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SAMPLING DESIGN FOR AQUATIC ECOLOGICAL MONITORING PHASE I REPORT

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Technical Report No. 85
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PHASE I REPORT

April 1982
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EXECUTIVE SUMMARY

Electric power generating facilities impact the aquatic environment by the discharge of waste heat, disposal or dilution of plant wastes, impingement and entrainment of organisms in cooling water discharge systems, and toxic effects of biocides used to control biofouling of cooling systems. The nature and extent of power plant effects on the aquatic environment depend on the design of the plant and the type of water body on which the plant is located.

Monitoring programs are required at power plants to ascertain whether plant impacts are within acceptable limits. The primary legislative authority for monitoring lies in the National Environmental Policy Act and in the Clean Water Act. Under the provisions of these laws, monitoring programs may be required to meet National Pollutant Discharge Elimination System permit conditions and U.S. Nuclear Regulatory Commission Environmental Technical Specifications.

Monitoring may also be mandated under state environmental legislation. In most cases regulatory agencies do not require a specific monitoring design, but rather a demonstration that the monitoring program is adequate to meet regulatory requirements. Consequently many different monitoring strategies have been employed, ranging from highly focused programs to general 'measure everything' approaches. A review of 162 monitoring programs showed that 19 variables were measured frequently and that these primarily evaluated impacts on organisms affected by waste heat discharges or entrainment/impingement. Most of these variables evaluated short-term local impacts; few attempts have been made to evaluate long-term population or ecosystem level effects.

In testing short-term effects most existing monitoring programs are spatially intensive in the area of the discharge and/or intake, with sampling repeated several times per year (often monthly) for several years. A typical program would be initiated two or three years prior to plant operation and would continue an additional two or three years into the operating period, although a number of examples exist where no pre-operational data have been collected.

The most commonly used tool to analyze monitoring data is the analysis of variance, which is usually applied in a factorial design, where plant status (pre-operational or operational), station location (control or affected) and, perhaps, time of year and depth are the factors. Major difficulties with such designs are that status is not a surrogate for plant effect and that natural year-to-year and season-to-season variability cannot be adequately distinguished from plant effect. Therefore, factorial designs are primarily useful for determining acute local effects and not for evaluating population level, chronic effects.

A review of previous attempts to apply optimization techniques to cost-effective design of sampling programs revealed that the earlier efforts generally emphasized allocation of sampling effort to obtain specified precision of estimates of mean values rather than maximizing the detectability of change. Also, there is a lack of adequate cost data to allow implementation of cost-effectiveness methods in monitoring design. A major need exists to develop methodologies to allow decisions regarding trade-offs between allocation of monitoring effort to short-term local effects and long-term population level effects.

Finally, a review of computer software for the analysis, display and storage of aquatic data revealed that SAS (Statistical Analysis System) is the most widely used software package. The technology to permit communication between computers is evolving rapidly and will be valuable for future monitoring data analysis. However, this is an industry-wide problem which is felt to be beyond the scope of this project. A major need that can be addressed is development of interactive software systems that can display monitoring network designs and provide statistical measures of their effectiveness in detecting long-term change.

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PREFACE

Utility companies are required to have extensive and costly pre-operational and operational phase environmental monitoring at sites approved for development. The purpose of this project is to develop methodologies that can reduce the cost of such monitoring efforts while increasing the value of the data collected. A four phase research effort is planned for this project: Phase I is a review of current practice; Phase II is the development of methods; Phase III is an application of the developed methodology at two or more sites; and Phase IV is information transfer to the utility industry.

Phase I findings are reported in this document. They are based on computerized literature searches, a review of literature in Electric Power Research Institute (EPRI) files, reviews of journal literature based on the knowledge of the project team members, interviews with utility managers responsible for aquatic monitoring and power plant impact assessment, and reviews of literature identified by these interviews. Table P.1 briefly summarizes the type and quantity of literature reviewed. Almost five hundred citations were acquired and evaluated in the process of preparing this report.

Computerized literature abstract services were used to access the 'gray' literature, including reports by consultants, government agencies, and utilities. The computerized systems used include the Utility Data Institutde (UDI) system, the Electric Power Data Base (EPRI), the National Technical Information Service, the U.S. Department of Energy, and the COMPENDEX (Engineering Index) systems. Searches were conducted under a variety of key words, such as Ecological Effects, Sampling, Aquatic Environment, and others. In addition to the computerized searches, more traditional reviews of the refereed literature were made.

Table P.1 Sources of Documents in Aquatic Monitoring Project Library

Document Source	Type of Document: Journal Articles	Type of Document: Technical Reports	Total	Percent of Total
Computer search	50	44	94	19
Search by team members (manual)[a]	79	91	170	34
Documents published by EPRI	9	42	51	10
Personal visit to document author	11	127	138	28
Personal visit to utility data institute		39	39	8
Other[b]		3	3	1
Total	149	346	495	100

[a]Includes documents obtained from researchers' personal collections

[b]Personal communications

Interviews were conducted during January and February 1982 with selected utilities, consultants, and government agencies with active interest in aquatic monitoring design (see Acknowledgments). These interviews were guided by a list of questions related to various aspects of the aquatic monitoring issue. The questions included were:

1. What system components have been monitored (historic and current trends) at the sites with which you are most familiar? Impacts considered were thermal, impingement/entrainment, and chemical discharge.
   - Ecosystem--phytoplankton, zooplankton, benthos, ichthyoplankton, juvenile fish, adult fish

2. What techniques have been employed to select sampling sites (historic and current trends)?
   - Regulatory requirements
   - Expert judgement
   - Statistical selection based on pilot studies
   - Models
   - How do techniques differ for different system components/ different impacts?

3. How are samples allocated, with respect to location, frequency, number of replicates (historic and current trends)?
   - What techniques determine allocation (regulatory require ments, expert judgement, statistical selection based on pilot studies, models)?
   - How do allocations and techniques employed in program design differ for different system components? Different impacts?

4. What statistical approaches have been employed in data analysis (across space and time; historic and current trends)?
   - Analysis of variance
   - Regression

- Advanced techniques
- Have techniques been incorporated in program design prior to implementation or applied ex post facto?
- How do you regard the question of probability distributions which are not normal?
- How do techniques differ for different system components? Different impacts?

5. Do you employ models in the post-data collection phase? If so, how is this done?

6. What techniques are employed in monitoring fish?
   - Sampling techniques
   - Expressions of population characteristics
   - Tissue analyses (including radionuclide accumulation)

7. What is the role of benthic monitoring?
   - Patchiness question
   - Usefulness of population vs. diversity measures

8. What would be your approach to analyzing power plant impacts in the absence of sufficient pre-operational data?

9. Have you used computers to:
   - Construct graphics interactively
   - Assist in communications (MAIL), especially among personnel in different disciplines

10. Have you made any use of photogrammetry or remote sensing in monitoring?

11. Have you conducted monitoring programs to discern the impacts of any of the following conditions:
    - Construction site runoff or effects of dredge fill?
    - Coal or flyash pile runoff or leachate?
    - SO_2 scrubber effluent or waste pile runoff or leachate?

12. What data are available concerning the costs of monitoring?
    - Setup costs/station selection
    - Operating costs
    - Cost-effectiveness (incremental cost for additional information)

13. For utilities--if not done by your own staff, with whom have you contracted to design monitoring programs?

14. What reports can you make available that concisely document experience relative to the issues raised above?

15. What data sets are available which would be amenable to case study later in the project?
    - In what form are the data (tape, cards, microfiche, paper)?
    - What are the characteristics of the programs generating the data (relative to selection of sampling sites, frequency, replication, length of record (pre-operational and operational)?

16. What has been your experience in monitoring at fossil-fueled vs. nuclear plants in terms of:
    - The extent of the programs (historically and currently)?
    - Regulatory control of the programs (historically and currently)?

This report is organized into three major sections. Chapter I examines the types of aquatic impacts studied by past environmental monitoring efforts. Chapter II review the objectives of environmental monitoring efforts and the process of defining objectives or hypotheses for such efforts. Finally, Chapter III reviews various techniques pertinent to monitoring programs, such as models, data management, sampling methods, statistical analyses, and cost-effective procedures which must be incorporated in a monitoring design protocol.

CHAPTER 1

AQUATIC IMPACTS OF ELECTRICAL GENERATION ASSOCIATED WITH FOSSIL FUEL AND NUCLEAR PLANTS

INTRODUCTION

The generation of electricity by fossil (coal, oil or gas) and nuclear fuels requires large quantities of water for cooling, for the dilution of water discharged if cooling towers are not used, and for the disposal or dilution of miscellaneous plant wastes (Parker and Krenkel, 1969; Hu et al., 1978). A typical generating station with an open-cycle cooling system and a 1,000 MWe capacity requires a flow of approximately 1,000 cfs for its cooling water (Utility Water Act Group, et al., 1978). As a result of these water uses, the operation of a power plant can impact or alter the aquatic system on which it is located in a variety of ways. In turn, the nature and extent of such alterations can vary depending upon the type of water ›dy on which the plant is located. Susceptible water bodies include lakes, rivers, estuaries, coastal (and offshore) marine environments, reservoirs (river impoundments) and/or cooling ponds (water bodies constructed specifically for cooling purposes).

This chapter reviews the findings of field and laboratory studies on the aquatic effects of thermal power generation. Field data were interpreted predominantly with parametric statistics (e.g., analysis of variance), although nonparametric and derived indices were used also. Regression models and mathematical, or simulation, models were used to develop predictions of the significance of observed changes. The results of these studies (primary sources) and interpretations made by experts in the field (secondary sources, including review articles and interviews with industry personnel) were used to develop the consensus opinion presented here.

SCOPE OF REVIEW

Once-through (open) cooling systems can impact the aquatic environment in three ways during operation: (1) impact due to thermal loading of the receiving water as heated effluents are discharged from the power plant, (2) impact due to impingement of organisms at or on the intake structures or entrainment of organisms into the condenser system with the cooling water, and (3) impact due to chemical loading of the receiving waters either as a result of chemicals leached from the condenser apparatus (e.g., copper) or as a result of the addition of chemicals to the cooling system for the control of biofouling organisms. The most commonly used biocide is chlorine, although other chemicals (and heat treatments) are used as well.

Impacts in the aquatic environment due to closed system cooling (cooling tower operation) include (1) intermittent or continuous removal of small volumes of water from the receiving system to replace water lost through evaporation from cooling structures (this removal represents a net loss of water volume or flow in the aquatic system;) (2) periodic or continual discharge of some fraction of cooling tower water and its associated chemicals (biocides, anticorrosion agents, erosion compounds and occasionally wetting agents) to remove fouling organisms, and salts that concentrate during operation (blowdown); and (3) reduced (as compared to once-through systems) levels of impingement and entrainment of organisms during water withdrawal.

Coal-fired power plants also can affect the aquatic environment as a result of on-site disposal of coal washing residues, flyash and/or heavy (bottom) ash. Typically, stack and burner wastes are treated on-site in settling ponds. These ponds act to remove toxic components (particularly heavy metals) that remain following the burning of fuel. Leachate from these ponds can drain into nearby surface waters, either deliberately or accidentally, and adversely impact the biota. Recently, leaching of selenium and other metals has been recognized as a potential problem associated with electrical generation (personal communication, Carolina Power and Light; Texas Utilities Service Co.).

Scrubbing of stack gases also generates acidic waste waters that can be discharged to the receiving system.

Monitoring programs and impact evaluations cross spatial/temporal boundaries. They generally have attempted to distinguish between first order (local) and higher order (system-wide) effects. First order effects are concentrated in the immediate plant vicinity or zone of discharge plume influence and are manifested only temporarily (i.e., in the short-term). Such effects might include the magnitude of mortality due to impingement or entrainment or the direct effects of higher temperatures or toxic chemicals. Higher order effects consider the long-term, system-wide implications of first order changes for populations that utilize habitats broader in area than the immediate thermal or chemical plume zone and for community stability and ecosystem structure and function. Such effects might include the extent to which thermally-induced increases in rates of primary production alter the structure of fish communities in cooling impoundments or an evaluation of increased larval cropping due to impingement on patterns of population recruitment and long-term stock size. In some instances these long-term effects may include areas outside the particular water body but inside the geographical zone of the population of interest. This is particularly true if migratory fish (e.g., striped bass, salmonids, etc.) are an important component of the system under consideration. Occasionally the combined effects of several plants have been considered. This chapter provides a summary of these aquatic impacts related to thermal power production. This summary is shown schematically in Figure 1.1.

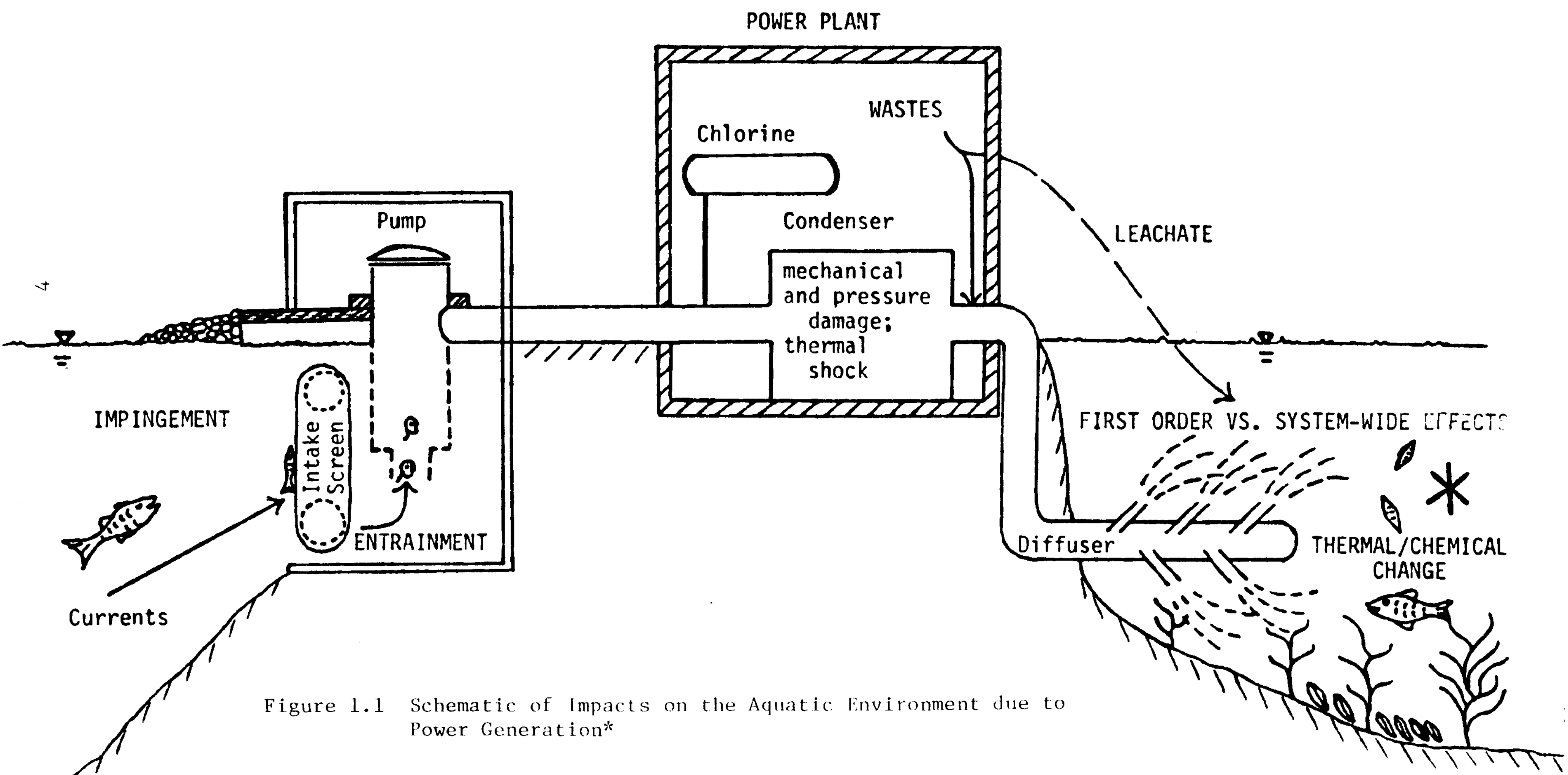

Figure 1.1 Schematic of Impacts on the Aquatic Environment due to Power Generation*

*following Coutant(1974)

IMPACTS DUE TO THERMAL LOADING

Thermal loading or calefaction refers to the input of excess heat to an aquatic system through the discharge of cooling water. The resultant effects or potential for effects received extensive (or perhaps excessive) publicity and generated much concern from both public and scientific communities during the late 1960s (Parker and Krenkel, 1969; Coutant and Brook, 1970). Some major fish kills and evaluations of the threat to sensitive populations (e.g., salmonids) in the early 1970s (Northeast Utilities Service Company, 1975; Jensen, 1974a; Hydrocomp, Inc., 1976) increased public awareness and may have contributed to the adoption of strict thermal standards for the nation's waters in 1972, which were revised in 1977 (see Chapter 2).

Since that time, scientific understanding of the first order effects of thermal loading due to power generation has improved and specific areas of legitimate concern have been identified. These include (1) impacts that occur in the zone of thermal plume influence, where heated effluents are not mixed with ambient water; (2) cooling systems in which the temperature change (T) experienced by entrained organisms (and at the immediate outfall) is much greater than $10^{\circ}C$; and (3) areas in which the addition of heat results in a final mixed temperature greater than $30^{\circ}C$ over a substantial part of the receiving water body (e.g., ratio of effluent volume to receiving water volume is high). These large temperature changes that occur during entrainment in the discharge area, and beyond the immediate dilution zone, are problematic both in temperate water bodies where cold water species (e.g., salmonids) dominate and in subtropical water bodies, where high ambient temperatures can push mixed temperatures beyond the tolerance limits of many warm water species. In water bodies where these conditions have occurred, short-term damage to specific populations of organisms has been observed frequently.

Specific changes in fish and/or macroinvertebrate fauna were observed in the discharge zone in many studies. For example, Cole (1973) showed evidence of changes in fish distribution in the discharge zone of a power plant on Lake Erie, but was unable to detect an overall change in fish abundance in the lake itself. Merriman and Thorpe (1976) observed decreased populations of fish near the outfall of the Connecticut Yankee Plant that were not extended into the mainstream of the Connecticut River. Ruelle et al. (1977) also found that some fish tended to decrease in abundance near a nuclear plant in South Carolina although, in this instance, the researchers were unable to attribute the changes to increased temperature. In many water bodies the resident species of fish appear able to avoid the heated zone and thereby decrease the potential for impact (Oak Ridge National Laboratory, 1981).

Carolina Power and Light Company (1979) observed decreased diversity in benthic fauna due to operation of the Robinson Steam Electric Plant, but noted that the effect was very local. Similarly, Wurtz (1979) found a depauperate macroinvertebrate fauna in the plume of Brunner Island Power Station over the period 1967-78, but found no overall effect in the main section of the Susquehanna River. In many studies, natural variability in benthic populations was so great that changes due to thermal loading could not be distinguished from natural variability (Martin Marietta Corp., 1980). At some sites, changes in invertebrate fauna were attributed to scouring of bottom substrates from high velocity discharges rather than to temperature changes (Merriman and Thorpe, 1976).

Localized damage to phytoplankton populations was observed at the Marshall Steam Plant on Lake Norman, North Carolina (Jensen, 1974b), but again these effects did not extend to the far-field. Lawler, Matusky and Skelly Engineers (1980b) found localized effects on planktonic populations in many smaller lakes and ponds constructed specifically for cooling purposes, but argued that such impacts should be acceptable due to the nature of the water body.

Some longer term changes due to widespread higher temperatures ($>30^{o}$ C) have been observed. Gonzalez and Yevich (1976) observed mortality of blue mussels in the effluent canal of Brayton Point Plant when temperature exceeded 27° C, while Carr and Giesel (1975) noted a decrease in biomass and abundance of several fish species of commercial importance during the summer when mixed temperatures in the receiving water exceeded 30° C. In some such instances, however, recovery during winter months was observed (Roessler et al., 1974).

In the majority of water bodies, these critical conditions (1, 2, and 3, page 5) do not occur and studies report no first order impacts resulting from thermal loading. With rapid mixing and in large water bodies, effluent volumes can preclude problems even in the discharge zone. Gore et al. (1979) reviewed monitoring studies from 25 nuclear plants and found no statistically significant impacts attributable to increased temperatures. Many workers have noted that temperature increases enhanced immediate rates of primary production and could thereby be a positive attribute in terms of increasing fish production in the water body (Cole, 1973; Gammon, 1976a,b; Moore, 1978; Carolina Power and Light Company, 1979; Illinois Institute Natural Resources, 1979). Recreational fisheries in many reservoirs have been enhanced by this overall increase in production (Battelle, Pacific Northwest Laboratories, 1979). A similar lack of adverse thermal impacts was noted in reviews done for the Utility Water Act Group (1978) and for this study (Chapter 2). Minimal or no adverse impacts were found due to thermal loading in the majority of coastal (Atlantic and Pacific) and riverine environments and in the Great Lakes (Utility Water Act Group et al., 1978). In most of these environments the volume of heated water is very small relative to the volume of water in the receiving body, and dilution alone probably mitigates any temperature problems that might affect individuals or populations of individuals living around the power plant.

Current understanding of the long-term and system-wide effects of thermal loading is more limited (Christensen et al., 1975; Coutant and Talmadge, 1977; Lawler, Matusky and Skelly Engineers, 1979; Sigma Research Inc., 1979a). Shallow and/or enclosed water bodies in subtropical regions are thought to be more susceptible to long-range, adverse effects than are deeper, more open or temperate water bodies

(Hall et al., 1978). This is due partly to the fact that many of the organisms found in subtropical systems may be functioning relatively near their limit of thermal tolerance (Roessler et al., 1974; Carr & Giesel, 1975) and partly to the fact that the macrophytes on which many of these systems depend seem more susceptible to thermal effluents than do the mixed phytoplankton assemblages in deep water environments (Bott et al., 1973; Kemp, 1981). Temperate water bodies can also be vulnerable if the dominant fishery is cold-water-adapted. Many salmonid populations function at temperatures close to their limits of tolerance in undisturbed water bodies (Bush et al., 1974).

Changes in behavioral patterns and physiological functions that show temperature dependence represent a more subtle but potentially more destructive, long-term impact of thermal loading. These include changes in migration patterns, overwintering habitats, spawning behavior, egg development, growth, emergence and so forth. The early emergence of aquatic insects has been observed in rivers that receive warmed effluents (Coutant, 1968; Mattice and Dye, 1978). Parkin and Stahl (1981) observed that certain species of Chironomidae, that live in effluent-warmed waters, have increased numbers of generations within a single annual cycle. Mathur et al. (1980) predicted increased zooplankton production in a heated reservoir on the Susquehanna River. However, some have suggested that these changes in the invertebrate fauna may be beneficial to the overall health of the aquatic system due to the maintenance of an ample reservoir of fish food (Illinois Institute Natural Resources, 1979).

Changes in fish behavior in response to artificially warmed waters also are an area of concern relative to long-term ecosystem stability (Langford, 1972). Such changes have been observed infrequently (Northeast Utilities Service Company, 1981; El-Shamy et al., 1981; Texas Instruments Inc., 1981). Upstream migrations of commercially important fish such as striped bass, shad and herring are dependent on temperature cues. Anadromous fish could fail to reach their spawning grounds if a thermal barrier were encountered on the upstream journey (Hall et al., 1978). This problem was observed in the Wabash River in Ohio

(Teppen and Gammon, 1976), but was predicted and did not occur in the Connecticut River (Merriman and Thorpe, 1976).

Similarly, heated effluents could induce premature spawning in resident fish populations, or result in reduced reproductive success. However, such problems rarely have been observed. Mathur and McCreight (1980) found no changes in spawning behavior and reproductive success in crappie populations in the Susquehanna River in the vicinity of Peach Bottom Atomic Power Station. Loi and Wilson (1979) were unable to identify any sublethal effects on spawning behavior of fish in the Chesapeake Bay that could be attributed to the operation of Calvert Cliffs Nuclear Station.

Changes in winter habitat preferences for different fish species that congregate near thermal outfalls have been reported widely in the literature (Oak Ridge National Laboratory, 1981). Such changes may represent a preference response of organisms living below their thermal optimum than an indication of adverse impact (Applied Biology Inc., 1977). In some instances this response can be very positive. Graham (1979) reported that the overwintering of manatee populations in water warmed by Florida power plants provided a replacement of the environment lost due to development of many of the hot springs in South Florida that used to provide winter refuge for manatees. Nevertheless, problems can arise. Many species acclimate more readily to declining temperatures than to rising temperatures. As a result, interruptions in plant discharges in winter have caused fish kills (Welch, 1980).

In summary, adverse thermal impacts appear to have been relatively limited either spatially or seasonally. The wholesale destruction of aquatic ecosystems by thermal pollution that was predicted 15 years ago has not materialized. Much of the damage initially anticipated did not occur partly because the mechanisms that protect organisms in natural environments, such as avoidance behavior, acclimation, and relatively wide tolerance ranges (Parker and Krenkel, 1969; Coutant and Brook, 1970; Bush et al., 1974; Coutant and Talmadge, 1977; Hall et al., 1978) were underestimated. In addition, potential problems have been avoided by careful siting of plants outside nursery and spawning grounds or in otherwise sensitive areas (Applied Biology Inc. et al., 1980; Martin Marietta Corp., 1980; Ontario Hydro, 1978) or by engineering designs that mitigate potential long-term impacts (Eiler and Delfino, 1974; Ontario Hydro, 1981). Some expected problems--for example, increases in parasitism and disease, and impacts associated with changing gas solubilities--have been reported only very infrequently and may not warrant a great deal of concern (Hall et al., 1978).

The problems that do exist are frequently confined to the immediate effluent zone. Where behavioral changes have been observed, system-wide implications have not always been assessed but have sometimes been interpreted as relatively insignificant in terms of population-level response. Nevertheless, in most instances, thermal effluents have not turned their receiving bodies into warmer waters that cannot support a viable, indigenous flora and fauna. As a result, many future studies of impact may focus on other types of disturbance as discussed below and consider thermal effects only briefly (New York State Electric and Gas Corp., 1981). Regulatory requirements under the Clean Water Act (Section 316a) will still necessitate some demonstration of minimal thermal impacts (see Chapter 2).

IMPACTS DUE TO THE ENTRAINMENT AND IMPINGEMENT OF ORGANISMS

Impacts from power generation due to impingement and entrainment are the major foci of current concern. Entrainment refers to the uptake of small or microscopic organisms with a volume of cooling water into the condenser system. Impingement refers to the impaction of larger organisms on the trash racks and screens at the cooling water intakes. Both processes can be lethal to the affected organisms. Entrained organisms experience rapid (usually $<$ 10 minutes) changes in temperature and pressure and high mechanical stress as they move through the condenser tubes and are returned to the receiving water. Changes in water chemistry (e.g., biocide application) may also occur during entrainment. Impinged organisms can die as a result of impaction, mechanical abrasion and loss of scales followed by increased susceptibility to disease, or internal hemorrhaging if impact velocity is large. Weakened fish often are re-impinged and killed even if they survive the initial impingement, or are left more susceptible to predation, as was shown by Coutant and Brook (1970).

The immediate mortality of organisms due to impingement and entrainment has been quantified reasonably well in many site-specific and laboratory studies (Chow et al., 1981; Hanson et al., 1977). Typically, entrained organisms are either phytoplankton, zooplankton, ichthyoplankton or epibenthic organisms. Fish greater than 3 cm in length are usually able to avoid entrainment. The majority of impinged organisms are juvenile (< 2 year) fish. Mortality due to entrainment ranges between 50 and 100 percent for phytoplankton, but is much less, 5 to 40 percent, for other organisms. Lethal effects on phytoplankton and, to a lesser extent, zooplankton, are usually discounted since the dead biomass is still available to consumers in the system (Briand, 1975), and reproductive rates for plankton are very high. Mortality of entrained shellfish and finfish larvae is more problematic since the loss of these organisms with long generation times from the ecosystem is more permanent. Similarly, young fish killed by impaction are no longer available to the ecosystem in which they lived because they are collected and usually discarded in land-fills. Thus, current impact assessments concentrate on assessing the total amount of mortality and its overall effect on the aquatic system rather than on measurements of specific mortality rates in different populations.

A great deal of work has focused on the development of mathematical models that simulate the various life stages, life history, and ecological behavior of affected shellfish and/or finfish populations (Lawler, Matusky and Skelly Engineers, 1975; Goodyear, 1977; Barnthouse et al., 1980a; Martin Marietta Corp., 1980; Tetra Tech Inc., 1980). The attributes of these models are reviewed in Chapter 3. Other studies have provided measurements of site-specific rates of impingement and entrainment during different flow regimes and have estimated mass effects on an annual basis (Weston Environmental Consultants and Designers, 1977; Hazleton Environmental Sciences, 1979;

Jude et al., 1980). Different methods have predicted widely varying effects in various water bodies over short- and long-time scales. Detailed modeling of striped bass and white perch populations in the Hudson River (McFadden et al., 1978; Barnthouse et al., 1980b) predicted a small (< 10 percent) decline in these populations, while similar models for striped bass in the Sacramento-San Joaquin Estuary (Ecological Analysts, Inc., 1981) predicted no long-term impacts. After almost 20 years of operation at Contra Costa, no decline in striped bass populations has been measured in the existing monitoring data. Extensive modeling and field studies of impingement and entrainment effects on fish populations near many of the Tennessee Valley Authority plants (Alexander, 1981; Dycus et al., 1981; McDonough, 1981), at Brunswick Steam Electric Plant, and at other plants or in the laboratory (Kedl and Coutant, 1975; Chadwick et al., 1977; Lawler, Matusky and Skelly Engineers, 1980c; Polgar et al., 1981) have predicted and/or determined only minimal system-wide damage. These observations may be partly due to compensatory responses in affected fish populations or partly due to the fact that only small percentages of populations are damaged (Jensen, 1981). Other estimates, however, suggest that in areas where power plants are concentrated (e.g., Great Lakes, Chesapeake Bay), adverse effects may be occurring since significant proportions of the total ichthyoplankton population pass through condenser tubes (Leslie et al., 1979).

The overall ability to quantify impacts associated with impingement- and entrainment-induced mortality is limited primarily by poor understanding of the effects of larval and young-of-the-year (YOY) mortality on population development (Voigtlander, 1981; Chow et al., 1981). The extent to which density-dependent mechanisms (e.g., increased availability of habitat, food, etc.) can compensate for mortality of juvenile fish is poorly understood by fisheries scientists (cf. Van Winkle, 1977b; Jensen, 1981). Other problem areas also exist. Accurate measurements of population size are difficult to obtain in many water bodies (Sigma Research, Inc., 1979b). System-wide effects of impingement and entrainment are clearly more economically critical if commercially important fish have migratory paths or spawning and rearing habitat in the vicinity of plant intakes and discharges. Migratory and/or schooling fish tend to be more affected than dispersed populations. The impacts on exploited populations are usually greater than on unexploited fish stocks (Goodyear, 1977) since populations may already be in the early stages of depletion and adequate reserves do not exist.

Many problems can be avoided through careful siting or engineering designs. Traveling screens used around intake points can decrease impingement mortality since fish are less likely to become impinged on moving screens (Tatham et al., 1978). The use of physical barriers, screen washings, offshore intakes and return systems also can help to reduce adverse effects of impingement. A great deal of entrainment can be avoided by siting intakes outside rearing grounds or by decreasing the mesh size of intake screens (Academy of Natural Science, Philadelphia, 1981; Schneeberger and Jude, 1981). Nevertheless, losses due to plant operation at many existing plants and at proposed plants with no protected sites available do and will exist (Texas Instruments, Inc., 1981). The extent to which measured and predicted mortality rates result in long-term depletion of fish and shellfish stocks is not known and should be a focus for future research.

IMPACTS DUE TO CHEMICAL ALTERATION OF THE AQUATIC ENVIRONMENT

Impacts due to chemical changes are associated primarily with the use of biocides and anticorrosives for cleaning and maintaining condenser tubes, disposal of stack scrubbing water and leachate from on-site waste treatment systems. The effects of biofouling substances have been evaluated extensively in both field and laboratory studies (Brungs, 1973; Stober and Hanson, 1974; Gentile et al., 1976; Mattice and Zittel, 1976; Oak Ridge National Laboratory, 1980). Evaluations of impacts associated with leachate, particularly from flyash and heavy ash ponds, and to a lesser extent, from coal piles and coal washings, are less abundant but will become more important as coal utilization increases at many plants in the United States (Skinner et al., 1980; Clark et al., 1981; Larrick et al., 1981). Many of the initial investigations of biocides, leachate and blowdown effects (Dickson et al., 1974) suggest that impacts are short-lived and very localized. Cairns et al. (1971) showed that recovery following spill contamination is possible. Few studies have evaluated other system-wide impacts.

Laboratory and, to a lesser extent, field data for evaluating the effects of chlorine and other biofouling substances have delineated several consistent patterns of biotic impact. These patterns show that toxic effects are:

1. Species specific with some organisms being very resistant while others are sensitive (Mattice and Zittel, 1976; Mattice, 1977; Marine Research, Inc., 1979);

2. Life-stage specific, with more sensitivity observed in very young larvae than in eggs or adults (Larson et al., 1977; Mattice, 1977; Stewart et al., 1979);

3. Site-specific, with temperature preferences of species affecting sensitivity (Brungs, 1973; Mattice, 1977); and freshwater species having different (often greater for chronic exposure) sensitivity than marine organisms (Mattice and Zittel, 1976);

4. Synergistic between a chemical and increased temperature or among several different chemicals (Capuzzo et al., 1976; Brooks and Seegert, 1977); and

5. Dependent on length and/or amount of exposure so that chronic, low-level exposures and acute, intermittent exposures are harmful at different concentrations (Gentile et al., 1976; Mattice and Zittel, 1976; Mattice, 1977; Hall et al., 1981).

In many field situations, direct mortality or sublethal effects of biocides affect a relatively small percentage of organisms in the receiving water body. However, larger proportions of a given population can be impacted if it is sessile and located near the plant outfall. The most critical problems exist in water bodies where shellfish are a major component of the ecosystem. These organisms tend to bioaccumulate metallic substances (e.g., copper) that leach from the condenser tubes. Some species of warm water fish have been shown to be affected chronically by copper (Lawler, Matusky and Skelly Engineers, 1980c). Additional concern has focused on the entry of long-lived halogenated compounds into aquatic environments (especially marine) and their subsequent bioaccumulation (Hall et al., 1978). However, the presence of chlorinated organic compounds in power plant effluents has not been proven and may be unlikely since appropriate catalysts rarely co-exist. Bioaccumulation of these, or metallic compounds may adversely affect long-term viability of important finfish and shellfish populations and limit their commercial value.

Similar concerns exist for investigations of leachate from waste disposal ponds. Coal piles, flyash and heavy ash are very rich in several heavy metals (e.g., arsenic and selenium), and these elements can be bioaccumulated (Jester et al., 1980). Bioaccumulation of leachate toxicants appears to be more significant in filter-feeding organisms and less of a problem in bottom-feeding or predatory organisms. Thus, in areas where filter-feeding organisms (oysters, mussels, etc.) are a major part of the food chain (e.g., estuaries), leachate toxicants from coal wastes may present a significant problem in the future. In many instances, however, the simple dilution of leachate waste may result in amelioration of all but the most localized impacts (Larrick et al., 1981). Bioaccumulation of leached toxicants by macrophytes, which can then be removed from the natural environment, may represent an inexpensive and effective treatment technique for some disposal problems (Clark et al., 1981).

A conclusive statement relative to the overall impacts of the chemical alterations associated with power generation is premature. Laboratory data on acute and chronic toxicity must be evaluated in more field situations. Effects of bioaccumulation and chronic toxicity from waste disposal leachate must be quantified. The severity of these effects in terms of limiting populations beyond the immediate plant outfall zone should be measured and/or assessed through the use of transport and accumulation models. The current data base shows that local and short-term toxicity problems exist for specific populations. Although some data indicate that long-term, system-wide effects exist, future efforts should concentrate on quantifying these problems. The potential for mitigation through engineering design also should be assessed.

DISCUSSION

The above review and Figure 1.1 summarize much of the current understanding of the effects or potential effects of power generation on aquatic systems. Several important conclusions emerge. Biological effects tend to be very species- and site- specific. The potential for mitigation, either through avoidance (e.g., siting) or careful design features (e.g., alteration of structures, or timing of withdrawal or disposal), encourages optimism. In some of the more exposed environments, long-term plant operations have had only very localized impacts (Hazleton Environmental Sciences, 1979; Ecological Analysts, Inc., 1981).

Effects due to thermal loading have generally been shown to be problematic only in shallow, enclosed and/or subtropical environments or in water bodies where cold water species (salmonids) dominate. The impingement and entrainment of phytoplankton almost never appear to cause a significant impact on the whole system and that of zooplankton appears rarely significant. The population and ecosystem level effects of larval (finfish and shellfish) entrainment and impingement and of chronic toxicity (particularly to shellfish) due to waste leachates are possible but have not been well documented and are poorly understood.

These problems may be considered by some to be significant only if important commercial or sports fisheries exist in the vicinity of the plant. Others may contend that any changes in species composition or ecosystem structure are unacceptable. In some instances effects noticeable at the population level may not be detrimental to trophic level transfers of energy and can even be favorable if growth rates and/or production rates increase. Some impacts may be so localized or occur so infrequently that continued monitoring may not be cost-effective or necessary (see Chapter 2). In some areas (e.g., the Great

Lakes), concentrations of power plants (energy parks) may mean that all effects are magnified and these cumulative impacts should be investigated.

Evaluations have been limited primarily by a lack of adequate (to site-specific constraints) methodologies, inappropriate statistical design, and large spatial and temporal variations in natural populations. A lack of definitive measures of impact at the community and ecosystem level also has presented analytical problems (Lawler et al., 1980b). Evaluation of the effects of impingement and entrainment is impossible without a good comprehension of compensatory mechanisms and good measures of stock size. Simulation models should be evaluated in further detail, and existing long-term data bases should be reanalyzed relative to whole system impacts. Multiple plant effects should be evaluated further (Tetra Tech., 1980). Differences between laboratory predictions (especially for toxicity data) and field observation should be quantified and evaluated in conjunction with predictive models. In this way, scientific understanding of aquatic impacts can be improved, and appropriate monitoring and mitigation techniques can be developed.

SUMMARY

The following discussion summarizes Chapter 1.

- The primary effects of power generation on aquatic environments are due to the discharge of heat and chemicals; leachate of toxic materials from on-site disposal of coal washing residues, flyash, heavy ash, cooling tower or stack scrubbing wastes; and impingement and entrainment of organisms during intake of cooling water.

- Biological effects of thermal loading are often very localized and short-term if plants are located where adequate dilution can occur.

- Behavioral and physiological effects of thermal loading have rarely been shown to be detrimental on a system-wide basis and, in some cases, have been beneficial in terms of increasing fish production without advancing eutrophication.

- Population level effects of thermal discharges appear to be restricted largely to shallow, enclosed water bodies and to subtropical or temperate water bodies with organisms operating near their thermal tolerance limits in a natural state.

- Population level effects of impingement and entrainment are species-specific and have been confined primarily to shellfish and finfish populations.

- Long-term depletion of fish stocks due to increased larval cropping by impingement or entrainment is poorly quantified and may represent a significant problem, particularly in areas having important populations of migratory fish and/or areas having several power plants on one water body.

- Most chemical effects of power plants (e.g., biocide discharge, leaching of wastes) appear to have been short-term and localized.

- Problems due to chemical toxicants are greatest in areas where sessile benthic organisms (particularly filter feeders) exist near the outfalls. Long-term effects on such populations, primarily as a result of bioaccumulation, may present a significant problem at coal-fired plants.

- Careful siting of plants, intake structures, and outfalls away from sensitive areas and outside of nursery and spawning grounds, can mitigate many of these problems.

- Design alternatives (e.g., diffusers, barrier, etc.) can lessen adverse impacts at existing plants.

CHAPTER 2
GOALS AND CRITERIA FOR MONITORING DESIGN

INTRODUCTION

The design of monitoring programs to assess impacts associated with power generation has been guided by (1) compliance guidelines that necessitate measurement of a wide variety of variables, and (2) preliminary identification and subsequent measurement of organisms in the aquatic community that appear to be susceptible to the operation of a particular plant. As the regulatory climate changes, the former, extensive approach will be increasingly replaced by the latter, more cost-effective approach. Often such evaluations will consist of specific hypothesis development and statistical tests that include identification of sampling stations, frequency of sampling, and determination of the exact variables to be sampled.

In this chapter, three major factors that affect sampling design are reviewed. These include (1) the legal or regulatory basis for monitoring of the aquatic impacts associated with power plant operation; (2) an analysis of the variables included in past monitoring programs that have provided useful information for assessment purposes, especially including suggested hypotheses of potential impacts that could be tested in future monitoring programs; and (3) the statistical basis for hypothesis testing and monitoring design. Methodological questions, related both to sampling technology and statistical assessment, are deferred to Chapter 3.

MONITORING REQUIREMENTS UNDER LEGISLATIVE AND REGULATORY AUTHORITY

Authority to require aquatic monitoring at electric power plants in the United States is derived primarily from the National Environmental Policy Act of 1969 (NEPA) and the Clean Water Act of 1977 (CWA). Most state involvement results from the specific designation of enforcement authority by federal agencies or passage of state-level versions of these federal laws.

On the basis of these federal and state legislative actions, various regulations have been promulgated that mandate and govern aquatic monitoring. Because the provisions of monitoring programs must be site-specific, these regulations are usually rather general in nature, especially those applying to biological monitoring. This generality leaves much to the interpretation of applicants, regulatory agencies, and interveners. In many past cases, the resulting monitoring programs have required extensive measurements of many biotic variables that may or may not have been useful in defining aquatic impacts. As noted above, there is evidence that this tendency is changing towards more scientifically directed, cost-effective approaches. As the understanding of aquatic monitoring improves, applicants are gaining approval to modify existing programs and institute new efforts having a focus that meets the intent of the underlying legislation. Monitoring design is likely to be further affected by presidential Executive Order No. 12291, issued February 17, 1981, which declared that regulatory benefits must exceed costs of compliance and set forth general provisions to achieve that objective.

Federal Legislation

Table 2.1 summarizes the federal statutes and regulations that govern aquatic monitoring at electric power plants. The most important of these are the National Environmental Policy Act of 1969, the Clean Water Act of 1977, and other federal legislation.

Table 2.1. Federal Laws and Regulations Concerned with Aquatic Monitoring at Electric Power Plants.

Legislation	Bearing on Monitoring	Regulation References (1)	Technical Guide Document References
National Environmental Policy Act of 1969	Provides authority to require monitoring for impact statements	E.O. No. 11514 43 FR 230 (general requirements)	
		10 CFR 51 (NRC regulations concerning NEPA requirements for nuclear plants) (2)	NRC 1975b, 1975c, 1976
Clean Water Act of 1977 Section 208	Established area-wide waste management planning for nonpoint sources which may involve power plant site storm runoff		
Section 301	Defined effluent limitations	40 CFR 423 (electric power plants generally) 45 FR 37432 (coal pile runoff)	
Section 303	Established receiving water quality standards and provides part of the basis for requiring receiving water monitoring under NPDES		
Section 304(a)	Defined conventional pollutants		
Section 306	Defined new source performance standards		
Section 307	Defined toxic pollutants		

Cont.

Table 2.1. Cont.

Legislation	Bearing on Monitoring	Regulation References (1)	Technical Guide Document References
Section 308	Provides part of the basis for requiring receiving water monitoring under NPDES		
Section 316(a)	Authorizes variance from requirement of best available technology for thermal effluents if it can be demonstrated through monitoring that the less stringent requirement will protect a balanced, indigenous aquatic community	EPA (1977a)	
Section 316(b)	Requires monitoring to demonstrate that best available technology which minimizes impingement and entrainment impacts has been selected for cooling water intakes		
Section 402	Established NPDES and associated permit program and provides general authority for monitoring to aid in setting permit conditions and demonstrating compliance	40 CFR 121 - 125	
Section 403	Established authority to regulate ocean discharges and require monitoring to demonstrate lack of ecological degradation	45 FR 65953	
Section 511(c)	Reserves final authority over water quality and consequent monitoring requirements to EPA. Requires EIS for any new source discharge permit issued by the federal government.		

Table 2.1. Cont.

Legislation	Bearing on Monitoring	Regulation References (1)	Technical Guide Document References
Coastal Zone Management Act	Governs state coastal land use programs, which may have monitoring requirements		
Endangered Species Act	Requires federal agencies to consult with the Fish and Wildlife Service concerning their proposed actions (FWS opinion could require monitoring to insure protection of endangered species)		

Notes: (1) Abbreviations: E.O. -- Executive Order; FR -- Federal Register; CFR -- Code of Federal Regulations.

(2) Superceded by Yellow Creek decision of 1978 (Docket Nos. 50-566 and 50-567), which ruled EPA has final authority over water quality and aquatic monitoring.

The National Environmental Policy Act of 1969

The National Environmental Policy Act of 1969, in the words of Sanders et al. (1979), "...provides the unifying and guiding philosophy for ecological effects assessment." The primary objective set by Congress for NEPA was to establish a national policy directed at environmental protection and enhancement. NEPA's specific provisions are few, most prominent among them being the requirement for an environmental impact statement (EIS) presenting an assessment of the anticipated consequences of each significant federal action. The act also created the Council on Environmental Quality (CEQ) to oversee the act's implementation. NEPA does not explicitly deal with monitoring questions. However, Sanders et al. (1979) pointed out that Section 102(c) of the act, which established the EIS structure, implies the need for long-term monitoring to predict impacts and judge the quality of predictions. In fact, among the federal environmental laws, only NEPA provides the basis for continuity from baseline, through pre-operational, to compliance monitoring during operations.

Executive branch actions stemming from NEPA have reinforced the implied need for monitoring with more concrete stipulations. Executive Order No. 11514 (1970) stated that federal agencies should monitor their activities in the interest of environmental quality, and directed CEQ to promote monitoring needed for impact prediction and assessment. In 1978 CEQ promulgated final regulations (43 FR 230) that provide general guidelines for EIS preparation and require monitoring to insure that agency decisions are implemented.

In the case of privately initiated actions that require federal approvals, CEQ designates a lead agency for overseeing compliance with NEPA. For nuclear power plants, the U.S. Nuclear Regulatory Commission (NRC) is the designated lead agency. Steam electric plants utilizing conventional fuel sources and built with private financing are not tied so directly to NEPA authority. In 1976, NRC promulgated regulations (10 CFR 51) for the general content of an applicant's environmental report, which then becomes the basis for the Commission's EIS. Because these procedural guidelines are inadequate to define substantive ecological matters, several regulatory guides (U.S. Nuclear Regulatory Commission, 1975a; 1975b; 1976a) have been issued under NEPA authority to provide more specifics. All of these guides have relevance to monitoring.

Regulatory Guide 4.7, Rev. 1 (U.S. Nuclear Regulatory Commission, 1975b) discusses factors to be considered when conducting a site selection study. Regulatory Guide 4.2, Rev. 2 (U.S. Nuclear Regulatory Commission, 1976b) specifies environmental report preparation. This guide requires an estimation of impacts on important species and the description of monitoring efforts during pre-operational and operational phases. It provides only general guidance on the scope of these programs, however.

Regulatory Guide 4.8 (U.S. Nuclear Regulatory Commission, 1976a) outlines the operational monitoring program necessary to assess the impact of plant operations on the receiving water. These Environmental Technical Specifications guide utilities and their consultants in designing monitoring programs for presentation to NRC for approval or negotiated modification. While the Technical Specifications may vary to respond to site-specific differences, their hallmark has been coverage of all trophic levels in a rather unfocused manner. Another major concern of the Technical Specifications has been sampling of impinged and entrained organisms. Review of a number of plant technical specifications indicates that particular provisions for guiding sampling station selection, replication, or data analysis were not generally included (Adams et al., undated a, b, c; U.S. Nuclear Regulatory Commission, 1975a). As a consequence of poorly conceived monitoring design in some cases, the resulting data have not always been amenable for use in the conclusive assessment of impacts. Figure 2.1 illustrates time lines for the various survey, design, and construction phases at a typical nuclear power plant regulated under Environmental Technical Specifications in 1977.

A major change in the regulation of monitoring at nuclear power plants occurred as a result of the "Yellow Creek" decision in November

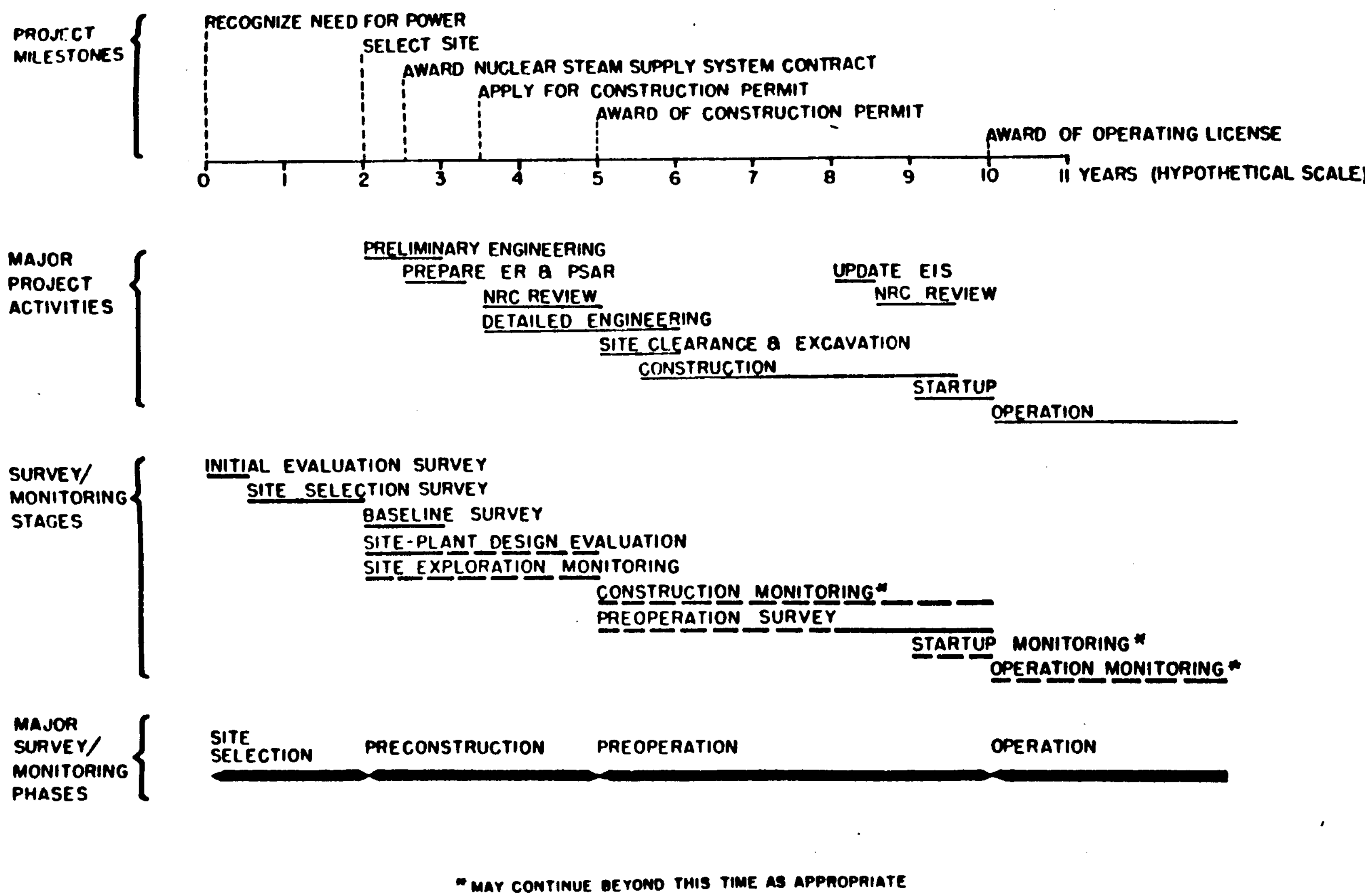

Figure 2.1. Typical development schedule for nuclear power plants in the United States in 1977, with major survey phases and their corresponding survey stages indicated. (Solid lines indicate likelihood of continuous or continuing periodic studies; dashed lines indicate that any studies, or continuing periodic studies, are likely based on potential sources of impact or regulatory requirements.) (From American National Standard Institute, Inc., 1977).

1978. In issuing the construction permit for the Tennessee Valley Authority's (TVA) Yellow Creek plant, NRC had required aquatic monitoring in excess of that designated by the National Pollutant Discharge Elimination System (NPDES) permit granted under the U.S. Environmental Protection Agency's (EPA) CWA authority. TVA sought relief from the extra requirements on the grounds that CWA Section 511(c)(2) prohibits federal agencies from invoking NEPA as authority for setting effluent limitations or requirements differing from those of EPA as licensing conditions.

The case (Docket Nos. STN 50-566 and STN 50-567) was decided in TVA's favor and upheld on appeal. The Atomic Licensing Appeal Board ruling, in part, stated:

> The Nuclear Regulatory Commission may not incorporate in licenses to build nuclear power plants conditions that call for review of the adequacy of water quality requirements previously established by the Environmental Protection Agency. Section 511(c)(2) of the Federal Water Pollution Control Act curtailed the spread of federal responsibility for water quality standards and pollution control among the various licensing agencies. That responsibility was shifted to EPA as its exclusive province. ...Each agency must continue to weigh any resulting degradation of water quality in its cost-benefit balancing pursuant to the National Environmental Policy Act. However, licensing agencies are now limited to the role of balancing, with substantive regulation of water pollution to be left to EPA.

Subsequent to the Yellow Creek decision, NRC has approached monitoring requirements in existing construction permits and operating licenses in a different manner. Upon petition from construction permit and operating license holders, the Commission has removed previously designated NRC monitoring requirements if EPA agreed. The programs normally have continued in some form under NPDES authority. It is unlikely that NRC would attach monitoring requirements to any new operating license applications.

To date, these practices represent policy changes only and have not been reflected in new regulations. A dialog concerning aquatic monitoring continues among applicants, EPA, and NRC; and NRC Regulatory Guides 4.2, 4.7, and 4.8 remain in use.

Clean Water Act of 1977

The Clean Water Act of 1977 (CWA) is the amended version of the Federal Water Pollution Control Act of 1972, the major objective of which is "...to restore and maintain the chemical, physical, and biological integrity of the Nation's waters." The main emphasis of the Act is the regulation of pollutant discharges; thus, monitoring under the CWA is primarily directed at effluent surveillance rather than receiving water conditions.

Section 402 of the CWA established the NPDES as the permit program to review and approve all pollutant discharges under designated limits. The EPA authorizes states to manage the permit process, if they request the authority and demonstrate capability. Issuance of a discharge permit for existing facilities depends on the applicant's ability to show evidence of being able to meet effluent limitations, defined under Section 301. New facilities are required to achieve a higher level of control than existing plants. CWA Section 306 established new source performance standards applying to such cases.

Pre-application monitoring under NPDES usually consists of measuring conventional pollutants, defined under Section 304(a), as well as others designated for the industry in question by the effluent limitation guidelines, in the receiving water near the site of the proposed discharge. Monitoring at this stage also routinely involves biological sampling to characterize species abundance and diversity in the nearfield. These data, along with the waste characteristics, are used by the state permitting agency to set effluent limits.

After permit issuance, compliance monitoring normally involves measuring the quantity and quality of the effluent before discharge. Required chemical and biological monitoring of the receiving water is rather uncommon, in general, although Sanders et al. (1979) reported that regional EPA officials are justifying it in some cases under the authority of Section 308.

Under CWA regulations, thermal effluents are subject to control by best available technology, meaning some form of closed-cycle cooling, unless a variance from that requirement can be justified under Section 316(a). Less stringent effluent limitations will be applied where an applicant can demonstrate that a balanced, indigenous population of fish, shellfish, and wildlife will be protected under the modified limits. This demonstration requires extensive pre-operational and operational monitoring.

U.S. EPA (1977a) has published draft technical guidelines for 316(a) demonstrations, which define several categories of studies. Type I is a non-predictive demonstration of lack of appreciable harm due to prior use of once-through cooling, whereas Types II and III are predictive of future effects. Type II involves monitoring representative important species (RIS), generally throughout the ecosystem, when the potential for significant impacts is high. Less involved Type III monitoring may be undertaken, with the EPA regional administrator's concurrence, when relatively low potential for impact exists.

The 316 (a) technical guidance manual sets forth decision criteria to determine the type of demonstration required and general study requirements for phytoplankton, zooplankton, ichthyoplankton, macroinvertebrates, habitat-formers (macrophytes, macroalgae, corals, sponges), shellfish, finfish, and other vertebrates. Section 316(a) variances are issued for ten years; thus, repeated monitoring could be required on that frequency.

Section 316(b) of the CWA requires that the best available technology (BAT) to minimize entrainment and impingement impacts be chosen for cooling water intakes. Studies must be performed to demonstrate that BAT has, in fact, been selected. U.S. EPA (1977b) also has issued a draft technical guidance manual for 316(b) demonstrations. It specifies the necessary site description, hydrodynamic data needs and biological survey requirements. Among the latter are general guidelines for sampling the water source for phytoplankton, zooplankton, macroinvertebrates, ichthyoplankton, and fish. The guide also covers sampling of impinged and entrained organisms. It provides very brief coverage of conventional statistics employed in data analysis, physical and biological models, and measures of ecological diversity.

Ocean discharges are subject to special regulation under Section 403 of the CWA. Under regulations published October 3, 1980 (45 FR 65953), the EPA regional administrator is directed to determine whether a proposed discharge will cause unreasonable degradation. Currently, the major applications of section 403 are at offshore sites. EPA will extend application of this section to shoreline facilities, including power plants, in the near future. Discharges in compliance with Sections 301(g) and 316(a) are assumed to be in compliance with Section 403, although, as yet no regulations for 301(g) have been issued. When there is insufficient information for the administrator to make a decision, discharge is permitted only in the absence of an alternative and must be monitored. Permission to discharge is revoked if monitoring shows it to cause degradation.

Other than drainage from coal piles, which is regulated under NPDES permits, stormwater runoff from power plant sites during construction or operation is not specifically controlled by the CWA. As a nonpoint pollution source, the site could be covered under Section 208 if it is in an area-wide waste management planning area.

Other Federal Legislation

Other federal laws that may have a bearing on power plant siting and monitoring programs are the Coastal Zone Management Act and the Endangered Species Act. The Coastal Zone Management Act governs state coastal land use programs and establishes estuarine sanctuaries. The Endangered Species Act, passed in 1973 and ammended in 1978, obligates federal agencies to consult with the Fish and Wildlife Service (FWS) to insure that their actions do not advance the decline of endangered species. FWS issues an opinion when consulted, which could include requirements for a monitoring program.

State Authority

States can become involved in the regulation of aquatic monitoring at electric power plants through transfer of authority from federal agencies, as well as state-level environmental policy acts or energy facility siting commissions established in some states.

The leading and most formalized example of the transfer of federal authority is state administration of NPDES permit programs, which is exercised by the majority of states. As an outgrowth of the Yellow Creek decision, NRC has relinquished authority over monitoring, in connection with both construction permits and operating licenses, on a case-by-case basis. In some cases, state agencies have assumed the oversight responsibility (e.g., Consumer Power's Midland, Michigan plant, and, as of 1981, Consolidated Edison's Indian Point, New York plant).

A number of states, including Maryland, California, Washington and Oregon, have passed state-level environmental policy acts. These statutes are analogous to NEPA and thus have established requirements for environmental impact assessments and monitoring for projects initiated by state government or which require state agency permits.

Several states, again including Maryland, California, Washington and Oregon, have set up some form of centralized siting commission for power plants or for energy facilities in general. These commissions are active to varying degrees in actual siting and the attendant monitoring. The most common model is one in which the commission reacts to applicants' siting decisions and serves as a "one-stop" permitting agency.

The prime example of a siting commission active in all phases of the decision-making process is the Maryland Power Plant Siting Program.

A cooperative effort of the Departments of Natural Resources and Health and Mental Health and the Public Service Commission, the Program establishes 10-year plans recognizing state power needs. The staff then proceeds to study potential power plant sites and prepare environmental impact statements for selected sites, which may be purchased by the state and made available to utilities. Among the Program staff's support functions is the coordination of monitoring of both general baseline conditions, especially in Chesapeake Bay, and of individual sites. State agencies thus develop data to evaluate environmental impacts with a single, uninterrupted program proceeding from baseline through the operational phase. Funding for the program is provided by a special tax on utility bills.

In summary, while some differences exist in the amount of regulatory authority assumed by states, in their requirement of impact assessment, and in their attention to siting, they do not substantially alter the regulatory pattern established at the federal level. Sanders et al., (1979), following a detailed review of procedures in three states, concluded that, "In general, states add few, if any, additional monitoring requirements to those called for under federal regulations or required by federal agencies.

ASSESSMENT OF PAST MONITORING EFFORTS

In response to regulatory requirements, past aquatic monitoring efforts have tended to collect data on as many species as possible rather than focusing on specific hypotheses and subsequent measurement of variables to test these hypotheses. An assessment of the effectiveness of particular variables (measured parameters) to provide useful information on aquatic impacts is needed. Such an assessment can benefit future monitoring designs by identifying and giving order to hypotheses of anticipated impacts. Each hypothesis should identify measurable parameters (variables) and indicate expected levels of change due to plant operations. Monitoring simply to describe an aquatic ecosystem is usually inadequate to determine environmental impacts (Van Winkle, 1977a). It is preferable, although in some instances idealistic, to decide a priori what degree of change is unacceptable. In theory, the level of sampling necessary to detect that amount of change in field data, or to predict that amount of change via mathematical modeling, then can be incorporated into the sampling scheme (Thomas, 1977). Many past monitoring programs have failed to develop hypotheses of expected impact, let alone consider levels of expected change.

Two general approaches have been used to determine such changes. One approach, known as synoptic surveys, involves intensive monitoring occurring over a short time period during which a wide variety of different parameters are sampled. The purpose of such sampling is to determine as fully as possible the exact state of an ecological community at a given time. This type of sampling is typically conducted at the site of a proposed power plant before the plant is constructed. Synoptic surveys are designed to determine what types of ecological communities currently exist and can, its authors hope, suggest whether there is any reason to suspect that the community would be significantly impacted by the facility under consideration. Such surveys also would be conducted after the facility had been constructed and put into operation to determine whether the ecological community had been impacted.

The second type of approach is known as baseline monitoring. In such an approach, sampling is conducted more frequently than in synoptic surveys (e.g., from weekly to monthly), but the number of different parameters that are sampled is greatly decreased. Rather than attempting an all-inclusive description of the environment (the object of the synoptic surveys), the baseline sampling attempts to measure at prescribed intervals the state of a few parameters that are chosen to represent the entire system. There is no clear dividing point between these two approaches; that is, there are no established criteria that would distinguish one from the other.

Temporal variability, methodological and technical errors, and the predictive capacity of developing information bases rarely have received adequate attention. Incomplete assignment of importance values to the components of complete data sets also has stifled the development of optimal monitoring programs.

Variable Identification

A variable that is appropriate for testing the existence of a hypothesized impact must meet several criteria. These include

1. The variable must be sensitive to the hypothesized impact,
2. The variable must be sufficiently stable (through time and space) to allow for variations due to power generation to be separated from other (including natural) sources of variation, and
3. Reliable sampling and analytical methodologies for detecting variation must exist and must be technically and economically feasible.

These criteria represent standards of minimum acceptance for monitoring variables.

Once a significant or nonacceptable change is detected in a chosen variable, other choice criteria become useful. Such criteria are desirable for increasing the utility of collected information but are not necessary for initial detection of an impact. These second-rank criteria focus on the significance of the detected change and sampling design. They include

1. The detected change in the variable can be classified as reversible or irreversible,
2. The detected change in the variable could have been predicted with confidence before sampling or with less intensive sampling,
3. The detected change in the variable will imply long-term effects on the population or ecosystem or both,
4. The detected change in the variable is very localized and/or represents only a temporary impact.

Thus, given a particular site and a particular generating plant (or set of plants) and the hypothesized impacts, these choice criteria could be used to identify components of a viable monitoring program. Inadequate information relative to each of these choice criteria has obviated their past utility.

A preliminary review of the assessment literature indicates that sufficient information exists relative to certain choice criteria (e.g., variable sensitivity to impact, sampling errors). This information is reviewed below. To date, the practiced art of impact assessment has been inadequate to address problems of variable instability and prediction of system responses to perturbation. As a consequence in most situations, the design of powerful and cost-effective monitoring programs has been impossible. Techniques that address these problems have been applied in other areas of water resource analysis and show promise for the field of impact assessment. Such techniques will be evaluated in Phase II.

Variable Sensitivity to Impact

A review of utility documents (including pre-operational and operational monitoring programs to fulfill requirements of NRC Technical Specifications, CWA sections 316a and 316b and/or other federal and state regulations) revealed 162 studies with sufficient biological data to evaluate the existence (or lack thereof) of an impact due to the operation of an electrical generation facility. Of these studies, 137 were reviewed in a comprehensive analysis prepared by the Utility Water Act Group of the biological effects of once-through cooling (Utility Water Act Group et al., 1978). An additional 25 studies were obtained for this analysis from cooperating utilities. Thirty-two of the studies used in the UWAG documents were re-reviewed to impose a consistent level of quality control on the interpretation of primary data sources. Studies in which data sources were documented poorly or in which methodological errors appeared to be unreasonably large were excluded. Thus, the studies included in this review were judged to have had a sampling program which was sufficient to show an impact if one existed.

The review represents a comprehensive summary of the sensitivity of 19 different variables that have been used frequently in monitoring programs. The majority of these monitoring programs focused on the effects of thermal loading and impingement and entrainment of important populations of organisms. Several evaluated chemical effects due to biocides (usually chlorine) either directly or indirectly. None evaluated impacts due to on-site waste disposal or bioaccumulation of toxic compounds. Eight studies evaluated long-term (life of the plant) impacts explicitly, and only two addressed the question of multi-plant effects. In all instances these evaluations focused on the depletion of commercially important fish stocks (primarily striped bass). Two studies evaluated the significance of trophic interactions between impacted and non-impacted populations.

The information was analyzed in the aggregate to clarify dominant trends and on the basis of water body type to focus attention on variables having potential for future monitoring programs. The site-specific categories and number of studies are shown in Table 2.2. The 19 variable categories and their judgement criteria are shown in Table 2.3.

The aggregated results of this analysis are presented in Table 2.4. An arbitrary value of >25 percent of reviewed cases was chosen to define whether a variable was effective in showing adverse effects versus no or positive effects. This "25 percent rule" was considered conservative as well as useful for the task at hand in view of the lack of any other criteria for judgment. Nine of the test variables consistently showed adverse impacts that could be attributed to plant operation (i.e., in more than 25 percent of the cases). Two of these variables showed that impacts were restricted either to the zone of plume influence or to an area immediately adjacent (phytoplankton production and distribution of sessile benthos). Two variables were useful only in marine and estuarine environments (macrophyte and macroalgal abundance). One of these variables showed long-term impacts while the others were very short-term, often on the order of days.

Table 2.2. Site-Specificity of Assessment Reviews

Water Body	Type of Cooling Systems	No. of Studies
Great Lakes and Connecting Waters	Open	22
Freshwater Lakes and Reservoirs	Open	29
Freshwater Rivers	Open and Closed	54
Tidal Rivers and Estuaries	Open	38
Marine/Coastal Environments	Open	19
TOTAL		162

Table 2.3. Assessment Variables and Level of Consideration

Community of Interest	Temporal Scale Considered	Spatial Extent	Ecological Level	Measured Parameter	Criterion(a) for Impact Evaluation
Phytoplankton	short-term (days)	plume influence	individual species community	numbers, biomass	increase or decrease relative to preop. or control condition
Phytoplankton	short-term (days)	plume influence	population	biomass of entrained cells	increase or decrease following entrainment
Phytoplankton	short-term (days)	plume influence	community	primary production	increase or decrease following entrainment
Periphyton	short-term (seasonal)	plume influence	community	biomass, abundance	increase or decrease relative to preop. or control condition
Macrophytes	short-term (seasonal)	plume influence	community	biomass, abundance, spp. distribution	increase or decrease relative to preop. or control of condition (incl. change in community structure)
Macroalgae	short-term (annual)	plume influence	community	abundance	increase or decrease relative to preop. or control condition
Zooplankton	short-term (seasonal)	plume influence	population	numbers, species	increase or decrease relative to preop. or control condition
Zooplankton	short-term (days)	plume influence	community	survival	percentage survival following entrainment (intake vs. outfall)
Benthos (sessile)	short-term (seasonal)	area near discharge	community	spp. distribution	change relative to preop. or control condition (incl. community structure)
Benthos (motile)	short-term (days)	plume	population	organism movement	avoidance of discharge plume
Meroplankton	short-term (days)	plume influence	community	survival	percentage survival following entrainment (intake vs. outfall)

Table 2.3. Cont.

Epibenthos	short-term (annual)	intake area	population	number impinged	numbers impinged on intake screens relative to total pop.
Adult and Juvenile Fish	short-term (annual)	plume influence (may incl. larger area)	community/ population	abundance distribution	attraction/avoidance of plume incl. seasonal effects
Adult Fish	short-term (days)	plume influence	population	survival	lethal effects of plume observed or not observed
Ichthyo-plankton	short-term (seasonal)	plume influence	population	number entrained	numbers entrained relative to total population (significant)
Ichthyo-plankton	short-term (seasonal)	plume influence	community	survival	percentage survival following entrainment (intake vs. outfall)
Ichthyo-plankton	short-term (annual)	intake area	population	number impinged	numbers impinged on intake screens relative to total pop.
Adult Fish	long-term (plant life)	water body (maybe multi-plant)	ecosystem, population	simulation model (e.g., multiparameter, Ricker)	depletion of long term fishery stocks
Zooplankton & Fish	long-term (> 5 years)	water body	ecosystem	simulation model (e.g. Lotka-Volterra)	change in prey has impact on predator population

Table 2.4. Aggregate Results of Assessment Review

Variable Number	Variable of Interest (measured parameter)	Percentage Showing Adverse Impact	Percentage Showing No or Positive Impact	No. of Studies
1	Plume effect on phyto-plankton abundance, species distribution	18	82	44
2	Entrainment effect on phyto. biomass	24	76	29
3*	Entrainment effect on phyto. production	55	45	55
4	Plume effect on periphyton abundance, species distrib.	22	78	32
5*	Plume effect on macrophyte abundance, species distrib.	70	30	10
6*	Plume effect on macroalgal abundance, species distrib.	85	15	7
7	Plume effect on zooplankton abundance, biomass, species distribution	16	84	67
8	Survival of entrained zoo-plankton > 80% (positive): < 50% (adverse)	8	75	40
9*	Plume effect on sessile benthos abundance, species distribution	28	72	68
10*	Plume effect on motile benthos	50	50	4
11	Survival of entrained mero-plankton > 80% (positive) < 50% (adverse)	0	100	2

Table 2.4. Continued

Variable Number	Variable of Interest (measured parameter)	Percentage Showing Adverse Impact	Percentage Showing No or Positive Impact	No. of Studies
12	Impingement of epibenthos[2] not significant/significant	0	100	7
13	Plume effects on adult/juvenile fish abundance and distribution	5	83	97
14	Lethal impact of discharge plume on fish population	8 (observed)	92 (not observe)	72
15	Entrainment of ichthyoplankton not significant/significant	11	89	46
16*	Survival of entrainment ichthyoplankton > 80% (positive); < 80% (adverse)	67	33	3
17*	Impingement of ichthyoplankton not significant/significant	32	68	34
18*	Modeling of predator-prey interactions (- or + effect on predator)	67	33	3
19*	Modeling of long-term depletion of important fish stocks	25	75	8[3]

*These variables show adverse impact in $\geq$ 25 percent of cases reviewed.

[1]Eight studies noted change in community structure.

[2]These are valuable species, shrimp, crab, lobsters in marine; mysids and amphipods in Great Lakes.

[3]Two studies considered multiplant effects.

The remaining 10 variables used in the studies reviewed either showed no impact that could be attributed to plant operation, or showed a positive effect. For example, phytoplankton biomass and fish abundance often increased following plant operation. In some instances, sample sizes are very small (e.g., meroplankton survival following entrainment, impingement of epibenthos, community structure analysis of benthic invertebrates). Such variables need further evaluation. For the most part, however, sample sizes were large ($n>30$) and results were quite consistent.

Results from the site-specific analysis indicate that certain test variables and certain types of impacts dominate in different water body types (Table 2.5). For example, macrophyte and macroalgal abundance are impacted frequently in marine, estuarine and lacustrine environments. Since the food web in such systems is frequently detrital-based (Wetzel, 1975), depletion of plant stocks could have adverse implications for other components of the system. Only a very few studies (three) looked at plume effects on mobile benthic organisms in estuaries. Since two of these studies showed adverse impacts, this may be an area that should be investigated further. Impingement and entrainment of ichthyoplankton seem to be a greater problem in the Great Lakes than in the other water bodies. Entrainment seems to be more problematic in marine and estuarine environments than does impingement. In all cases, modeling efforts were only reported for systems having important commercial fish stocks. Such limited use of modeling may be reasonable in view of the fact that there was no depletion of fish stocks in many environments (marine and lacustrine, including reservoirs and Great Lakes), and the associated costs are large.

The analysis indicates that several commonly measured variables may be poor diagnostic tools for impact assessment. These include phytoplankton, periphyton and zooplankton abundance, entrainment

Table 2.5. Site Specific Results of Assessment Review

Variable Number*	Percentage of Studies Showing Adverse Impact** Marine Coastal	Tidal Rivers Estuaries	Rivers	Lakes and Reservoirs	Great Lakes
1	0 (2)	23 (13)	0 (9)	30 (10)	20 (10)
2	0 (3)	0 (1)	28 (18)	66 (3)	0 (4)
3	17 (6)	69 (13)	46 (13)	44 (9)	71 (14)
4	N/A	0 (1)	36 (14)	20 (10)	0 (7)
5	N/A	71 (7)	N/A	66 (3)	N/A
6	66 (3)	100 (4)	N/A	N/A	N/A
7	25 (4)	15 (13)	9 (23)	35 (17)	0 (10)
8	8 (12)	0 (8)	0 (9)	100 (1)	10 (10)
9	20 (5)	0 (15)	8 (25)	14 (14)	33 (9)
10	0 (1)	66 (3)	N/A	N/A	N/A
11	0 (2)	N/A	N/A	N/A	N/A
12	0 (3)	0 (1)	N/A	N/A	0 (3)
13	0 (9)	16 (19)	5 (38)	0 (18)	0 (13)
14	6 (18)	0 (2)	3 (32)	22 (18)	0 (2)
15	N/A	0 (6)	0 (22)	0 (9)	71 (7)
16	100 (1)	50 (2)	N/A	N/A	N/A
17	29 (7)	17 (12)	20 (5)	0 (4)	100 (6)
18	N/A	66 (3)	N/A	N/A	N/A
19	50 (2)	33 (3)	0 (1)	N/A	0 (2)

*Variable numbers follow Table 2.4.

**Data are given as percentage of site-specific studies, with number of studies in parenthesis.

effects on phytoplankton, zooplankton and ichthyoplankton populations, impingement of nekton, and plume effects on population size and distribution of fish, ichythyoplankton, and benthic macroinvertebrates.

The lack of observed changes in fish and macroinvertebrate distribution and abundance variables may be more the result of errors introduced through inadequate sampling methodology and/or extreme natural variability rather than the result of an absence of impact. Since these groups have been shown to be sensitive to other types of disturbance (Cairns, 1979; Welch, 1980; Karr, 1981) and frequently have economic importance, resolution of these methodological problems is necessary before a complete understanding of impacts can be developed. A further discussion of some of these problems is in Chapter 3.

The small amount of data relative to system-wide impacts, trophic interactions and long-term impacts prohibits any conclusion regarding either the occurrence of such impacts or the feasibility of using measurements of such variables for assessment purposes. Similarly, as mentioned previously, bioaccumulation and effects from on-site waste disposal are not represented in the studies reviewed. Tissue analysis of test shellfish (or finfish) may be a useful diagnostic tool for the assessment of bioaccumulation (Goldberg et al., 1978). Measurements of microbial activity (Cherry et al., 1980) appear to provide a rapid, inexpensive test variable for assessment of the potential for impacts due to chemical wastes.

A critical element in the Phase II effort will be the use of expert judgement in aquatic biology and ecosystem analysis to define and prioritize the hypotheses that should be addressed in aquatic ecosystem monitoring design.

STATISTICAL TESTING OF HYPOTHESES

Once a series of hypotheses concerning the impact of power plants on aquatic system has been identified, the next step is to employ statistical methods to test these hypotheses. The framework for hypothesis testing theory consists of selection of null and alternative hypotheses, often denoted H_0 and H_1 , respectively, corresponding to alternate states of nature that the test is designed to detect. The null hypothesis usually corresponds to the state of nature, in which certain 'nominal' conditions hold. For testing against ecological change, the natural null hypothesis is no change (two sided test) or no positive (or negative) change (one sided test). The important concept is that the burden of proof placed on the test is to disprove the null hypothesis. It is emphasized in most elementary statistical texts, and ignored with disturbing frequency in practice, that failure to detect change does not prove the null hypothesis, it merely fails to reject it (Breiman, 1973).

Implementation of hypothesis testing theory requires the specification of a test for discriminating between the null and alternative hypotheses on the basis of a sample data set. Many statistical tests are referenced in textbooks and in the open literature. Usually, the particular form of a given test is the result of two considerations, (1) ease of calculation of the distribution of the test statistic when the null hypothesis holds, and (2) efficiency of the test under certain specified conditions. The background for the first consideration is shown schematically in Figure 2.2. The "recipe" followed in hypothesis testing first requires application of the test, which yields a test statistic, T. T is then compared to a particular range of the test statistic, (T_{c_1}, T_{c_2}). If T lies outside this range, the null hypothesis is rejected; this corresponds to a conclusion that the data indicate some change larger than that which would be expected due to variability from all sources other than the primary effect. Figure 2.2 shows the simplest case, where the probability distribution of the test statistic, T, when the null hypothesis holds is symmetric; therefore, $T_{c1} = -T_{c2}$.

In general, the width of the interval (T_{c_1} , T_{c_2}) decreases with sample size. For instance, consider the well known t-test for the difference in two means. Here, the critical tests statistic is $T_{c_1} = -T_{c_2} = -T_c = S\, t_{1-\alpha/2,2n-1}/\sqrt{n}/2$ where S is the pooled sample

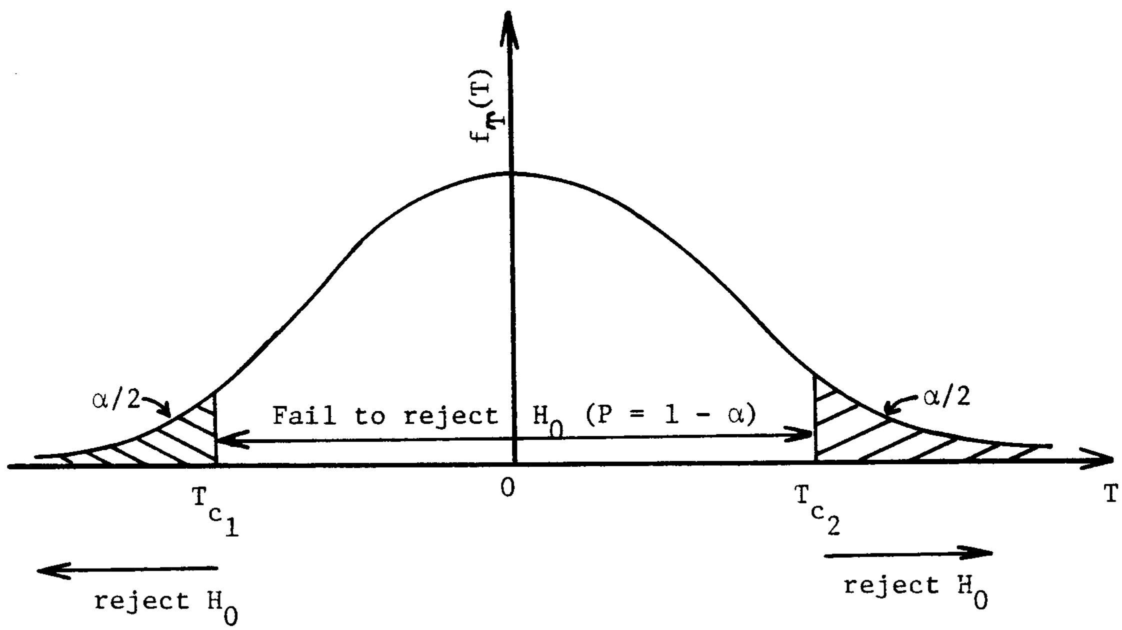

Figure 2.2. Typical distribution of test statistic T for $\Theta = \Theta_0$.

standard deviation of groups 1 and 2 (both having equal sample sizes, n) and where t is the Student's t variate at probability level $1-\alpha/2$ and 2n-1 degrees of freedom, and where is the significance level of the test. In summary, the test statistic, T, under the null hypothesis has a specified distribution (in this case, Student's t) that accounts for the variability of the test statistic from all sources other than the primary effect (the difference in the underlying means). The critical value of the test statistic is set large (or small) enough to reduce the probability that the null hypothesis will be incorrectly (by chance) rejected to .

Normally, α , known as the significance level of the test, or type I error probability, is specified a priori. Of more interest in monitoring design is the type II error probability, β , which is the probability of failure to reject H_0 given that H_1 is true. This is often expressed as the complement of β , known as the power, which is the probability of rejecting H_0 given that H_1 is true. This probability is defined by the critical value of the test statistic, T_c, and by a parameter N (known as the noncentrality parameter) of the form $N = K\ E/\sqrt{n}\sqrt{\sigma}\ n$. The characteristics of a given test are often summarized as power curves, such as those shown (conceptually) in Figure 2.3, where power is plotted as a function of the underlying change level, E, and the noncentrality parameter. For large values of the noncentrality parameter, a given test will have high power for all but very small changes, E. On the other hand, for small values of the noncentrality parameter (small sample size or large standard deviation) the test will have low power even for large changes.

Ideally, one would desire a test with both small α and high power. In practice, to determine the power one must specify α, n, and the level of change, E; the trade-off between the elements controlled by the designer (sample size and, to some extent, variability) becomes apparent.

The desire for high detectability of change, or power, suggests a natural formulation for the network design problem: maximize power for

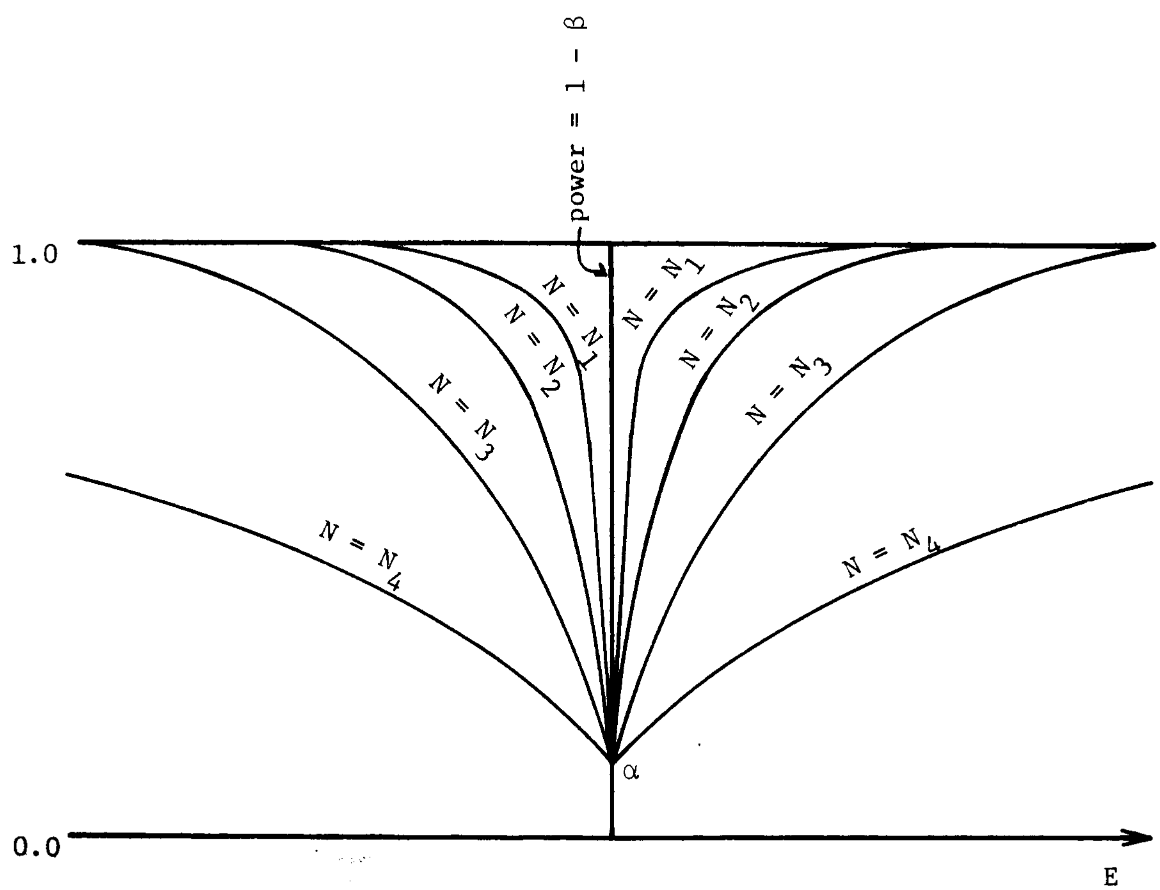

Figure 2.3. Typical family of power curves.

fixed E, α, and n; specifically allocate the (fixed) sampling effort so as to achieve the most effective (minimum residual variance) network for a specified change level. Alternative formulations are to minimize n for fixed E, α, and power or to minimize E for fixed n, α, and power. In such formulations, E is often known as the minimum detectable difference.

Finally, it should be noted that there is a natural relationship between power and N, so that in either formulation the more tractable noncentrality parameter, N, may be used as a surrogate for power. The importance of allocating sampling effort so as to reduce the residual variance, σ_*^2, should be emphasized; this is often the only element of the experimental design that can be controlled by the designer. The use of power functions for determination of the trade-offs between number of replicates, number of stations, and sampling frequency in factorial designs for power plant monitoring has been described by McCaughran (1977a,b), McKenzie et al. (1977), and Moore and McLaughlin (1978). Examples of the application to aquatic sampling design include the Marine Review Committee's Palos Verdes (southern California coastal) benthic studies, the Philadelphia Academy of Natural Science's work on the Chesapeake Bay, and The Analytical Sciences Corporation's (TASC) work at Brayton Point, among others (The Analytical Sciences Corporation, 1979; Academy of Natural Sciences, Philadelphia, 1981; Garrison, 1981).

As indicated above, statistical analysis for change is conducted in a hypothesis testing framework which results in some natural objective functions for network design. For instance, given the existence of two distinct data generating mechanisms (e.g., an ecological indicator sampled at different times or in control and affected zones), such questions as the optimal allocation of sampling to the two groups can be addressed easily. The results obtained are, however, directly tied to the particular test to be used. For instance, if the objective of a network is detection of changes in the variance rather than in the mean of the indicator, a different sampling protocol is indicated.

Further, care must be taken to account properly for all sources of variation. For instance, testing for differences in the mean biomass of a benthic organism inside the area usually dominated by the thermal plume from a power plant, and in a remote control area, can be highly misleading if there are differences not attributable to plant operation in the biological communities in the two areas. For this reason, tests for differences in means are not widely used to detect ecological change. More commonly used are factorial treatment designs, where samples are classified according to as many of the sources of variation noted in Chapter 3 as possible. For instance, sample groups, or blocks, may be defined to account for the physical characteristics of the sites (inside or outside plume area, substrate type, water depth, and season or month of data collection, as well as before/after plant operation). The most common method for analyzing data so classified is the analysis of variance, which in essence tests against the null hypothesis that each of the block means is equal to the grand mean over all blocks. Although this method tests for changes in the mean, it does so by comparing the ratio of the block variances, which has an F distribution, rather than comparing the sample means directly.

Where factorial designs are used, as with paired tests, great care must be taken to avoid confounding the plant effect with other factors. One of the most difficult factors to account for properly in such designs is time. If the response of the biological community to seasonal environmental changes were uniform from year to year, i.e., if seasonal effects were deterministic, inclusion of seasons as factors would present no problems. However, large variations in biological response do occur from one year to the next and are attributable to variations in light, temperature, and other climatic factors that may or may not be classifiable. Therefore, an improperly designed experiment may result in a conclusion of significant change before and after plant operation, which in fact has no relationship to the existence of the plant whatever. The difficult aspect of accounting for temporal variability is that the only way properly to characterize it is to collect data over a long period of time. All other sources of variability can be resolved, at least in theory, through additional short-term sampling effort. This problem, which argues for the use of time series analysis as opposed to factorial designs, is discussed later in this chapter.

In addition to paired tests and factorial designs, which address the question of changes in means, methods are also available for testing differences in variability (e.g., variance, ranges) and extremes. Tests can also be made for change in the entire probability distribution of a variable. Other tests, which recognize the multivariate nature of biological indicators, can be made for changes in diversity indices and for other measures of change in community structure. In many cases, the distribution of the test statistic can be derived or approximated in closed form; in others, it must be estimated by Monte Carlo simulation.

The complexity of the monitoring design problem should be apparent from the variety of ways in which ecological change can be manifested, and from the variety of tests that may be used to assess change. The problem is that, while it is at least theoretically possible to formulate an optimal monitoring design for any fixed objective, the form that ecological change associated with power plant operation may take, if any, is usually not known a priori. Further, even for a particular type of effect and associated test, monitoring protocol will be dependent on such additional considerations as the probability distribution of the data, which usually cannot be identified from even modest sample sizes.

Clearly then, the selection of objectives is a critical step in the monitoring design process. What is needed are monitoring designs that are robust with respect to a range of objectives, insofar as the type of change may not become apparent until after data collection has begun. Although there have been several papers published in the ecological literature on optimal monitoring design (Saila et al., 1976; TASC, 1979), and quite a few in the water quality field (e.g., Dandy, 1976; Kitanidis et al., 1978; Moore, 1973), all of these are methodologically oriented, i.e., they emphasize the techniques used in the optimization, and not the selection of objectives. To our knowledge, no previous work has been conducted in the aquatic community to address the problem of multiple monitoring objectives.

SUMMARY

The following points summarize Chapter 2.

- The National Environmental Policy Act of 1969 (NEPA) and the Clean Water Act of 1977 (CWA) provide the regulatory basis for monitoring of aquatic impacts due to power generation.

- The Environmental Protection Agency (EPA) is the lead regulatory agency having authority over water quality and aquatic monitoring at electrical power plants. The Nuclear Regulatory Commission (NRC) is party to the establishment of monitoring the programs at nuclear power plants but is subject to EPA authority.

- Regulations under the CWA require monitoring according to NPDES discharge permit conditions and for demonstrating the acceptability of impacts due to thermal discharge, cooling water intake and ocean discharges.

- Regulatory specifications for aquatic monitoring programs are very general; and, although technical manuals exist, the specific design of monitoring programs has been on a case-by-case basis.

- Specific hypotheses of impact and expected levels of change due to plant operation are rarely defined as part of the design of a monitoring program.

- A review of 162 monitoring programs showed that 19 different variables were measured frequently (Table 2.3). Most variables tested effects due to heat or entrainment/impingement. Seventeen of these variables evaluate short-term impacts on specific individuals, populations or communities of aquatic organisms. Two variables evaluate long-term impacts at the population or ecosystem level; long-term evaluations were done only at eight plants.

- Nine of the test variables consistently showed adverse impacts that could be attributed to plant operation (Table 2.4).

- Several commonly measured variables (including phytoplankton, periphyton and zooplankton abundance, and entrainment effects on phytoplankton, zooplankton) did not prove to be very useful in identifying impacts due to plant operation.

- Other variables (including plume effects on population size of fish, ichthyoplankton and benthic invertebrates, and long-term effects on finfish stock size) had limited utility due primarily to variability associated with sampling techniques and/or excessive amounts of spatial and temporal variability in the natural population.

- Bioaccumulation effects on shellfish populations were measured only rarely.

- The development of statistically valid monitoring programs requires the identification of hypotheses of impact, control and non-control sites and appropriate test variables. The most commonly used null hypothesis (H_o) is that there is no change in the test variable.

- The power of a test (probability of rejection of H_o given H_1 is true) is defined by the critical test statistic T_c and the noncentrality parameter N, which is a function of the expected level of change, E, the sample size, and the residual variability.

- Power, or detectability of change, serves as a useful objective function for development of optimal monitoring programs.

- The optimal allocation of sampling effort depends upon the ability of the investigator to test the hypothesis of impact, the natural variability of the test variable, the ability of an investigator to sample the test variable adequately in control and non-control areas, the extent to which such efforts can maximize the power of the statistical test, and the choice of the test itself.

- Season-to-season and year-to-year temporal variability in natural populations presents the most serious limitation to the design of monitoring programs that test specific hypotheses of impact at the population level in a statistically valid manner.

CHAPTER 3

METHODS FOR THE DESIGN OF MONITORING NETWORKS

INTRODUCTION

Chapter 3 reviews methods that have contributed to the design of aquatic monitoring and presents an analysis of the results of those efforts. Included are computer-based data handling approaches, model applications, statistical techniques, an analysis of monitoring cost-effectiveness, and available sampling procedures.

Some of the reported methods have not as yet received wide acceptance for use in aquatic monitoring in the electric utility industry. In some instances, however, experience with these methods in other settings shows promise of benefiting monitoring design at power plants as well. Where this is true, research will be required to investigate the usefulness and performance of the techniques in power plant applications. Some methods are likely to require adaptation and further development in order to be employed in these applications. These research and development efforts will be pursued in Phase II of the project.

DATA MANAGEMENT/STATISTICAL ANALYSIS SOFTWARE

A typical monitoring program can generate very large quantities of data. For each sampling station, information on dates, times, locations, etc. must be recorded to identify each data set. Replicate data sets must be labeled, and there may be numerous parameters observed at each time. Data must be edited to determine whether stations are correctly labeled, and whether the data are acceptable. Different contractors often monitor each trophic level, introducing another complication. Software must be developed or selected to file, edit, retrieve, and display these data. If each contractor is permitted to select his own computer and software system, the probability for the development of serious interface problems rises. Furthermore, if each contractor employs a different set of statistical software to conduct routine or special statistical analyses, the results may not be comparable.

One of the tasks of a monitoring design protocol is to learn from past monitoring efforts. This requires that statistical properties of the data can be readily examined and that proposed sampling designs can be evaluated by estimating the characteristics of the data to be produced. While the handling of data by a computer is technically feasible, the lack of standardization of such technology can often lead to serious demands in project resources to complete these routine tasks.

EPRI has funded several major efforts to develop data base management systems for aquatic impact assessment (Ecological Analysts, Inc. et al., 1981; Atomic Industrial Forum et al., 1978; Tetra Tech, 1979). Environmental information has been organized in the form of computerized bibliographic data bases by EPRI (Ecological Analysts, Inc. et al., 1981; Atomic Industrial Forum et al., 1978) as well as by National Technical Information Service and other abstracting services. Each utility or consultant routinely develops computerized data bases or monitoring data on aquatic impacts. EPRI Research Project 1489, under the management of I. P. Murarka, (Ecological Analysts, Inc. et al., 1981) developed a pilot data base system to store, retrieve, and analyze data from aquatic environmental studies. The emphasis of that project was to integrate existing software into an easy-to-use operational environment. The UPGRADE analysis system developed for the Council on Environmental Quality (Ecological Analysts, Inc. et al., 1981) was the basic building block of the prototype system. The basic data maintained in the system were site characteristics, water body characteristics, station characteristics, species lists, and time-oriented observations. UPGRADE provides user-prompted graphics, mapping and statistical packages. Since many utilities have used the Statistical Analysis System (SAS) in their environmental data analysis, the prototype system has a full interface with SAS (to be discussed later). The commercial data base management system IDMS (Cullinane

Corp.) was selected over System 2000 for use in the pilot system. The prototype system was implemented on the Boeing Computer Services system, which has a telecommunication network to over 400 cities in the U.S. and offers both IDMS and System 2000 to users. Both text data and numerical data were placed in the prototype system. The cost of tape conversion and data loading was found to exceed this project budget allocation, and all data were transcribed from hard copy reports, except for STORET data (which already existed on UPGRADE).

The experience with the prototype environmental studies data base pinpoints the major problem of data management. Once a data base is developed on a given computer, the problem of accessing these data on another computer remains. If the transfer is to be made to an identical computer at another location, then tapes can be made and transported. If the computers have different operating systems, hardware or software characteristics, then a translation must be made. Soi and Aggarwal (1981) summarized surveys of computer networks that could permit direct transfer of data between computers:

> "As an advancing technology, the evaluation of large data networks offers many unusual and challenging problems to the designer, manager, and operator.... Larger scale distributed networks still need extensive research before they become a practical and economic reality."

Feng (1981) and Power (1981) reported the progress made by manufacturers of large computers to permit networking between classes of their own computers. Bass (1981) reviewed progress made by research establishments to link their own in-house computers and the progress of the International Standards Organization (ISO) (Schindler, 1981) reference model proposed to standardize interconnections.

Given this state of computer networks, with existing off-the-shelf capability to access many computers through a terminal network, but with major problems to interconnect computer-to-computer, the manipulation of data collected by utilities will most probably continue to be perfomed by transfer of tape files or by access to data files via a microcomputer used as an interface terminal. The use of the UPGRADE-based environmental data base is currently beyond the state-of-the-art, but the use of such a system to share and display data from a given file is feasible using interactive terminals.

Sutron Corporation (1981) has demonstrated the development of real time acquisition of data for hydrologic data bases, and others have developed similar systems for acquisition of air quality data. The cost of such systems is several hundred thousand dollars for computer hardware, plus an equal amount for software, and about one hundred thousand dollars for a central receiving station. Sensors and transmission equipment can cost tens of thousands of dollars per station, and maintenance costs require one percent of the capital cost per month. Given the large cost of data base management and acquisition systems, Phase II of this project can effectively focus only on demonstrating that interactive and incremental evaluation of monitoring data to improve monitoring design is feasible.

Software packages are computer dependent and the decision to use existing software or write new software will depend on the available or selected computer, the level of software development or support personnel available, and the software writing capability of the project personnel doing the sampling.

The Ecological Analysts, Inc. et al., (1981) project on environmental studies data bases selected the computer where the UPGRADE system was developed and then selected a data management system available on that computer (IDMS) on the basis of least cost. The Marine Ecological Commission (MEC) computer selection provides a contrasting perspective. Their computer was selected on the basis of the choice of the SAS software package for statistical analysis. Most utilities appear to employ IBM equipment that supports the SAS software for their statistical analysis of aquatic data. Introduced in 1970, the SAS provides data storage and access, report writing, statistical analysis and file manipulation to over 2,000 users. Data may enter from any input device in any format. SAS requires IBM hardware and is not available on the University of Washington system, but can be accessed using the Washington State University IBM computer network. SAS features over 60 statistical procedures, including variance and covariance analysis, nonlinear regression, nonparametric analyses, and time series analysis. An interactive graphics system, SAS/GRAPH, is also available.

An alternative software system, the Statistical Package for the Social Sciences (SPSS), was established in 1968 and now has over 2,500 users. The system is compatible with many computers. SPSS is available on the University of Washington CDC system. SPSS is user oriented and has the ability to read almost any raw data file. Files are readily accessed, sorted, merged, and saved. The recent addition of a report writer has increased flexibility, and an interactive version is available. SPSS performs most commonly used statistical tests and time series analyses including partial correlation analysis, variance and covariance analysis, linear and nonlinear modeling, Box-Jenkins time series analysis, and nonparametric tests.

SAS and SPSS file manipulation abilities are comparable. Both systems contain the basic statistical procedures, but it appears that SAS is superior for complex and non-numeric statistics. The SAS programming language is more difficult to learn and to use than are the simplified SPSS commands. One limitation of SPSS output is that only four significant digits beyond the decimal point are displayed. SPSS does have simpler file management procedures (easier to use by non-computer-oriented personnel). Green (1979) has reviewed over thirty other statistical packages, but places SAS and SPSS as the most commonly used system.

As mentioned earlier in the discussion of the pilot EPRI data base system (Ecological Analysts, Inc. et al., 1981), once the computer is selected and the statistical software is chosen, a data base management system (DBMS) must be developed or acquired. The IBM compatible System 2000, the Cullinane Corporation IDMS, and the Boeing/UW RIM systems are typical of the large-scale data base management systems in use. Major conceptual differences in the management of the data base exist, with hierarchical structure concepts of System 2000 and IDMS differing from the relational algebra model of RIM-type systems. While data base management systems are expensive to develop or acquire, and labor intensive to operate and maintain, they do not present a major technicalh problem for monitoring network design. The choice of the DBMS is usually one of convenience for the researchers, based on available computers and the available software and on the preferences of the computing staff.

Technology to permit direct communication between large computers is evolving rapidly, but is not cost-effective at present. Phase II of this project cannot make a significant contribution to this industry-wide problem. Existing software for statistical analysis of data and graphic display of these data are widely used by the aquatic monitoring community. The major need of aquatic monitoring designers may be for an interactive software system that can display monitoring network designs and provide specific statistical measures of the effectiveness of these networks. The ability to update these effectiveness measures with new data would be facilitated by an interactive system. In addition an interactive system could direct a researcher to the availability and advantages of employing modeling techniques or other assessment methods (e.g., time series analysis) for a specific monitoring network. Cost-effectiveness models (e.g., linear or dynamic programming) might also be usefully employed within an interactive system. In such a fashion, preliminary data, collected during pilot sampling programs, could be rapidly incorporated into the network design process. The following two sections discuss the analytical and cost-effectiveness models that have been used in past monitoring programs. Some of the errors inherent in sampling for specific populations and alternative sampling techniques are discussed in the final section of this chapter.

MODELS USED IN AQUATIC IMPACT ANALYSES

While there has been limited use of models in plant siting and impact assessment, the specific use of models to design a monitoring system has rarely occurred. The following discussion surveys the literature describing aquatic models and evaluates the model use potential for aquatic monitoring system design. Before presenting a detailed description of generic types of models that address various aspects of power plant impacts on aquatic systems, a brief history of model evaluation is presented in order to establish the strong disciplinary biases and precedents that influence the choice of models.

Weaver (1948) has observed that the physical sciences have had two centuries to develop two-variable science to a highly refined activity. On the other hand, during that time, the biological and social sciences focused on qualitative activities that collected, described, classified, and correlated subjects. Eventually, the disorganized complexity of these numerous subjects was characterized using statistical techniques that described the behavior of the subjects in terms of average properties, distributions of attributes, and correlations. The development of the digital computer has permitted the manipulation of sets of complex interdependent statements describing systems of complex interactions of a moderate number of variables, but very few data sets exist to validate such models. While there has been some cross fertilization of various scientific disciplines, physical scientists and engineers tend to construct two-variable models of greater complexity (for example, three-dimensional models of the movement of water in time and space), and biological scientists tend to focus on descriptive studies of particular species. Both of these basic disciplines tend to reject the complex, multiple variable, dynamic models, suggested by ecosystem concepts, since those models are major departures from existing paradigms (Mar, 1978).

A classic illustration of these strong disciplinary views is the debate by Lauer and Christensen (1981) at the Fifth National Workshop on Entrainment and Impingement. Lauer argued that generic and site-specific data interpreted by an expert familiar with any given site can provide information in a more efficient and comprehensive fashion than can models. Models were described as difficult to understand (complex mathematical notation), fraught with hidden assumptions and untested assertions, and requiring more data to test than an expert needed in order to resolve any given question. Christensen countered with the difficulty of extrapolating statistical data to cases where potential impacts have yet to be experienced and the difficulty of conducting site-specific impact studies when it is infeasible to establish a control for statistical analysis.

There exists a wide spectrum of model types ranging from simple conceptual models, fundamental physical models (e.g., $F = ma$, $E = mc^2$), and empirical models validated on exhaustive data bases, to complex, dynamic, multi-parameter, nonlinear ecosystem models such as proposed by Tetra Tech (1980). The use of complex models to extrapolate existing knowledge and to formulate statements that estimate causal relationships is still a pioneering activity. Such models will not succeed until experts in fields representing the bodies of knowledge integrated in the models accept the models' descriptions. The acceptance of particular models increases as the body of knowledge they encompass is tested and retested. Thus, models, like statistical data or experience, gain acceptance with time and use. Since ecosystem models are relatively new and the integration of models of various compartments in a food chain is not an accepted activity by many biologists, the acceptance of ecosystem models will occur slowly, while the acceptance of physical models of flow or heat dispersion has already occurred over the past century.

Generic Types of Models Used in Aquatic Impact Analysis

Five types of models used to address aquatic input are shown in Table 3.1. This literature review presents the basic concepts of each generic type of model and of the potential of the models to assist in the definition of what to look for, where to look, and how often to look in order to detect an impact with a given level of confidence or power.

Table 3.1. Summary of Generic Models Reviewed

- Circulation and hydrodynamic models
  - Open channel flow
  - Two-dimensional stratified flow
  - Three-dimensional tidal flow
- Plume Models
  - Sewage plume models
  - Thermal plume models
- Source Models
  - Entrainment
  - Impingement
  - Biocides
  - Nonpoint
- Water Quality Models
  - Temperature
  - Dissolved Oxygen
  - Toxicity
- Aquatic Ecosystem Models
  - Steady state
  - Dynamic/spatially integrated/passive populations
  - Dynamic/spatial detail/passive populations
  - Dynamic/spatial detail/mobile population

Hydraulic Models

A fundamental building block of most studies of aquatic impacts of power plants is the description of the circulation or flow of the intake or receiving waters before and after the introduction of power generating facilities. As noted by Baca et al., (1973): "The problem of mathematically modeling the hydraulics of most fluid systems is not, in general, a problem of formulating theoretical constructs or governing equations but rather of devising workable numerical techniques to solve established hydrodynamic or hydraulic equations." The dynamics of any fluid system are governed by laws of conservation of mass, momentum, and energy. Open channel flow models used to describe rivers, streams and unstratified impoundments employ one-dimensional inviscid flow, hydrostatic pressure, and incompressible flow. More complex models employ two-dimensional inviscid flow, wind stress, and density stratification to describe flow in lakes and reservoirs. Still more complex models add geologic forces such as tides, Coriolis force, etc. to those used in lake models and may use three-dimensional formulations to describe tidal flow in estuaries and bays. The level of detail used in describing the spatial and temporal characteristics of these waters controls the complexity, accuracy, and cost of the model results. Models can vary from simple annual water budgets to advanced real-time modeling of complex estuary hydrodynamics such as those models developed by Leendertse (Tracor, Inc., 1971).

Given the long history of mathematical modeling of fluids, there is general acceptance of the basic theory of numerical methods used in the practice of power plant design. Precise description of localized velocity fields is still a challenging activity, but generalized description of averaged or aggregated flows by models can be used in the design of monitoring networks to define flow fields.

Plume Models

Historically, the major application of plume models in water has been in the study of wastewater outfalls. The study of thermal discharges presents much lower dilution rates than encountered with wastewater outfalls (10:1 rather than 100 or more:1) and much larger volumes than municipal discharges. Thermal discharges have higher momentum associated with larger volume flow rates, but the application of momentum jet theory has been limited by the ability of hydrodynamic models to provide at a reasonable computational cost sufficient detail of the velocity fields near the discharge. Tracor, Inc. (1971) has reviewed the earlier models that employed dimensional scaling factors and analytical solutions. Asbury and Frigo (1971) methods are typical of the empirical scaling models used to estimate thermal plume area, prior to use of complex hydrodynamic models. More recent efforts have incorporated the momentum jet into the hydrodynamic models discussed in the last section and solved for results with and without a discharge. Plume models used in the design of outfalls or discharge canals should provide data necessary to estimate the near- and far-field mixing zone.

Source Models

The major concern in power plant impact studies has been physical and thermal damage due to entrainment or impingement of fish and other organisms and toxicity of biocides. Models of survival rates for entrained or impinged organisms are well developed (Tetra Tech, Inc., 1979) and could be useful in predicting where and when survival could occur and the densities of organisms that could be anticipated in the discharge. Similarly, research into the impacts of chlorination has provided sufficient data to suggest that impact areas may be estimated and simple sampling designs established.

The formulation of models of releases of other power plant pollutants (e.g., nonpoint sources such as coal pile runoff and ash leachate, plus point sources such as scrubber waters, blowdown, etc.), that may contain organic and metal pollutants, is in its infancy. TRC-Environmental Consultants Inc., (1981), proposed a model of coal pile runoff based on the Ohio State version of the Stanford Watershed model coupled with models of chemical and physical leaching of the coal. While there is a wide spectrum of storm runoff models that describe water quantity and quality, the TRC effort is one of four that applies these models to power plant runoff. There is sufficient knowledge of nonpoint runoff models that their application to simplify monitoring network design is feasible.

Water Quality Models

Much of the interest in mathematical models of water quality has been motivated by a concern for impact of discharges upon dissolved oxygen and temperature of the receiving waters. Historically, the development of these models has proceeded in the following way:

1. Formulation of the concept for one or a limited number of parameters;
2. Application of the concept to a simple problem;
3. Refinement and improvement of concepts and solution techniques;
4. Application to multi-dimensional problems with complex boundary and/or initial conditions; and
5. Incorporation of individual processes into the larger framework of a multi-parameter model without major changes in the original concepts.

The last step is one in which the developers and users of water quality models have begun to make use of the principles enunciated by ecologists. Requirements for careful analysis of thermal discharges emerged during the late 1950s. It had become apparent that power plant condenser cooling water discharge and the impoundment of rivers for hydroelectric projects were modifying the temperature regime of rivers, lakes, and estuaries. Initial models of cooling water discharges (Edinger and Geyer, 1965) and of impoundment of freely flowing streams and rivers (Burt, 1958; Raphael, 1962) formulated the heat transfer across the air-water interface either in terms of the heat budget directly, or indirectly through a Taylor series expansion of the heat budget in the neighborhood of the expected water temperature. The models were formulated in terms of linear first- or second-order differential equations or box models, eliminating the need for complex solution techniques.

Primarily as a result of advances in solution techniques, these models have been extended to complex environments with time-dependence and, in some cases, more than one spatial dimension (Hydrocomp, Inc., 1976; Jirka, 1977; Watanabe and Connor, 1976). Despite the sophistication of these models, they have been limited to one of the following scales:

1. Near-field where jet mixing and advection are important;
2. Intermediate or transition region where advection and buoyant spreading are important; and
3. Far-field where advection, diffusion, and surface heat loss predominate.

Stolzenbach and Adams (1979), however, have recently developed and applied techniques incorporating the models of all three scales.

Relatively little change has been made to the heat budget terms, although Wunderlich and Gras (1967) have done a thorough review of the state-of-the-art.

The consensus of the literature is that temperature models for rivers, lakes, reservoirs, and coastal regions are reliable and accurate. In general, the major difficulties in thermal modeling arise from incomplete knowledge of (1) vertical or horizontal coefficients of eddy diffusivity, (2) turbulent transfer of water vapor and sensible heat across the air-water interface, and (3) velocity fields in the receiving waters. The size of the error associated with the latter is related to the spatial and temporal complexity of the hydrodynamics. One might expect, therefore, the uncertainty associated with simulations in estuaries or coastal environments to be greater than in rivers, with simulation uncertainty for lakes and ponds falling somewhere in between.

Mathematical models of dissolved oxygen have changed little in concept since the work done by Streeter and Phelps (1925) and have little direct relationship to power plant impacts. Oxygen is a primary state variable in ecosystem models, and the Streeter-Phelps formulation continues to be used. The number of source and sink terms has been expanded to include sediment oxygen demand, nitrogenous oxygen demand, algal respiration and photosynthetic production of oxygen (O'Connor and DiToro, 1966). A great deal of effort has been devoted to obtaining consistent estimates of important rate constants (U.S. EPA, 1972; Covár, 1976; Zison et al., 1978). The major advances, however, have been in the development of solution techniques, making possible the application of the model to a wide variety of problems (Crim and Lovelace, 1973; Roesner et al., 1977; Dailey and Harleman, 1972; Johanson et al., 1976).

The water quality modeling approach to toxicity has been to describe the initial dilution and subsequent advection and diffusion of individual chemical species. The resulting concentrations have been compared with laboratory bioassay data. Available transport models for lakes, rivers, and coastal areas have been used to describe the hydrodynamics, and models also are available to describe the chemical equilibrium (Morel and Morgan, 1972). Acute toxicity data are available for many important or indicator organisms, though these estimates may span two orders of magnitude. Information on chronic toxicity is less reliable. A common, and frequently criticized, approach has been to apply factors (generally equal to 0.1 or 0.01) to the acute toxicity LC50s and to use the results as estimates of chronic toxicity. There still is a great deal of uncertainty in establishing water quality criteria, and model formulations are not rigorous enough to gain acceptance.

Aquatic Ecosystem Models

Although advances in the knowledge of ecosystem interactions have been made, the development of ecosystem models and their application have been influenced primarily by those concerned about either predicting or controlling environmental impacts. While the distinction is becoming fainter as those concerned about environmental impacts become more eclectic and as ecologists become more involved in analysis of environmental impacts, there are still traces of these two viewpoints.

The concepts of conservation of mass and linear, or quasi-linear, transformation processes, expressed in differential form, have proved very useful for dissolved oxygen and temperature models. These same concepts had been used by ecologists (Riley et al., 1949; Riley, 1965; Steele, 1965; Lotka, 1956) to describe structure and productivity. It is, therefore, not surprising that as the consciousness of environmental impact analysts expanded, the two approaches would merge. Chen's (1970) conceptual model is one of the earliest examples. Subsequent models, similar in construct, are those of DiToro et al. (1971); DiToro and Matystik (1980); Baca et al. (1973); Scavia and Park (1976); Behren et al. (1975); and Water Resources Engineers, Inc., (1975). The most comprehensive ecosystem models of this type are those described by Tetra Tech (1979), and Larimore et al. (1979). These models have made extensive use of the concepts developed by ecologists, but were constructed for the purpose of control and/or management.

Models developed for the purpose of studying structure and productivity have, in many cases, used the linear or quasi-linear systems approach, too. The energy/matter circuit approach (Odum, 1960; Hall et al., 1977) was developed to examine the flow of energy and matter in ecosystems (Odum, 1957). Although developed as a systems approach to community structures, it also has been applied to a wide variety of environmental problems (Jansson and Wulff, 1977; Kemp et al., 1977; Kelly and Spofford, 1977; Hall, 1977; Richey, 1977; McKellar, 1977).

There have been a number of well-designed studies to assess the completeness of the ecologic models dealing with eutrophication (i.e., nutrient cycles and primary productivity, (Moore et al., 1976; Canale et al., 1980; DiToro and Matystik, 1980; DiToro and Connolly, 1981; Scavia et al., 1981a,b). Some of these have applied state estimation techniques, first-order uncertainty analysis, and Monte Carlo methods to the problem of uncertainty in eutrophication models. These results indicate that, for the nutrient cycles and primary production, the ecologic models are valuable tools for environmental assessment.

Such is not the case for constituents in the more comprehensive models such as benthic organisms, macrophytes, periphyton, and the various species and life stages of fishes. Swartzman et al. (1978), for example, reviewed the fish population models that are used to assess entrainment mortalities and concluded that, "...no presently existing impact model can be used to make quantitative predictions." Hall (1977) was somewhat more optimistic, but because of the variability in fish population densities, as well as in spatial distributions, it is hard to place much confidence in the simulations. It might be expected that the chance of doing so would be greater in a controlled environment such as a lake. The most recent studies (Larimore et al., 1979) are not helpful on this subject. Well-designed studies of fish populations and life cycles are needed in order to develop confidence in fish population simulations.

The spectrum of ecosystem model complexities ranges from simple nomographs (Vollenweider, 1969; Schaefer, 1957) to complex hydrodynamic-ecosystem formulations (Tetra Tech, 1981a), but modifications are required to permit estimates of variance in space or time. The problem with complex models is that hundreds of parameters are included in the formulation, and most of these parameters are taken from the literature or are estimated. The amount of on-site data required to parameterize these complex models for precise predictions on a site-specific basis appears to be prohibitive. If one were hoping to use models to extrapolate between two observations in space or time, the use of a model or relationship containing hundreds of constants would be impractical. Instead, the least number of constants would be sought.

The primary role of complex ecosystem models appears to be as a framework and laboratory to test the formulation of hypotheses concerning the interrelation of compartments or physical factors. These complex models can be used as laboratories to define the sensitivities of one compartment to changes in other compartments and to validate simpler models that abstract detail or interactions.

The use of large, complex, deterministic models in sampling network design appears limited to answering spatial and temporal suggestions on where the organism is. They also can establish dilution patterns or define flow patterns for organisms or pollutants that are swept by the currents. Another use of these models may be to establish the relative importance of pollutants or organisms on a specific organism (suggesting what to look for). The weakest models appear to be those predicting the migration patterns of mobile organisms in time and space. The forecast of when and where to sample algal blooms or schools of fishes is difficult.

EPRI has funded Lawler, Matusky and Skelly Engineers (1980a, 1980b) to prepare handbooks of methods for assessing population and ecosystem effects of impingement and entrainment of major fish and invertebrate species as well as the entrainment of phytoplankton and zooplankton. The project reports are comprehensive and describe sampling methods for estimating cropping and population size, physical models for estimating entrainment, and methods of estimating entrainment/impingement effects on long-term population dynamics. Over two hundred references are cited, and complete quantitative descriptions of sampling designs and analytical methods are presented. Included are reviews of the extensive experience of the authors, developed in the evaluation of impingement and entrainment impacts. In comparison with their previous efforts (Lawler, Matusky and Skelly Engineers, 1975, 1979) (LMS), the physical models proposed for estimates of entrainment effects in the handbook were simple mass balance models, rather than the complex transport equations of dispersion and advection found in hydrodynamic models.

Swartzman et al. (1978) have reviewed the LMS models as well as the 1973 Oak Ridge National Laboratory (ORNL) models of the Hudson River and the Summit plant, the United Engineers and Constructors 1975 model, the Johns Hopkins University model developed for Summit, and the University of Rhode Island (URI) model developed for Millstone. While the Swartzman et al., (1978) analysis was not included in the Lawler, Matusky and Skelly Engineers (1980a) handbook, the findings were consistent. Table 3.2 from Swartzman et al. (1978) summarizes the major attributes of the models examined. The LMS models were found to contain more information and complexity in their description of egg production, aging and recruitment, and swimming ability than the other models. Swartzman et al. (1978) observed that the LMS models use a density dependent mortality model that strongly compensates for losses induced by entrainment. A dilemma was identified concerning whether to focus on catch statistics, egg production, or juvenile densities to estimate population impacts. Using sensitivity analysis of the model parameters, Swartzman et al. (1978) demonstrated that the compensatory mortality function is the dominate factor in these models. The number of physical segments, the entrainment factor, and the advection and avoidance factors combined could account for sensitivities approaching the compensatory mortality factor.

A review of mathematical models used in environmental impact assessment for stream electric power plants published by the U.S. Fish and Wildlife Service (FWS) (Bloom et al., 1977) was found to be qualitative but representative of the reservations held by many on the use of the models examined by Swartzman et al. (1978) and Lawler, Matusky and Skelly Engineers (1980a). Major reservations are the lack of consideration of interaction with the dynamics of other species and habitat dynamics, the lack of detail on impacts of perceived importance, (i.e., biotic community, local habitat, etc.), and the lack of good information on the physiological and behavioral characteristics of the populations of interest (see Table 3.3). An example of the view of a Fish and Wildlife Service scientists is the testimony of Goodyear (1979) criticizing the use of data in evaluating the intrinsic growth rate parameter in the Ricker Model used for the Hudson River striped bass population. The major point concerned the lack of accurate and long-term catch data to separate sampling errors from power plant impacts.

Table 3.2 Comparison of Model Approaches.*

Criterion		LMS	ORNL	ORNL Summit	JHU	URI
Compensation	Density-Dependent					
a.) y-o-y	Highly d-d	✓				
(young-of-year)	Weakly d-d		✓			✓
	d-i (density-independent)				✓	
b.) fishery d-d model			✓	✓		
Hydrodynamics						
Use detailed Hydrodynamic Model		✓	✓		✓	✓
Model Approach						
Use fisheries catch data			✓	✓	✓	
Use detailed y-o-y sampling data		✓	✓			
Adult Impact Assessment						
Use Leslie matrix		✓	✓	✓		✓

* Swartzman et al. (1978)

The current state of entrainment and impingement impact models reflects the strong bias of the biologist and modeler for detail in their area of greatest interest. The overemphasis on detailed hydrodynamic entrainment models appears to be ebbing, and new thrusts are being made to refine compensatory mortality submodels and to seek longer term time series data for improved population dynamic models. The inability of most of the present models to recognize species interaction between fishes and between predator and prey remains.

Relatively simple models of entrainment, entrainment/impingement mortality, and first-year recruitment could be useful when combined with survey and preconstruction data to define the number and frequency of monitoring stations needed to detect long-term power plant impacts. The greatest need appears to be cost-effective monitoring design to describe the dynamics of the fish populations in space and time. Focusing on the short-term and near-field impacts of the power plant on fish alone is inadequate to estimate long-term impacts. Haven (1975) suggested the combination of input/output theory and dynamic modeling as an alternative framework for biological impact modelings. The use of input/output theory reported by Lettenmaier and Richey (1978) in carbon budget analysis has yet to be applied to ecosystem analyses.

The models proposed for evaluating power plant impacts on fish populations require physiological and behaviorial data that are not routinely obtained in impact monitoring studies. The design of intensive experiments to define rates of growth, death, predation, etc. does not appear compatible with routine impact monitoring that occurs at infrequent intervals in time. Also, the time series of data for catch or for stock sizes needed to parameterize a population model are not produced by short-term sampling efforts. If models are to be used to design impact sampling systems, more experimental studies of rates and long-term time series studies will be necessary.

Hydrodynamic models are based on well accepted principles of conservation of mass, momentum, and energy. The accuracy of these models depends upon the spatial and temporal detail selected and the

Table 3.3 Fish and Wildlife Summary of Biological Impact Models (from FWS 1978).

	Model: Millstone	Model: Hudson River	Model: Crystal River
Issue	Entrainment of winter flounder eggs and larvae on population in Niantic Estuary.	Examination of power plant impacts on young-of-the-year striped bass.	Examination of power plant impacts on biomass and biomass transfer rates near plant.
Hydrodynamic Model	287 sections, 305 meters square; 60 second time step; needs tidal input and transport and growth of eggs and larvae. Uses standard hydrodynamic model, dispersion, Coriolis, advection, etc.	72, 2 mile reaches 4 hour time step; transport and growth of eggs and larvae included; uses tidal averaged flows.	Simple mass balance of bay - no spatial resolution.
Population Model	Use eggs, larvae, yearling and 12 adult age classes; density dependent survival of eggs and larvae; other age groups density independent; very little effect of discharge	Included in hydrodynamic model to account for rapid growth and death of eggs and larvae; model verified by lumping one week of data for ten week segments.	5 compartment ecosystem model phytoplankton, zooplankton, macro fauna, detritus and nutrient; need initial biomass and rates of flow within and between compartments; verified with three years data and sensitivity analysis.
Reservations	Model of biology not verified; circulation model verified, but ignores wind and density gradients; no vertical movement of eggs and larvae; natural mortality effects not recognized; adult age classes should be combined, since no density dependence.	No consideration of impact from natural factors, other species predators, etc; model required extensive behavior and physiological data of striped mass; too much emphasis on young-of-the-year and no consideration of total population.	Does not address individual species; ignores salinity; no spatial detail.

numerical techniques selected for integration. Both fluid motion and temperature patterns can be readily predicted by these models.

Aquatic ecosystem models have been patterned after the hydrodynamic models using sets of linear or quasi-linear differential rate equations to describe the biological processes of growth, death, predation, etc. These models require many more parameters because each biological component becomes a state variable, and there are inadequate field data to validate these models. Furthermore, the causal relationships are not well established. The combination of active organisms and passive organisms in a single model presents challenging conceptual problems. In most situations, there does not exist adequate time series data to calibrate and validate dynamic models of annual biological populations.

While hydrodynamic models can provide insights where passive organisms may be abundant, the lack of time series data precludes good estimates of where active organisms, especially fish, will be. The large aquatic ecosystem models may provide insight into the interaction between biological components, and the possibility that impacts on one compartment may impact other compartments, but the models must be reformulated to provide probabilistic rather than deterministic predictions.

STATISTICAL ASPECTS

Background

From a statistical standpoint, the problem of detecting long-term ecological change is one of discriminating power plant effects from other sources of variability affecting the aquatic biological community in the vicinity of the power plant. A useful analogy is the classic problem in communication theory of detecting signal (power plant effect) from noise (variability due to non-power-plant-related causes). All characteristics of the health of aquatic ecosystems, such as species counts, biomass, or such multivariate measures as diversity and abundance, are subject to a number of sources of variability or noise. Included are:

1. Spatial variability (station-to-station). This is a particularly significant source of variability for communities and species that exhibit strong clustering tendencies. The situation is further complicated for mobile species, where spatial and temporal variability may be large, even for small time scales.

2. Depth variability. Depth variability has an origin similar to spatial variability and can be important for all but benthic organisms.

3. External environmental effects (variable predation, climatic effects, hydrodynamic variability, etc.).

4. Measurement and laboratory analytical bias and error.

5. Seasonality.

Variability is usually parameterized by the variance, or second central moment of the probability distribution of the variable of interest. In the simplest case where all effects are independent, the variance is additive. Therefore, for a typical monitoring design that allocates samples to multiple sites, depths, and seasons, as well as before and after power plant operation, the variance of the mean (counts or biomass) of a particular species might be approximated conceptually as

$$\sigma^2 = \sigma_A^2 + \sigma_B^2 + \sigma_C^2 + \sigma_D^2 + \sigma_E^2 + K \cdot E^2 \tag{3.1}$$

where E is the level of change attributable to operation of the power plant and K is a constant undefined at present. If the variance attributable to non-plant effects is denoted as

$$\sigma_*^2 = \sigma^2 - KE^2$$

the ability of a monitoring network to detect plant induced change can be related to the ratio E/σ_*. For $E/\sigma_* \gg 1$, the plant effect will be relatively easily detectable using any reasonable sampling design. For $E/\sigma_* \ll 1$, it will be very difficult to detect the effect, even with high levels of sampling effort in both space and time. Therefore, the ratio E/σ_* is the primary determinant of detectability of change; although sampling effort (represented as n, the product of number of stations, number of depths, sampling frequency, and length of sampling program) is important, it has a lesser effect than the E/σ_* ratio. This is so because the power (probability of detecting a change of level E, discussed in Chapter 2) is a nonlinear function of a parameter of the form $kE\sqrt{n}/\sigma_*$. If this ratio is used as an (dimensionless) effectiveness ratio, effectiveness is seen to increase with σ_*^{-1}, but only with the square root of the level of sampling effort. Clearly, then, attempts to reduce the variability can be very successful in increasing sampling effectiveness. This is the motivation for stratified sampling designs that reduce variability attributable to depths, stations, and seasonal variation. Seasonal effects are, however, a complicated source of variability, and it is our belief that improper modeling of multi-year seasonality invalidates many existing monitoring programs and casts suspicion on conclusions drawn from much of the data that has been collected.

Thus far, nothing has been said as to how sampling effort (n) might be allocated to stations and depths (spatial designs) as opposed to sampling frequency (temporal designs). Such questions lie in the realm of statistical design, discussed subsequently in the cost-effectiveness section of this chapter. It should be apparent, however, that a major consideration in the time-space trade-off is the ability to reduce variability due to sources 1, 2, and 5. The general comments made here apply only to changes that affect the mean of the ecological indicator of interest; in some cases the operation of a power plant may affect the variability attributable to one or more of the sources 1-5 above. In such cases, the detectability of change will take a somewhat different form than that suggested above. However, in the simplest case, and that which almost all existing monitoring networks are designed to detect, the primary concerns of monitoring design must be (1) to reduce underlying variability, σ_*, to the greatest extent possible so as to increase the E/σ_* ratio; and (2) to allocate samples most effectively in space and time so as to achieve the highest effective sample size, n. The following section reviews monitoring design methods that are directed toward this end.

Statistical Methods

In assessing the applicability of statistical methods to any particular power plant monitoring situation, it is necessary to consider the wide range of plant settings and aquatic community types that may be impacted. For this purpose, a two-way classification scheme of species type and plant location is useful. The scheme proposed recognizes three species types: fish, plankton, and benthos; and four physical setting types: streams/rivers, cooling lakes/reservoirs/lakes, large lake/estuaries, and coastal environments. The suggested classification matrix is shown in Figure 3.1. It recognizes that the methods used to assess changes in relatively immobile benthic organisms, for example, will be different from those used to detect changes in fish, which are highly mobile. Likewise, sampling designs may vary greatly between lakes or reservoirs, which are relatively closed systems, and between coastal settings or large lakes (e.g., the Great Lakes), where it is difficult or impossible to isolate the population impacted. In the remainder of this chapter we will use the site/species classification matrix to identify the appropriateness of various statistical methods.

In Chapter 2 some statistical tests have been mentioned in the context of general monitoring design issues. In this section, various statistical methods are reviewed that either have been used widely in past power plant assessment studies, or that we feel may have application to one or more of the elements in the matrix of Figure 3.1. The methods considered fall into six general classes: (1) classical tests, (2) nonparametric tests, (3) multivariate methods, (4) time series analysis, (5) state estimation, and (6) kriging.

Classical Tests

As noted in the preceding section, classical statistical tests for change make certain assumptions about the probability distribution of the data and the form that (power-plant induced) changes may take. So long as these assumptions are met, classical tests are usually the most efficient method of testing for change. For instance, the analysis of variance assumes that the data are independent, normally

SITE TYPE

SPECIES TYPE

	stream/ river	cooling lake/ reservoir/ lake	large lake/ estuary	coastal
fish				
plankton				
benthos				

Figure 3.1 Species/Site Matrix for Mehods Classification

distributed, and homoskedastic. The importance of the first assumption depends on the amount of correlation present between observations; however, its effect usually can be minimized by careful experimental design such as avoiding collection of replicates at very short time intervals, or with inadequate spatial separation. Except for some distributions with very heavy tails, the second assumption usually is not critical since the test deals with sample means, which by the Central Limit Theorem tend to have a normal distribution regardless of the distribution of the data. The third assumption is the most critical, and likely to be violated. Transformations of the raw data, e.g.,logarithmic, are often employed to minimize this problem. In any event, the reliance that can be placed on the results obtained from classical statistical tests is directly dependent on how well the data conform to the most critical assumptions of the test. Unfortunately, this conformance is often difficult to verify, since the sample sizes usually involved do not provide a basis for conducting tests of, for instance, homoskedasticity or independence with reasonable power.

Classical tests are applicable to all elements of the site/-species matrix, although the particular tests used will vary.

Nonparametric Tests

Nonparametric tests attempt to relax some of the assumptions inherent in classical tests, and in so doing offer improved robustness against a range of underlying probability distributions. Therefore, while for any particular distribution there is usually a classical, or parametric, alternative that is more efficient than its nonparametric counterpart, the nonparametric test will often perform much better when the assumptions of the parametric test cannot be met. Lettenmaier (1975) has employed Monte Carlo methods to define the power of nonparametric tests.

Two widely used classes of nonparametric tests are based on permutation and rank statistics. Permutation tests are applicable when the data can be assigned to some independent ordinal scale, such as time of collection or distance from a waste discharge outfall. If this is the case, permutations of the ordinate, under the null hypothesis, will be equally likely, and this will form the basis for computation of critical values of the test statistic. The other popular class of nonparametric tests is based on ranks. In such tests the raw values of the data are replaced by their ranks in a pooled sample. Under some fairly general assumptions, the probability levels, hence test statistics, are independent of the particular form of the probability distribution of the data. Bell et al. (1981) summarized a number of nonparametric tests applicable to monitoring of benthic species data.

In many cases, there is a close analogy between a parametric and a nonparametric test. For example, the anology to the t-test is Mann-Whitney's test, which is based on the sum of the ranks of the data from one of the partitions (e.g., before plant operation) in the pooled data. The Kruskal-Wallis test, and Friedman's tests (Conover, 1971) are analogous to the analysis of variance in certain situations.

Although nonparametric tests are generally more robust than parametric ones, they are not universally advocated. Green (1979) for instance, claims that the assumptions of many parametric tests, such as the analysis of variance, are not too restrictive if carefully applied, and that the increased complexity of non-parametric tests do not justify their use. A related criticism is that the results of nonparametric tests are often not as quantitative as those of their parametric counterparts; for instance, Mann-Whitney's test will identify the direction (increasing or decreasing) of a change, but not its magnitude. As for classical tests, nonparametric tests are applicable to all elements of the species/site matrix, although the particular tests will vary.

Multivariate Statistical Tests

In many cases the objective of a monitoring network will be to detect changes in communities, rather than in individual species, requiring tests that synthesize information about many species. This is an area that has generally received less attention than univariate (parametric or nonparametric) tests; however, some useful work has been done recently in this area. Van Belle and Fisher (1977) proposed a nonparametric test, based on ranks, that is applicable to a family of multivariate indices such as diversity measures. The test statistic weights each index by the distance from a pollution source such as a sewage outfall or thermal discharge. The critical values of the test statistic are then computed under the assumption that the distances are random.

Word (1978) has developed an infaunal trophic index for the Palos Verdes region of the southern California coast. The index appears to be quite stable, i.e., is minimally affected by sampling variability. There is not, at present, a rigorous statistical basis for testing for change in the index, but it appears that a statistical framework could be established.

Engen (1978) has written a monograph discussing the statistical basis for a family of species-abundance models, including well known diversity indices as well as some more complex forms. Although the work is not directed toward the assessment of change per se, it does form the basis for conducting tests of some of the models proposed.

Multivariate methods have been most widely used in benthic assessments, primarily at coastal and estuarine sites, although this is probably the result of lesser concern over benthic impacts at the other site types rather than inapplicability of the techniques. Some limited application has been made to plankton. We are unaware of any such use in fish assessment.

Time Series Analysis

Time series analysis is a separate branch of statistics, which treats time as the sole independent variable. In general, a time series consists of a sequence of observations, $\{X_t, t = 1, \ldots n\}$ as a process sampled at equal time intervals. The models used are of the form:

$$X_t = f_1\ (X_{t-1},\ X_{t-z} \ldots X_{t-p}) + f_2(\varepsilon_{t-1},\ \varepsilon_{t-2} \ldots \varepsilon_{t-q}) + \varepsilon_t + N_t \tag{3.2}$$

where ε_t is the residual noise and is independent of the observations at times up to and including t. N_t is a deterministic mean function which may account for changes in X_t caused by power plant operation.

The most widely used models (e.g., Box and Jenkins, 1970) treat $f_1(\cdot)$ and $f_2(\cdot)$ as linear functions and use small values (usually less than 2) for p and q. These models account for seasonality by a process of seasonal differencing. The form of the functions $f_1(\cdot)$ and $f_2(\cdot)$ is determined by the persistence structure of the sequence, which is a function of the time frequency of sampling. This explicit treatment of persistence, or dependence between observations, is in marked contrast to the widely used classical tests, which all assume independence.

The primary advantage of the time series models is that time is an explicit variable, thus year-to-year and season-to-season variability are accounted for and are not confounded with the plant effect, N_t. The disadvantages are that relatively long record lengths are required to allow parameter estimates sufficiently accurate to yield high power against change, and multiple stations present some problems. Missing observations and irregularly collected data have presented major problems in the past, but some recent work (e.g., Lettenmaier, 1980; Toyama and Veneziano, 1980) has reduced these complications.

Lettenmaier et al. (1978) computed power functions for time series models applicable to assessment of power plant changes. This work indicates that rather long series of observations, e.g., on the order of 10 years of data, might be necessary to achieve reasonable power against changes on the order of $E/\sigma_* \simeq 1.0$. This presumes the existence of only a single station; to our knowledge no work has been done to estimate time/space trade-offs for reducing σ_*. It is clear, however, that the level of sampling effort at any sampling date required to support time series models would be much lower than is presently the case.

There have been very few applications of time series analysis to assessment of power plant impacts, primarily due to the short record lengths available and to the failure of existing power plants to obtain consistent time sequences of observations suitable for time series analysis. Notable exceptions are a study of exposure panel data at Millstone Point (Brown and Moore, 1976) and an analysis of phytoplankton data at Zion (Murarka et al., 1976). In the case of the Zion data, it is worth noting that McKenzie et al. (1977) found a significant plant effect using the analysis of variance, while Murarka et al. (1977) found no effect using time series analysis. The probable cause for this is the inability of the analysis of variance to characterize temporal variability properly, which can result in incorrect identification of plant effects that are really only manifestations of natural seasonal and annual variability.

Time series analysis is potentially applicable to all elements of the site/species matrix, although its application to fish monitoring presents some peculiar problems, such as many seasons with zero values for migratory species.

State Estimation

In a simplified sense, state estimation may be viewed as a generalized form of time series analysis, which replaces equation 3-2 with a multivariate formulation. The functions f_1 and f_2 may be driven by external variables, as well as the noise vector and vector of previous observations. Typically, for aquatic systems, the external variables might include light, temperature, rainfall, streamflow, etc. This modified form of equation 3-2 is augmented with a measurement equation, which allows for measurement or sampling error. The predictions of the state (e.g., the aquatic indicator) of the system made from the measurement and dynamical equations are then combined using an algorithm that weights both on the basis of the relative amount of measurement and on the modeling error.

The principal advantage of this formulation of a time-dependent model is that the model structure need not be wholly estimated from the data, as in time series analysis. The general form may be imposed externally. A secondary advantage is that there is no requirement for equal sample spacing in time. State estimation models are most appropriate where some knowledge, in the form of a model (preferably in differential form), exists as to the dynamics of a particular variable. A good example is phytoplankton, for which there are many models that approximate the general form of the dynamics, but cannot account for all factors affecting phytoplankton growth. Rather than using a simplistic form that models phytoplankton biomass, for instance, only as a function of the previous observation(s), state estimation allows the driving functions (light and temperature) to be included, thus reducing the magnitude of the residual noise.

An additional advantage of state estimation is that the variances of the predictions are also predicted and are independent of the observations. Therefore, it is possible to optimize the allocation of samples *a priori* so as to result in the lowest prediction variance. It is also possible to formulate the model so that the parameters can be estimated adaptively, i.e., each new observation would be used to improve the parameter estimates. If the model is formulated in such a way that the parameters themselves reflect the change that might be induced by the operation of a power plant, a test for change can be so constructed.

The only attempt of which we are aware to apply state estimation to power plant assessment is the work of Schrader and Moore (1977), which assessed the effect of a thermal discharge on Massachussetts Bay. To our knowledge, no application to ecological monitoring has been made. Applicability of the technique would be highest for plankton, for which the most highly developed dynamical models are available.

Kriging

Kriging is a technique developed by French mining geologists to estimate the areal extent of ore deposits. Considerable theoretical work in the area in the last ten years has increased the flexibility of the method to the point that it is presently a useful technique for estimating areal precipitation for storm events, groundwater head contours, groundwater quality, and other processes for which point measurements must be extrapolated over an area. Its use in aquatic monitoring was explored by Hughes and Lettenmaier (1980). It has never been used in power plant monitoring studies to our knowledge; however, it may be applicable to such studies in conjunction with time series analysis. In this case, pointwise estimates from a number of stations would be aggregated at each time, using kriging, to form a single time series of, for instance, total biomass of an ecological indicator. Alternatively, two time series corresponding to affected and control areas might be so formed.

The primary difficulty with kriging, as illustrated by Hughes and Lettenmaier (1981), is that it requires relatively large sample sizes for parameter estimation. It remains an open question, however, as to whether intensive pilot program data could be used to estimate model parameters, which might then be applied to subsequent, more limited data sets to estimate the time series in the manner suggested above. Another potentially useful application of kriging in power plant monitoring is mapping of the extent of thermal plumes, which must be known to classify stations as either affected or control.

Kriging is applicable only to variables that are not rapidly changing in time, i.e., so that the time scale of changes is larger than the time to complete a sampling survey of all stations. Therefore, it appears to have little potential for application to fisheries estimation. It may, however, be useful for plankton and benthic estimation under the limitations discussed above.

Strategic Implications

By far the most common statistical method for assessment of power plant related changes in the aquatic environment is the analysis of variance. McKenzie et al. (1977), in their review of monitoring programs required under the Nuclear Regulatory Commission's Environmental Technical Specifications, claim that "this approach is probably the best available methodology...."

Although its use is widespread, there are some serious problems with the kinds of factorial designs to which the analysis of variance is most suited. In the most straightforward design, where each of the sources of variability discussed earlier is a factor, the major problem is that status (preoperational vs. operational) is not a reasonable surrogate for plant effect. That is, although the application of the technique may be procedurally correct, the mere existence of a status effect is not sufficient to allow one to infer that the plant is responsible.

Recognizing this, McCaughran (1977a) proposed use of the interaction effects in a two-way factorial design (status by site) where sites may be classifed as affected (e.g., within the plume) and control. The test, in essence, is whether the difference between status for the control stations is different from that in the affected area. The idea is that by considering the interactions, rather than the status effect directly, year-to-year (and perhaps season-to-season) variability can be reduced or eliminated.

In addition to the questionable implied assumption that control-affected station differences or ratios (differences in transformed, logarithmic space) will be constant in the absence of a plant effect, McKenzie et al. (1977) identified three other weaknesses with this approach:

1. Interaction effects are distorted by transformation of the data. Although the analysis of variance is relatively robust to violation of the normality assumption, it is less robust for small sample sizes and highly non-normal distributions. Many biological

indicators are in fact highly non-normal and so are usually transformed via the logarithmic or square root transformations prior to testing. The inability to make such transformations may result in problems in interpreting the results of the analysis of variance tests.

2. If the effects of station and site are not additive in the absence of a plant effect, a significant interaction effect may be present that is not attributable to plant operaiton. It is very difficult to determine whether such nonadditivity, which may result from correlation in the data or higher order dependency, is present.

3. Assessment of interaction effects in multifactor designs becomes very difficult because the higher order interactions are confounded with the site-station effects. The alternative is to limit the analysis to two-way tests; however, the failure to stratify increases the residual variance, which reduces the detectability of change.

An alternate approach, suggested by McKenzie et al. (1977), is to select control-affected station pairs, then to analyze the station differences in a factorial design. Thus, the site information is extracted by differencing a priori, and the purpose of the analysis of variance is to determine whether the differences show a change with status, as well as with other factors such as depth, season, etc. Unfortunately, the problem of identifying suitable control-affected station pairs may be a serious limitation to this approach. For instance, at Southern California Edison's San Onofre generating station, the discharge area is characterized by a rocky substrate, while the nearby areas otherwise suitable as controls all have sandy substrates. An additional problem relates to the original issue of seasonal and annual variability and to the implicit assumption that such effects can be accounted for by differencing, either in untransformed or logarithmic space. This assumption amounts to a statement that the natural variability in the differences or ratios from year to year is small enough so that a plant effect will not be observed.

Although it should be possible to conduct experiments to determine whether this is a reasonable assumption, to our knowledge, no such attempts have been made.

Based on these difficulties with existing methods, our opinion is that present designs cannot conclusively attribute environmental change to the operation of power plants, particularly where the effects are modest as compared to natural variability. More attention must be paid to the establishment of long-term sampling programs; in general, the existing short-term, intensive programs do not appear to be cost-effective. In the following section, some suggestions are made for establishment of monitoring strategies that place more emphasis on characterization of temporal variability.

COST-EFFECTIVENESS

As described previously, the process of biological monitoring is both complex and multidimensional. This section addresses the general concept of cost-effectiveness as it applies to the design and operation of biological monitoring systems. In its most generic form, cost-effectiveness can be defined as the efficient use of economic resources to obtain a goal or objective. In the context of biological monitoring, the concept of cost-effectiveness implies one of the two following approaches:

1. For a given budget, sample to maximize the amount of information obtained.
2. For a given level of information, sample to minimize the cost of obtaining that information.

Information, as interpreted here, can assume several forms: the descriptions of variables, a statistical characterization of that variable, expert knowledge, or added insight into the performance of a statistical test. Thus, cost-effectiveness implies a trade-off of two conflicting objectives: first, that of obtaining information that helps characterize the biological system under investigation, and second, minimizing the economic resources required to acquire that information.

Biological Monitoring Design Options

A variety of factors impact the determination of a cost-effective biological network. The potential purposes of the network under consideration here are assumed to be the detection of change in the biological or chemical environment as measured by:

1. Temporal change: either the existence of long-term trends (or shifts), short-term (episodic changes), or both;
2. Spatial change: changes between areas impacted by a discharge and those areas not impacted; and
3. Time/space factorial designs, as described in the statistical aspects section.

From a cost-effective standpoint, several fundamental decisions are faced by the designer of the network. Of primary importance are:

1. The type of network (synoptic or baseline) most appropriate for the goal of the network;
2. Location of the stations comprising the network and whether they should be fixed or vary with time;
3. The type of biological and chemical samples that should be taken and whether there are any samples that would supply more information than others;
4. The level of confidence to be accorded the information to be gathered. This confidence level will affect the number of samples taken and the number of replicates taken for each sample; and
5. The definition of criteria by which (1)-(4) can be evaluated in terms of total system cost.

Cost of Sampling

An adequate analysis of cost-effective biological monitoring requires an accurate estimate of the cost of sampling. Unfortunately, the cost of sampling is highly variable and often site-specific. In an attempt to develop general or average costs, the components of the cost of sampling can be divided into:

1. Salaries of those involved directly with sampling and their supervisors;
2. Travel expenses to and from the sampling site;
3. Depreciation of the sampling equipment purchase ;
4. Rental of equipment not purchased;
5. Laboratory costs for sample analysis; and
6. Data processing and computer costs.

During Phase I of this research attempts have been made to gather cost information concerning biological and chemical analyses. Three avenues have been explored: obtaining cost data from universities, from power utilities and from private consultants. Greatest success has been achieved in obtaining data from power utilities and from universities. Preliminary contacts have indicated that private consultants view these data as proprietary and are hesitant to release the information without reimbursement.

From preliminary analyses of cost data received from universities and power utilities, it appears that cost data are often not maintained in a form that allows the easy calculation of sampling cost per sample. That is, although cost data do exist, dividing the components of costs into the six categories suggested previously (1-6) would be difficult at best. It appears that considerable effort will be required to supplement the current base of sampling cost, and this effort will be accomplished as a task of Phase II of this research, where the goal will be to determine as accurately as possible the primary components of the costs associated with each type of biological and chemical sample taken.

Models of Cost-Effectiveness

Several mathematical models have been developed in the literature that address, at least in part, the concept of cost-effective monitoring. Four of these models are reviewed in this section and the limitations of each are discussed. Although varying in their level of complexity, each of the models does share some important characteristics. Each model requires as input the estimated cost of sampling for each species or parameter that is to be monitored. Each model assumes that the number of locations at which samples are to be taken is either fixed or is bounded by an upper value that has been determined previously. Another assumption made by each of the models is that the data to be analyzed are normally distributed (or log-normally distributed). The validity of this assumption has not been tested in any of the studies, and sensitivity analysis has not been conducted to determine the importance of this assumption to the results. Most importantly, the models assume that the basic monitoring strategies available for designing networks can be characterized as:

1. The number and location of the sampling stations;
2. The frequency at which the station is sampled;
3. The number of replicates that are taken at a station.

Confidence Intervals of Means

Perhaps the simplest models to be published concerning cost-effectiveness of sampling networks are those that indicate the number of samples required to place confidence intervals around the mean of some biological parameter. Although these models are not actually cost-effectiveness models per se, they do serve to illustrate the fundamental ideas of efficient allocation of resources and are the basis of the more complex models that follow.

The purpose of the confidence interval models is to indicate the smallest number of samples required to make statistically meaningful statements about the mean of distribution. The procedure begins by establishing the confidence level to be applied to the estimate of the mean. For instance, if a 95 percent confidence level is sought, this implies that the test will determine a range of values that would, in ninety-five cases out of one hundred, contain the mean of the distribution in question. The statistical test used for this procedure is known as the Student t-test, included in most elementary statistical texts.

The width of the confidence interval, as noted in the previous section, is of the form $K \cdot \sigma_{*}/\sqrt{n}$. If the mean and variance of the distribution are known (this is typically taken to be the mean and variance of the sampled population), the number of samples (n) necessary to reach the desired confidence level can be determined directly. This test must be modified slightly to test trends or changes in the parameters. A surrogate for minimizing cost (i.e., minimizing the number of samples) can be calculated for a given level of information desired.

Without changing the basic structure of the model, this concept can be applied to sampling more than one parameter in the following way: Given a fixed amount of funds available for monitoring, an objective might be to characterize each of the parameters of interest with the same level of precision. By observing the variance of the data taken for each of the samples, one could, through a trial and error approach, determine sampling frequencies for each parameter to equalize the level of precision with which each is characterized.

Although this procedure does not directly address either the existence of trends or costs, it does illustrate the trade-offs that exist between cost (the number of samples taken) and the confidence one can place on the estimate of the sample mean. Again, it should be emphasized that the statistical test used does assume that the data have certain distributional characteristics. However, for large sample sizes, the Central Limit Theorem, which states that the probability distribution of the sum of n random variables (such as the mean) becomes normal as sample sizes become large, reduces the impact of these assumptions.

Lagrange Multipliers

Duffy et al. (1981) have presented a direct and useful approach to cost-effectiveness. Although a full description of the technique is too lengthy to reproduce here, the procedure will be summarized. These authors suggest two types of change that are important: long-term population change and spatial population change. The goal of monitoring in the context of these types of change is to determine whether or not one would accept a hypothesis that a population did not change significantly based on data collected.

Suppose that one wishes to measure the long-term change in fish population by examining the average annual catch. A mathematical expression for this change could be:

$$y_{i,j,k} = \log((\text{catch})_{i,j,k} + 1) - \log((\text{catch})_{i,j,k}' + 1$$

where $y_{i,j,k}$ is the long-term effect measure for station i, sampling day j, and sampling year k. (Note that the value of one is added to the catch to prevent the inevitable problems that occur when the catch is zero and the log is undefined). What is desired is the determination of whether one can accept the hypothesis that the long-term change is zero on the basis of a set of sampling data.

The approach taken by Duffy et al. (1981) is to apply the Student t-test to this situation in a slightly different form than described in the previous section. Several variables must be defined:

$\bar{y}$ = mean of long-term population change averaged over station and sampling days

$s_{\bar{y}}$ = standard error of the mean estimate

t_p = t statistic at the p confidence level

Based on the t-test the following relationship exists:

$$t_p \leq \bar{y}/s_{\bar{y}}$$

That is, for a two-sided t-test, the t statistic must be less than the mean estimate divided by the mean standard error in order for the null hypothesis of no change to be accepted. From this equation, the minimum amount of change, y_{min}, that can be detected can also be determined:

$$y_{min} = t_p \cdot s_y$$

If two different years are compared, year k and year k+1, the ratio of the catch can be written as:

$$y_k = \ln \left[\frac{(\text{catch})_k}{(\text{catch})_{k+1}}\right]$$

This implies that the smallest detectable average catch ratio (D) that can be calculated for an annual average is:

$$D = \exp (t_p \cdot s_y)$$

From the previous equation, it is obvious that the minimum level of detection is a function of the standard error of the estimate of the mean (s^2_y = variance of the mean number of samples taken). The standard error, however, can be thought to consist of the variance due to spatial variance, temporal variance, and residual variance or:

$$s^2_{\bar{y}} = \frac{s^2_t}{n_t} + \frac{s^2_s}{n_s} + \frac{s^2_e}{n_r n_s n_t}$$

where

s^2_t = estimated variance component of the time effect
s^2_s = estimated variance component of the station effect
s^2_e = estimated residual variance
n_t = number of sampling days
n_s = number of sampling stations
n_r = number of replicate samples

One goal of the design of the network would be to design it in such a way so as to minimize the sum of the variances and thus maximize the information obtained. This goal must, however, be traded-off against the goal of minimizing costs. Assume that monitoring costs can be approximated by the equation:

$$C = n_r \cdot n_s \cdot n_t \cdot C_r + n_t \cdot C_t$$

where C is the total cost

C_r = cost of processing one sample
C_t = overhead of one day of sampling

All of the components needed to determine the optimal combination of costs, sampling days (i.e., frequency), number of stations, and the number of replicates to be taken are now provided. As mentioned previously, this problem can be addressed by either minimizing the cost for a given level of confidence or maximizing the confidence for a given level of financial commitment. Assuming that the problem is to minimize cost, it can be posed in the classical Lagrangian form:

$$L = n_r n_t n_s \cdot C_r + n_t \cdot C_t - \lambda [s_{\bar{y}}^2 + s_{e/n_r n_s n_t}^2 + s_{t/n_t}^2 + s_{s/n_s}^2].$$

Duffy et al. (1981) solves the problem in the classic way by taking the partial derivative of the equations with respect to the variables of interest, setting them equal to zero and solving the set of simultaneous equations that result.

In summary, what this procedure does is to determine the number of stations, the sampling frequency, and the number of replicates that should be taken to minimize the cost of a monitoring system given a specified level of confidence. What is assumed by the model is that the t-test is a valid test for the parameter of interest. Of equal importance is the assumption that the variance due to time, station and residual can be calculated accurately. The model does not supply information concerning how costs should be allocated between different types of species or parameters.

Dynamic Programming

A procedure that is very similar to that just discussed but that uses a different optimization technique to determine the best allocation of resources was presented by Saila et al. (1976). The basic approach involves establishing a cost function in the same way as suggested by Duffy et al. (1981) and minimizing the cost of determining the desired statistics at a given confidence level. Dynamic programming rather than Lagrange multipliers is used to perform the constrained optimization. The procedure requires dividing the problem into 'stages' at which resources are allocated. There is nothing inherently superior about the approach suggested by Saila et al. (1976) versus that of Duffy et al., (1981), but the results raise a point that is valid for both the Lagrangian approach and the dynamic programming approach. For either procedure, it is possible that the value of the decision variances may prove to be non-integer. (That is, the results may suggest a nonsensical answer that implies that 3-1/4 station be used or that 5-1/2 replicates be taken.) This does not suggest that the procedure is not useful, only that care must be taken in interpreting the results. Alternatively, integer programming or mixed integer programming may prove a superior approach to those used in the studies cited.

Linear Programming

A third technique suggested to solve this problem is presented in Loftis and Ward (1980). The procedure suggests the use of linear programming to solve for the best design. Unfortunately, the unique nature of this solution technique (the requirement that the objective function and the constraints be written as linear functions of the decision variables of the problem) makes linear programming an unlikely candidate. To overcome these difficulties, Loftis and Ward made several simplifying assumptions that did considerable damage to the validity of their model formulation. Although these authors were unsuccessful in arriving at a meaningful formulation, it is still possible that linear programming techniques can be applied to some form of the sample allocation problem.

Summary of Cost-Effectiveness Literature

From the literature reviewed, it is clear that efforts have been made to develop a framework for cost-effective monitoring of biological data. These efforts have fallen short in three areas:

1. The tests developed assume the application of the Student t-test, and little effort has been taken to justify its use. Also, the cost-effectiveness approaches have all considered only one type of error, Type 1 error. The formulations do not address the power of the tests (the Type 2 error). As was pointed out in Chapter 2, the power

of the statistical test is as important as the efficiency of the Type 1 detection capability.

2. Although cost data are a requirement of all of the models, little effort seems to have been devoted to the development of an accurate cost data base that would allow cost-effectiveness analysis in a consistent manner. Cost data currently seem so fragmented that efforts to summarize them will continue in Phase II.

3. The models reviewed do not have the capability of distinguishing the value of information to the user. That is, information is not weighted according to how useful it is to the decision process, but rather information is viewed of as having equal value.

Recommended Monitoring Design Approaches

Existing power plant monitoring programs are heavily biased towards spatially intensive allocation of sampling effort, with little attention to establishment of data records that would lend themselves to characterization of temporal variability or the time history of power plant-related ecological changes. Thus, existing impact detection strategies are oriented toward the identification of effects that will be manifested through examination of spatial, rather than temporal cross-sections of the biological processes of interest. It is our belief that these existing monitoring strategies represent an excessive allocation of resources to short-term sampling. As noted earlier, when the impact to standard deviation ratio, E/σ_*, is large it will be relatively straightforward to identify impacts. In such situations, periodic intensive surveys, at frequencies on the order one-to-two per year, should be sufficient to identify the effects. For more modest impacts, i.e., $E/\sigma_* \ll 1$, it is doubtful that existing monitoring methods will be able to detect change, and the potential for misidentification of changes is high. For such low level, chronic effects, the only reasonable sampling strategy is the collection of long-term data, in the form of time series, which will allow proper characterization of the natural time variability. Assessment techniques for identification of change, if any, from such sequences must properly characterize this variability. It may be useful, in conjunction with the establishment of such a continuing network, to make use of a skeletal spatial network, with paired control/affected stations, if possible, to reduce the variance of the observational sequences.

In such a context, the key questions for monitoring network design include:

1. What should be the relative allocation of resources to the short-term, intensive program (for identification of short-term, catastrophic effects) and the continuous program (for identification of chronic effects)?

2. How many stations should be utilized in the continuing programs, and if more than one station is used, how should the multiple station information be handled (e.g., by differencing of control and affected stations, estimation of areal means, or by modeling of the observations as a multivariate process)?

3. How often should intensive surveys be conducted, and what minimum level of change should they be designed to detect?

4. How frequently should samples be collected in the continuous program?

5. What criteria should be used for discontinuation of both the intensive and continuous sampling programs?

6. How long should the continuous program be operated before a power plant goes on line?

In our Phase II work plan we propose specific methods to approach these issues.

SAMPLING METHODS

Implementation of cost-effective monitoring programs is dependent upon the availability of sampling techniques that are reliable and reasonably accurate in terms of measuring various components of the ecosystem. Technical problems in sampling methodology exist for the quantitative evaluation of certain populations of interest. In addition, spatial and temporal variations in distribution and abundance of organisms can introduce large sources of error. Some of these problem areas and alternative sampling methods for fish, benthic macroinvertebrate, and planktonic populations are discussed in this section.

Fish

Population Estimates, Area/Volume Density Methods

Quantitative estimation of the absolute abundance of fishes may be made if the area or volume sampled per effort is known. The density of fish in a given habitat area or volume is the catch per effort divided by the area or volume sampled (Treschev, 1978). Such methods may be generalized easily by stratifying the sampling by habitat or by area, depth, and/or time; and the catches by species, and/or age or size. This method has been applied to a wide variety of situations and sampling gears (Kjelson, 1977). These are summarized briefly below:

1. Blockage. Here, an area, usually a small stream, tidal channel or creek, or a cove in a lake or reservoir, is blocked by a barrier net. The fish in the closed area are removed by repetitive net sampling or by electrofishing. The removal method (Zippin, 1958) is used to analyze the declining pattern of catch per sampling effort. Destructive sampling with rotenone is also used to collect fish in the closed-off area. Blockage methods are among the most accurate of fish assessment methods. Unfortunately, they can be applied only in very specific situations where the area can be closed to prevent fish from leaving. It takes a long time (hours or days) to obtain a single sample. It is virtually impossible to take replicate samples except in small streams.

2. Trawls. Trawling is a general method of catching fish by using a funnel-shaped net towed from a fishing vessel. Trawls can sample a wide variety of habitats, including surface, midwater, and on and near substrate. Trawls vary in size from small tucker trawls, or frame nets that can be operated from an outboard skiff, to the immense otter trawls used from factory ships in marine demersal fisheries. The use of trawls for quantitative fish assessment is well developed (Alverson and Pereyra, 1969; Grosslein, 1971; Houser and Dunn, 1967; Taylor, 1953; Hoese et al., 1968; Edwards, 1973; Oviatt and Nixon,

1973). The major problem with the use of trawls is uncertainty in the dimension of the net opening. It is very difficult to measure this for large trawls. Uncontrolled effects, such as currents from surface winds and tides, greatly affect the efficiency of the net by altering the dimensions of the opening as well as the sampling depth (Wathne, 1977). Subtle changes in the net rigging, and such vessel characteristics as mass and horsepower also influence the catchability of the net (Alverson and Pereyra, 1969).

3. Miscellaneous fixed sampling area/volume devices. These methods include encircling nets such as purse seine, longhaul seine, and lampara seine; enclosure devices such as drop nets and lift nets; and beach seines that encircle U-shaped nearshore areas. These methods collectively sample very restrictive habitats. Beach seines and enclosure methods sample either very shallow or nearshore areas of limited bottom debris or vegetation. Conversely, encircling methods have minimum depth constraints and require large and expensive vessels from which to operate the gear.

4. Baited hooks and traps. Recently, a sampling method has been developed for baited capturing devices such as hooks and traps (Eggers et al., 1982). The area fished with this type of gear is estimated in an experiment that measures the competitive interaction of these gears at different distances along the ground line. This method can provide assessments of fish populations in habitats that, because of extreme bottom relief or debris, are impossible to sample with conventional assessment methods (Eggers et al., 1978).

5. Acoustic. Acoustic methods involve counting or integrating the echos from targets in a transducer beam field of known volume (Cushing, 1973; Forbes and Nakken, 1972). The advantage of acoustic sampling methods is that once the equipment (including echosounder, transducer and recording, digitizing, and integrating equipment) is acquired, it is very inexpensive to obtain continuous monitoring of large volumes of habitat in short periods of time. However, acoustic methods do have many problems, the most serious of which is their

inability to identify attributes of the targets. Other problems exist:

(a) Echos from fish near the surface, near the bottom, or within schools are not detected.

(b) It is not possible to determine size and species composition of the target without companion net sampling.

(c) Conversion of the integrated signal to biomass or abundance requires estimates of the mean target strength of the community sampled. Target strengths are difficult to measure, and lack of this information can seriously bias derived biomass estimates.

(d) The spectrum of target strengths sensed by the gear must be filtered in order to process the data by computer. High frequency echosounders can resolve small targets (i.e., zooplankton and ichthyoplankton), but have limited range. Low frequency echosounders have great range, but can only distinguish large targets. Different configurations of frequency, depth range, and target-strength windows must be used for assessments of zooplankton, small fish, and large fish.

6. Ichthyoplankton. Ichthyoplankton sampling gear is diverse, and includes low speed nets, high speed nets, pump samplers, and diver-operated gears (Bowles and Merriner, 1978; Smith and Richardson, 1977). Ichthyoplankton net sampling methods have some advantages over trawls in that mouth opening areas are well known and accurate measurements of the volume filtered can easily be obtained by using flow meters. The low speed net is the most generally used method for sampling ichthyoplankton (Bowles and Merriner, 1978; Smith and Richardson, 1977). However, with low speed nets it is difficult to fix the depth from which the sample is taken, particularly if currents or surface drift are strong. These methods are subject to contamination by organisms from near surface strata unless opening and closing devices are used. High speed nets avoid the depth bias, but the high pressures that result from high towing speeds cause some organisms to be extruded through the nets. All ichthyoplankton sampling methods are subject to clogging and subsequent loss of efficiency. Pumping methods are very accurate with respect to volume and habitat sampled, but rapidly swimming organisms readily avoid pump intake velocity fields and are under-represented in the samples.

In general, all area/volume density methods suffer from the following problems (Eggers, 1982). (1) The distribution of fish in the natural environment is highly contagious so that even with perfect knowledge of area/volume fished and gear efficiency parameters one would observe highly variable catches per effort (Cassie, 1962, 1968; Pennington and Grosslein, 1978; Taylor, 1953; Lanarz and Adams, 1980). This problem is more acute for bigger, higher trophic level organisms that operate on a large spatial and temporal scale (relative to the area/volume sampled by the gear used to catch them). (2) For some gears, accurate estimates of the area/volume fished are not available. (3) All area/volume density methods are inefficient (Kjelson and Colby, 1977). Fast swimming fishes can avoid the gear, small organisms slip through the meshes, and clogging inhibits the flow of water through ichthyoplankton nets. These inefficiencies collectively cause routine underestimation of the absolute fish abundance (Eggers, 1982; Kjelson and Colby, 1977).

The most serious flaw in area/volume density methods is that it is virtually impossible to achieve community- and population-level assessments. Each sampling method can sample accurately only a very specific habitat. To achieve a community- or population-level assessment, it is necessary to assess fish densities in all habitats of the system. This requires a variety of sampling gears. If the relative efficiency properties of these gears are unknown, a biased assessment of the fish community results. There are very few examples of fish community assessments based on area/volume density methods. One of those few (an assessment of the Lake Washington fish community, Eggers et al., 1978) used acoustics, vertical and horizontal gill nets, midwater trawl, and baited minnow traps. This study took eight years to complete. Bagenal (1979) attempted a fish community assessment with a variety of net sampling methods in three small freshwater lakes in Finland. The study was later evaluated by comparison to rotenone sampling of one of the lakes following completion of the net sampling program. Bagenal concluded that no fishing gear can provide accurate assessment of fish populations.

Population Estimates, Mark/Recapture Techniques

These techniques are perhaps the only truly reliable means of estimating the absolute abundance of natural populations. The statistical properties of these estimators are well-developed (Cormack, 1968; Seber, 1973). The mark/recapture method involves a comparison of proportions of marked and unmarked animals in sequential samples taken from a population with an unknown number of unmarked individuals, but with a known number of marked individuals. The statistical theory is based on assumptions necessary for the number of marked individuals in a sample to be a Poisson, binomial, or multinomial random variable (for the case of infinite populations or finite populations where sampling is with replacement) or a hypergeometric or multihypergeometric random variable (for the case of a finite population where sampling is without replacement). The theory assumes that animals do not lose their marks, that random mixing of marked and unmarked animals occurs, and that there are no differences in catchability of marked and unmarked animals.

The major objective of mark/recapture studies is to estimate abundance, but with multiple marking schemes it is also possible to estimate mortality, emigration, or immigration rates. If some rough estimate of the population abundance is available, it is possible to determine the number of marked animals as well as the sample sizes in the sequential samples that are required to achieve the desired level of accuracy in the derived estimates (Robson and Regier, 1964). The problem with mark/recapture techniques is that, in general, a sizable fraction of the population must be marked in order to recover enough tags to make accurate estimates. This is prohibitively expensive for large or open populations.

Population Estimates, Fisheries Management Techniques

The applicability of fisheries management techniques to the monitoring of power plant impacts has been reviewed by McKenzie et al. (1978). The two most promising of these are (1) catch removal techniques, and (2) regression techniques that consider the functional relationship between lake or reservoir fish yields and productivity indices for the same water bodies.

1. Catch removal techniques. Fisheries managers have been able to track long-term dynamics of large and complex fish populations and communities (Gulland, 1977). This is because the fishery can be easily and accurately monitored with respect to catch per fishing effort, age, fish size, and species composition. These variables are accurate indices of population characteristics, provided that evolution in the power of the fishing fleet has been monitored and that effort units have been appropriately modified. If fishery catches are sufficient to reduce stock size, which may be suggested by a declining catch per effort over time, then estimates of absolute fish abundance can be made by the Leslie/DeLury method (Leslie and Davis, 1939; DeLury, 1947).

Monitoring of fisheries that are operating on power-plant effluent receiving systems can provide indices of power plant impact. If commercial fisheries are currently operating on the receiving system, then these data are reliable and are readily available at low cost. Data from sports fisheries are often unavailable and, even when available, are less reliable than are data from commercial fisheries. However, suitably designed creel censuses can provide accurate indices of fish population dynamics. The major problem with fisheries monitoring data is that the fishery itself represents a significant impact on the population. It is very difficult to separate fishery-induced changes from changes caused by power plants or other environmental factors.

2. Regression techniques. Regression techniques that estimate fish production, yield, or standing crops in lakes and reservoirs, based on indices for system productivity such as a morphoedaphic index or total dissolved solids, have been developed from studies of fish yields from numerous lakes and reservoirs (Ryder et al., 1974). These techniques may provide a convenient way to relate fish lost through entrainment and impingement to potential fish yields in systems where this information is not directly available. This methodology also may

provide a basis for evaluating fish yields from lakes and reservoirs on which power plants are sited. Anomalous regression models from lakes and reservoirs with power plants would indicate power plant-induced impacts at the fish population level.

Impingement Estimates

Juvenile and adult fishes die when they are impinged upon debris screening devices by the force of water flowing through the screens. Impingement is an engineering problem as well as a biological problem. Impinged fish may reduce the operating efficiency of the power plant and at times may interrupt the flow of cooling water, forcing a shut-down of the plant (Sharma, 1978). There have been great advances in engineering solutions for impingement in the last decade (cf. Section V, Jensen, 1978). These involve often elaborate moving screens that gently collect the fish and automatically return the live fish to the receiving system.

Three issues must be resolved to evaluate impacts of fish impingement. These are (1) estimating densities of fish in the general vicinity of the intake (2) estimating the numbers of fish impinged and (3) inferring the population level effect of the impingment mortality. Items (1) and (2) are particularly important in designing engineering solutions that may involve strategies of plant operation and intake siting.

Methods for estimating the density of fish in the vicinity of an intake have been discussed previously. Estimating the numbers of impinged fish is a straightforward problem of counting the numbers of live, dead and stunned (i.e., those alive but which exhibit some delayed mortality) fish collected on the screens or in the trash bins. The sampling theory for estimating numbers of impinged fish has been derived by Murarka and Bodeau (1977) and Murarka et al. (1978). Stratification is useful to reduce sampling effort during non-critical periods and to increase the sampling effort during critical periods. Subsampling procedures may be necessary when numbers of impinged fish are great. Inferring population level effects is discussed below.

Entrainment Estimates

Mortality of larval and juvenile fish small enough to pass through debris screening devices at power plant intakes results from the cumulative effect of a variety of stresses (including temperature, pressure, mechanical, and chemical changes) that are encountered as they pass through the plant condensers. Comprehensive reviews of this problem are given by Schubel and Marcy (1978), and Jensen (1977, 1978, 1981). The major problem in the study of entrainment of larval and juvenile fish is to estimate the rate of mortality due to the cumulative effect of these stresses. Two approaches have been used to evaluate this mortality. The first approach involves generic studies using small-scale simulators of power plant condenser conditions (Kerr, 1953; Coutant and Kedl, 1975; Ginn et al., 1978; New York Univesity Medical Center, 1979; Cada et al., 1981; Crippen et al., 1978), and pressure (New York University Medical Center, 1975) and temperature-tolerance bioassays (Otto et al,. 1976; Schubel et al., 1976; Hettler and Clements, 1978). These studies have been unable to provide quantitative estimates of entrainment survival. However, they have demonstrated that certain larval and juvenile fish are able to survive short-term stresses of the magnitude encountered in power plant condensers (Jinks et al., 1981). Other simulator studies have shown that certain species that have small and fragile larvae have very high entrainment mortality (Cada et al., 1981).

The second approach involves making site-specific estimates of entrainment mortality by sampling intake and discharge to determine the fraction of organisms that survive entrainment (Barnthouse et al., 1978; Lawler, 1977; Stevens and Finlayson, 1977). The rate of entrainment mortality is taken to be one minus the fraction of live organisms in the discharge sample divided by the fraction of live organisms in the intake sample. The major statistical problem with this method involves bias due to the following (Boreman and Goodyear, 1981; Jinks et al., 1981):

1. Destruction of dead organisms by the plant;

2. Differences between intake and discharge sampling methods with respect to extrusion of live and/or dead organisms, efficiency of capturing live and/or dead organisms, and mortality due to sampling;

3. Delayed mortality;

4. Sampling the same water mass.

Items (1) and (3) cause underestimation of actual entrainment mortality. The direction of the bias resulting from (2) is variable depending on the particular difference in the intake and discharge sampling methods (Boreman and Goodyear, 1981). The bias resulting from (2) is a problem when the water velocities in the intake and discharge are different. Some reduction of these biases can be achieved by careful design of the ichthyoplankton sampling program. The available methods were evaluated by Smith and Richardson (1977), Cada and Hergenrader (1978), Bowles et al. (1978), and Bowles and Merriner (1978). These biases are most acute in the sampling of high velocity offshore discharges and in sampling species/life history stages that are especially sensitive to the stresses imposed by the collection process (Jinks et al., 1981).

Population Level Effects

The nature of power plant impacts on fish communities can be roughly categorized into two types:

1. Plume effects. Fish either avoid plumes if plume temperatures are higher than those they prefer, or alternatively are attracted to plumes when the plume temperature is closer to the preferred temperature than is the ambient temperature.

2. Population level effects. These include the loss of habitat due to plume effects and the collective effects of entrainment and impingement mortality on the long-term dynamics of fish populations in the receiving system.

Detection of plume effects involves comparison of fish distribution in the control and noncontrol areas. Standard statistical methods are used to compare catch per effort in the treatment and control areas. The problems with this method are that fish are highly mobile and that there is a great deal of short-term variation in the abundance of fish in any given habitat due to their behavioral responses to ambient light, temperature, prey availability, etc. Net sampling methods require substantial amounts of time to set, fish, and retrieve the gear as well as process the catch. It is very difficult and expensive either to replicate sampling efforts or to turn over sampling units fast enough to detect these short-term effects. The plume effect, then, is confounded with natural variation. Acoustic methods are the only fish assessment techniques that can sample large enough volumes of habitat in short-time resolution sufficient to control this natural variation (Thomas, 1979).

The detection of population effects involves monitoring the population over time. The methods available for monitoring population abundance include analyzing fishery catch data and conducting system-wide assessments with a suite of area/volume density methods. Most fish species are long-lived, particularly the valuable exploited species. Most first order power plant impacts involve direct mortality to larval and young-of-the-year juveniles due to entrainment and impingement. It may take several years for these effects to manifest themselves in the adult population fully recruited to the fishery or susceptible to a particular sampling method. It is very difficult to monitor this kind of population response in the time frame within which power plants are licensed and constructed.

The second problem in detecting population level effects is that the statistical hypothesis testing framework (see Chapter 2) requires a separate treatment and control population. It is often not possible to define treatment and control populations in a receiving system because fish readily move between these areas. Further, it is unlikely that a "sister" system similar enough to the impacted system to serve as a control exists. An exception can be large riverine systems were areas upstream and downstream from power plants may qualify as sister systems that can be considered for control treatments.

Evaluation of the time series of a population response to a power plant may be the only realistic means for determining impact. Unfortunately, past studies of fish population level responses indicate that fish populations are dynamic and are very responsive to man-induced perturbation (e.g., Great Lakes, Loftus and Regier, 1972; Lake Washington, Eggers et al., 1978; marine fisheries, Gulland, 1977). In view of this dynamism in fish populations, it can be difficult to separate natural variation from anthropogenic effects unrelated to power plants in the emerging time series of a fish population's response.

Benthos (Macroinvertebrates)

The distribution and abundance of benthic (bottom dwelling) macrofauna are determined in part by the physical and chemical characteristics of the substrate. The majority of benthic organisms respond to variations in substrate such as its composition (including organic matter content and particle size distribution), the local current velocity, depth, algal or plant cover, oxygen content (in water and sediment), and temperature regime. Consequently, organisms usually do not distribute themselves uniformly among different substrata. In some environments, organisms are found in clumped or patchy distributions which themselves can be either uniformly or randomly distributed within specific sets of substrata.

As a result of the variability found in both substrate types and population distributions, no single sampling method can be applied in all environments. Sampling locations (distribution in space) and sampling gear are site- (and perhaps investigator-) specific. Quantitative sampling of benthic organisms initially requires an estimate of the amount of area available under the range of physical conditions present at the site(s) being evaluated. Once this has been estimated, sampling locations and appropriate collection devices can be identified. The means by which sites can be identified and the advantages of different collecting tools are reviewed briefly for different water bodies.

The extreme heterogeneity of benthic environments normally requires that sampling sites within a given area be stratified (Gonor and Kemp, 1978). Stratification refers to the division of an area into subareas (strata) that are assumed to be more variable among themselves than are the areas within each strata or than are areas within sets of strata. Strata are defined typically by a dominant physical or biotic feature. For example, elevation or organism discontinuity might be an appropriate division in an intertidal environment, while cobble size and current velocity might be more appropriate in a riverine environment. In lakes and reservoirs, depth may be the most discernible feature for stratification, although substrate composition and plant cover can be important in littoral areas. Regardless of how strata are determined, sample sites (and replications) must be allocated among the different strata. Allocations of sampling effort commonly are based on the proportion of total area found in any given substrata, although this is not necessary; it may be more appropriate to sample some strata more intensively (e.g., strata that house important species) and others less intensively (e.g., strata located beyond the depth at which an organism of interest normally occurs).

The validity of comparisons between control and non-control stations will depend upon the extent to which these stations are located in strata having nearly identical conditions. Locating such strata is often virtually impossible. Other errors may arise from inaccurately estimating stratum area or from assuming uniform variance among populations located in any given stratum (Gonor and Kemp, 1978). A detailed discussion of these errors and some solutions for them are found in Cochran (1977) and will not be reviewed here. It is important to note, however, that lack of comparability in sampling sites from control and non-control stations is one of the largest sources of error inherent in the use of macrobenthic data for impact assessment. In many studies, observed differences in macrofauna could not be attributed to plant operation due to a lack of consistency between sites.

Once strata have been defined, sampling sites within strata are located either on a transect (line) or quadrant (grid) system (Greeson, et al., 1977). All possible positions are located along the transect line or within the grid system on the basis of sample area and practicality (i.e., a transect cannot be divided infinitely). These positions are usually numbered with a two-digit system. A random selection process is then used to locate the position of each site within a given transect or grid and allows the investigator to collect as many replicate samples within a given stratum as has been deemed desirable. A transect can, of course, cross stratum boundaries. In areas in which strata are poorly defined (e.g., lakes and salt marshes), a linear transect on which sites are randomly located (via a numbering system) may be a reasonable substitute for stratified random sampling. Detailed discussions of site determination can be found in Cochran (1977), Swartz (1978), Gonor and Kemp (1978), and Tetra Tech (1981b).

Once sampling sites are located, sampling gear must be chosen. The available devices can be placed into three categories: in situ samplers, surface samplers, and dredge (or core) samplers. In situ samplers are left in place (< 24 hours to > a month) in the desired site to capture or to be colonized by a natural assemblage of organisms. They include such samplers as buried substrate or basket samplers, multiple plate samplers, emergence traps and drift nets. These samplers are used widely in stream and riverine environments and to a lesser extent in littoral areas (Hynes, 1970; Merritt and Cummins, 1978). The advantage of these samplers is that they integrate short, temporal differences at any given site, and they allow a quantitative evaluation of environments that are difficult to sample (e.g., drift in rivers, emerging insects, hyporheal community). Their disadvantage is that the chosen substrates tend to be species-specific and may not be representative of the naturally occurring substrates. Colonization of artificial substrates may not reflect the natural community. Emergence traps may not cover all important strata. Quantification of sample area also is a common problem in the use of these sampling devices. Loss of samplers can also be a problem in heavily populated areas or at exposed (turbulent) sites.

Surface samplers include any sampler that delineates a known area of substrate and allows the investigator to remove (usually manually) all organisms into a collecting net or bag. Subsequently, organisms are separated from debris. The most common of these devices include the Lium sampler, the Surber sampler, or a simple frame that delineates a known area of substrate surface. The Lium and Surber samplers are more useful in estuarine or riverine situations where water is shallow and where current velocity is sufficient to wash organisms into a collection net. The frame is more appropriate for use in areas where organisms are largely stationary (e.g., rocky intertidal zones). The advantage of surface samplers is that they allow for the rapid collection of organisms from a known area. These samplers are especially useful for cobble-bottomed environments or for rocky intertidal zones in which most organisms are fixed in space. Samples from control and non-control sites can be compared as long as substrates are directly comparable and are sampled to a constant depth. However, loss of organisms during collection can be problematic in rapidly flowing waters, and sampling must generally be confined to shallow environments. A further disadvantage of these samplers is their restriction to sampling at only one moment in time. In areas where benthic organisms are extremely active and/or very patchy in their distribution, data collected with surface samplers tend to be very variable. In such situations it may be reasonable to use *in situ* samplers or to locate fixed stations along a transect that can be sampled over short time intervals.

Dredge, grab, or core samplers remove a known volume of substrate and its resident organisms from the test site. Samplers such as these are useful primarily in soft-bottomed environments (e.g., lakes, estuaries, and coastal sediments). They include such devices as the Ekman, Peterson, Ponar, box corer, VanVeen, Smith-McIntyre, and Hess samplers (Greeson et al., 1977; Swartz, 1978). Comparisons of their efficiency indicate that depth-of-grab is the source of most variability (Word, 1976; Swartz, 1978). Swartz (1978) recommended the use of a 0.1 m^2 Smith-McIntyre grab in coastal environments. While most of these samplers are operated from surface vessels, diver operation may be a more reliable means of obtaining a constant bite at some locations. Dredge-type samplers require extensive sorting of organisms from sediment and debris; therefore, sample processing can be very expensive. Nevertheless, dredges and cores are the only quantitative sampling devices available for many soft-bottomed, deep environments.

The utility of all these devices is limited primarily by defining comparable strata in a highly heterogeneous environment. In many instances, natural variability in organism distribution within and among strata is so great that differences due to environmental perturbations are not easily distinguished. The number of samples required to define mean density and sample variance is often so great as to preclude its inclusion in a monitoring program. Of 68 studies reviewed for Chapter II in which macrobenthos were sampled, only 19 studies showed any impact from plant operation. In many instances, investigators observed significant differences in benthic populations and community associations but were unable to attribute these differences to plant operation. Differences were associated commonly with differences in substrate composition or with other physical distinctions between control and non-control sites. In other situations, the lack of significant impact was attributed to the fact that the thermal plume did not contact the bottom, hence no impact due to plant operation would be expected. In spite of the popularity of benthic sampling for pollution studies (Cairns, 1979; Tetra Tech, 1981b), such samples have not always been useful in detecting impacts due to power generation.

Plankton

Spatial/Temporal Considerations

Planktonic populations include all organisms that exist suspended freely in the water column. They include phytoplankton (autotrophic and heterotrophic producers), zooplankton, and ichthyoplankton (consumer) organisms. Such organisms are normally dependent upon water currents and density gradients for transport, although certain taxa are capable of limited motion. As a result of this relative passivity, the uniformity of planktonic distributions is a function of the extent to which a water body is well-mixed. Since complete mixing occurs rarely in all but the most shallow and exposed water bodies, planktonic populations are typically patchy in their distribution through space. Depth stratification also occurs. Phytoplankton respond differentially to depth primarily as a function of light availability, while zooplankton populations may migrate vertically on a diel basis (Vollenweider, 1974; Edmondson and Winberg, 1971). Additional variability in spatial distributions occurs as a result of microspatial differences in nutrient availability, temperature, and predation pressure. Seasonal variations in these and other factors (e.g., tides, winds, thermal stratification, life histories) also affect distribution patterns.

These characteristics normally do not limit the use of plankton data in monitoring studies. Nevertheless, quantitative sampling of these populations must recognize this lack of uniform distribution. Allocation of sampling effort should be made on the basis of hydrographic conditions if such information is available. For example, hydrodynamic characterization of discharge dilution zones can guide the location of sites for characterization of populations located within and outside of the thermal plume. Sampling within and outside of Langmuir circulation cells may be appropriate in large wind-swept water bodies. Appropriate randomization of sites within hydrographic strata should be included in the final sampling design in order to account for microspatial patchiness. Typically, spatial randomization of sites is accomplished through the use of transects or grids. Care should be taken to include stations that are representative of both littoral and pelagic regions. Choice of sampling gear can reduce many of the problems associated with depth stratification and, in some instances, patchiness. The advantages and disadvantages of different gear are discussed below.

Sampling Gear

Water and its resident plankton can be sampled either continuously through the use of pumps and net tows or discretely through the use of depth-integrating or closing chamber water samplers. In general, population densities or other functional attributes (e.g., production rates) of the plankton are calculated for subsamples of larger collected volumes into which organisms are concentrated either through filtration or centrifugation (Tonolli, 1971).

Pump samples are used primarily for phytoplankton, while net tows are used primarily for larger zooplankton. Continuous pump samples are useful if population densities are low and large volumes of water need to be collected at specified depths.

Pump samples can be centrifuged continuously for concentration and later characterized chemically or measured for *in situ* fluorescence. The agitative nature of continuous centrifugation precludes its use for taxonomic or rate determinations. Without centrifugation, pump samples can be used for any characterization. Tows can include anything from a primitive net pulled behind a boat to the sophisticated Clarke-Bumpus plankton sampler, which quantitatively samples a known water volume at a specified depth. In either case, mesh sizes can be varied to sample different sized organisms. Typically, larger zooplankton are sampled with a pore size of 158 m, while smaller zooplankton are collected with a pore size of 76 m. Mesh sizes $<$ 50 m can be used for phytoplankton although morphological asymmetry will still exclude many organisms. Nannoplankton cannot be sampled adequately with towed net samplers. Towed samplers are only useful in large open water bodies that have sufficient depth to accommodate the tow (e.g., large lakes, coastal areas, fjords, etc.) Tow samplers integrate differences due to space and can therefore be useful in extremely heterogeneous environments. Population densities must be low or net clogging can be problematic. Other problems can arise if zooplankton are able to avoid capture by swimming. A complete discussion of these samplers is found in Tranter (1968).

Closing chamber samplers are useful for obtaining discrete samples at specified depths. Such samplers consist of an open-ended bottle (typical volume 1- to 5-l.) that is lowered to a known depth and then closed automatically with some trigger mechanisms ('messenger'). Common bottles include the Van Dorn, Ruttner and Friedinger (Vollenweider, 1974). Both vertical and horizontal samplers can be employed although vertical bottles such as the Van Dorn have limited utility for collecting near the bottom or in very shallow water. Samples are generally transferred to glass or polyethylene bottles for later analysis. Samples must be taken at frequent depth intervals (e.g., every 1 to 3 m.) and at many sites in order to characterize the population adequately. The collected volumes are usually inadequate to characterize zooplankton densities unless organisms are very abundant and easily captured. Closing samplers are used primarily for phytoplankton populations.

Depth-integrating samplers are also used primarily for phytoplankton sampling. These samplers usually consist of some type of open-ended tube that can be lowered through the water column, closed and then raised to the surface. The homogeneous sample thus collected integrates stratification due to depth and can be used to characterize the total density or biomass below a point on the water surface. Such samplers are not useful for rate characterization, since production rates will vary with light and temperature distributions, which are not usually simple linear functions of depth. Nevertheless, such samplers are very useful for biomass and taxonomic determination in shallow- and medium-depth water bodies. Pumps also can be used to collect all the organisms in a given column of water.

A variation of a depth-integrating sampler has been used for zooplankton sampling. This method, the vertical net haul, consists of an open net that is lowered to the desired depth and then raised rapidly to the surface (Tonolli, 1971). Collected organisms can then be washed into a small volume of water and saved for analysis. The method has the advantages of integrating depth differences and concentrating organisms at the same time, although organism escape can be a problem.

SUMMARY

The following points summarize this chapter.

- Hydrodynamic models are constructed from well-accepted principles of conservation of mass, energy and momentum. Fluid motion and temperature can be simulated with these models and the accuracy of the models is dependent on the spatial and temporal detail chosen.

- Aquatic ecosystem models use linear or quasi-linear differential equations to describe the biological processes such as birth, growth, movement, predation, and death. Equations to describe these processes are not well-accepted by all ecologists, thus their application to modeling is limited.

- Aquatic ecosystem models are typically data-intensive, and data bases sufficiently large to calibrate ecosystem models do not exist at most power plants.

- The technology to permit direct communication between computers is evolving rapidly and will be a valuable tool to designers of monitoring networks in the future. However, this is an industry-wide problem and Phase II of this project will not attempt to make a contribution in this area.

- Existing software for the analysis and display of aquatic monitoring data is available and used widely. Most prominent among the software packages are SPSS and SAS.

- A major need of aquatic monitoring designers is an interactive software system that can display monitoring network designs and provide statistical measures of their effectiveness.

- A variety of statistical tests exist for trend detection including classical techniques (such as the analysis of variance), nonparametric tests, multi-variate tests, time series analysis, state estimation techniques and kriging.

- Although the analysis of variance is by far the most commonly used statistical test for power plant-related ecosystem impact, it is hampered by the difficulty of distinguishing long-term change from natural season-to-season and year-to-year variability.

- The present emphasis of most aquatic monitoring programs on short-term, intensive sampling should be altered to support longer term, lower level sampling efforts as well, which are amenable to time series analysis. This will allow the monitoring programs to address problems of long-term, population level impacts, and allow identification of decision points relative to cessation of monitoring efforts.

- An important consideration in the design of monitoring networks is cost-effectiveness. Appropriate cost-effective monitoring designs would either maximize the detectability of change for a fixed cost, or minimize the cost of detecting a fixed level of change.

- Sampling cost data are an essential element of cost-effective monitoring design methods; however, these data are difficult to obtain due to the general lack of records by public agencies and universities that would allow sampling costs to be identified, and due to proprietary concerns of private firms.

- Although mathematical optimization models have been suggested for monitoring network design, these models have focused on confidence levels for identification of means, rather than on detectability of change per se. Further, the value of monitoring data to utilities for planning and operational considerations has not been considered by any of the existing design methods.

- Extreme variability in space and time distributions, lack of adequate control sites, poor understanding of compensation mechanisms, and the expense of a comprehensive sampling program limit the ability of most monitoring programs to define fish population effects in a quantitatively reliable manner. Time series analysis of long-term data sets may be a necessary prerequisite for impact evaluation.

- Spatial heterogeneity in benthic environments and hence in benthic fauna limit the extent to which such samples have been able to show impacts due to power generation.

- Sampling errors do not normally limit the utility of phytoplankton and zooplankton samples for detecting impacts due to power generation.

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