

Flemer DA, Mackiernan GB, Nehlsen W and Tippie VK (1983) Chesapeake Bay: a profile of environmental change. US EPA, Region 3: Philadelphia, PA: 200 pp.

Grosslein MD and Azarovitz TR (1982) Fish distribution. MESA New York Bight Altas Monograph 15. NY: Sea Grant Institute of State University of New York and Cornell University: 182 pp.

Hargrave BT (1980) Factors affecting the flux of organic matter to sediments in a marine bay. In: Tenone KR and Coull BC (eds) *Marine benthic dynamics*. Columbia, SC: Univ. South Carolina Press: pp. 243-263

Heck KL Jr and Horwitz RJ (1984) Statistical analysis of sampling data to assess impact in marine environments. In: White HH (ed) *Concepts in marine pollution measurement.* College Park, MD: Maryland Sea Grant: University of Maryland: pp. 233-245

Holland AF, Mountford NK, Hiegel MH, Kaumeyer KR and Mihursky JA (1980) Influence of predation on infaunal abundance in Upper Chesapeake Bay, U.S.A. Mar Biol 57:221-235

Kristensen E (1984) Effect of natural concentrations on nutrient exchange between a polychaete burrow in estuarine sediment and the overlying water. J Exp Mar Biol Ecol 75:171-190

Likens GE (1983) A priority for ecological research. Bull Ecol Soc Amer 64:234-243

Malone TC, Hopkins TS, Falkowski PC and Whitledge TE (1983) Production and transport of phytoplankton biomass over the continental shelf of the New York Bight. Cont Shelf Res 1:305-337

Martin Marietta Corporation (1980) Summary of findings: Calvert Cliffs Nuclear Power Plant aquatic monitoring program: Vol I: Rpt. PPSP-CC-80-2. Baltimore, MD: Martin Marietta Corporation Environmental Center

Officer CB, Biggs RB, Taft JL, Cronin LE, Tyler MA and Boynton WR (1984) Chesapeake Bay anoxia: Origin, development and significance. Science 223:22-27

Orth RJ and Moore KA (1983) Chesapeake Bay: an unprecedented decline in submerged vegetation. Science 222:51-53

Riley GA (1956) Oceanography of Long Island Sound, 1952-1954: IX: Production and utilization of organic matter. Bull Bingham Oceanogr Coll 15:324-343

Rothschild BJ, Jones PW and Wilson JS (1981) Trends in Chesapeake Bay fisheries. In: *Improving management of Chesapeake Bay.* Trans. 46th North American Wildlife and Natural Resources Conf Washington DC: Wildlife Management Institute: pp 284-298

Sellner KG (1983) Plankton productivity and biomass in a tributary of the upper Chesapeake Bay: I: Importance of size-fractionated phytoplankton productivity, biomass and species composition in carbon export. Est Coastal Shelf Sci 17:187-206

__________ and Olson MM (In Press) Copepod grazing of red tides in Chesapeake Bay. Third Int Conf in Toxic Dinoflagellates. New York: Elsevier

__________ and Horwitz RJ (1983) Plankton interactions in the Patuxent River estuary: Field studies of community composition and density, with a deterministic model of the production of fecal matter, sediment oxygen demand and nutrient regeneration. Rept No 82-14F. PA: Acad Nat Sci Phila: 119 pp.

Taft JC, Taylor WR, Hartwig EO and Loftus R (1980) Seasonal oxygen depletion in Chesapeake Bay. Estuaries 3:242-247

Thomas JM (1977) Factors to consider in monitoring programs suggested by statistical analysis of available data. In: Van Winkle W (ed.) *Assessing the effects of power-plant-induced mortality on fish populations.* New York: Pergamon Press: pp 243-255