Proceedings From the EPRI Workshop on 316(a) Issues: Technical and Regulatory Considerations

October 16-17, 2003

Technical Report

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PRODUCT DESCRIPTION

This report documents the proceedings of a recent workshop co-organized by American Electric Power (AEP) and EPRI on thermal discharge issues. Thermal discharge issues are receiving increasing attention from government agencies and electric power generating companies. This report examines recent developments and future trends.

Results & Findings

On October 16-17, 2003, at AEP's headquarters, over 120 people met in a workshop cosponsored by EPRI and AEP to exchange new developments in technical, regulatory, and legal information on Section 316(a) of the Clean Water Act (CWA). This section of CWA regulates thermal effluents and provides for a variance from both technology-based limits and water quality standards if the thermal discharge ensures the protection of a balanced, indigenous population of fish and wildlife in and on the water. Key emerging issues include interaction between thermal effluent and other pollutants (for example, nutrients and acid mine drainage); total maximum daily loads (TMDLs); differences in fish responses between lab and field; relevant fish response for assessment purposes; common currency for cross pollutant trading; and acceptance criteria for ecological enhancement.

Challenges & Objective(s)

The proceedings will be valuable to the regulated community, regulators, researchers, consultants, and environmental attorneys. The report will aid in designing research, monitoring, assessment, regulatory and management programs for thermal-electric power plant cooling systems that are both environmentally protective and cost efficient.

Applications, Values & Use

Information generated at the conference, and presented in this report, provides the first major update in the area of 316(a) since a national conference held in 1975. As permit renewal applications for existing plants and designs for new plants are considered, this report provides valuable technical, regulatory, and contact information. Power companies and regulators should be especially aware of new application opportunities that are developing with respect to water quality trading and ecosystem enhancement.

EPRI Perspective

During the last 5-10 years, research in EPRI's 316(a) and (b): Fish Protection Program has been strongly focused on 316(b) issues as a result of the United States Environmental Protection Agency's (USEPA's) effort to develop national 316(b) implementation rules. Now that the rule making has been completed, there is a growing interest among EPRI members in 316(a) research. A major motivation for the workshop was to identify current and emerging issues in thermal discharge management. Information generated by this meeting will be used by EPRI and its advisors to guide future research. A significant outcome of the meeting was acknowledging institutional memory loss: results and implications of much of early 316(a) research has been lost by both regulators and the industry. As a result, EPRI has initiated the creation of a living, web-based, reference document on thermal discharge topics and issues.

Approach

EPRI designed the scope of the meeting to include multiple and diverse perspectives, case studies, recent developments, and future trends. Well-known and highly regarded state agency personnel, researchers, power company employees, consultants, and lawyers were invited to make specified presentations. Notification of the meeting was widely circulated to attract a large, diverse audience. At the meeting, open discussion by all participants was encouraged. Lunches and dinner were arranged to provide additional time for participant interaction.

Keywords

316(a)
Variances
Workshop
Thermal discharges
Environmental impacts
Case studies

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OVERVIEW OF 316(A) WORKSHOP AND FUTURE RESEARCH NEEDS

316(a) Workshop

On October 16-17, 2003 at American Electric Power's (AEP) headquarters in Columbus, OH, over 120 people met, in a workshop cosponsored by the Electric Power Research Institute (EPRI) and AEP, to exchange new developments in technical, regulatory, and legal information on Section 316(a) of the Clean Water Act (CWA). This section of the CWA regulates thermal effluents and provides for a variance from both technology-based limits and water quality standards if it can be demonstrated that the thermal discharge "will assure the protection and propagation of a balanced, indigenous population of shellfish, fish, and wildlife in and on that body of water". The 316(a) program was very active in the 1970s as thermal dischargers conducted studies to determine whether they might qualify for a variance. To disseminate the state of information on thermal ecology at that time, two U.S. Department of Energy (DOE)-sponsored workshops were held, the last in 1975 in Augusta, GA.

Anticipating the need to renew expiring 316(a) variances, to provide guidance in conducting studies for new or changed discharges, and to find out what has changed in the 316(a) arena since 1975, workshop co-chairs Dr. Bob Goldstein of EPRI and Mr. Rob Reash of AEP planned a workshop open to all interested parties. The number of attendees and their broad diversity surpassed expectations. Attending were representatives from electric power companies, governmental agencies, universities, national laboratories and research institutions, as well as environmental attorneys, consultants, and cooling tower manufacturers. A list of attendees, their affiliations, and contact information is shown in Appendix A.

During the course of this two-day workshop, the group heard over 20 presentations on 316(a) and related topics, including legal and regulatory perspectives, fish physiology and behavior, community balance and multimetric indices, thermal modeling, water quality trading, ecosystem enhancement and temporary cooling towers. In contrast to the seventies, there was more emphasis on the thermal discharge response of biological communities versus individual organisms and single populations. This reflects a growing sophistication with respect to ecological science over the intervening three decades. The workshop agenda is shown in Appendix B. All presentations are available by contacting Bob Goldstein at rogoldst@epri.com. In the next few paragraphs, a sampling of the presentation is provided.

Kristy Bulleit of Hunton and Williams, counsel to the Utilities Water Act Group, who reviewed the regulatory/legal history of 316(a) and lessons learned from past studies, gave the opening paper. One of her principal conclusions was that thermal limits tailored to a specific site are most appropriate for minimizing ecological risks.

A number of attendees were involved in the 316(a) program dating back to the 1970s. Chuck Coutant of Oak Ridge National Laboratory (ORNL) was one of the co-authors of the original 316(a) technical guidance completed in 1977. He and Mark Bevelhimer, of ORNL, provided an update on the current state of knowledge of the relationship between water temperature and biological effects and the need to develop tests and data sets to create a better understanding of fish responses to temperature exposures in-situ.

Several data sets generated over the years at power plant sites are multi-decadal in length. Dilip Mathur of Normandeau Associates discussed a continuing data set he began back in the 1960s at the Peach Bottom Atomic Power Station on Conowingo Pond in Pennsylvania, and the value of the data to answer a diverse set of questions about thermal exposure and fish response in the reservoir.

Just over half of the presentations focused on recent site-specific case studies in nine Eastern, Southern, and Midwestern states. John Petro of Exelon discussed the challenges of managing fisheries in cooling lakes in Illinois that were receiving more thermal energy as discharging power plants operated at higher capacity factors. John Balletto of Public Service Enterprise Group (PSEG) presented monitoring and modeling strategies at the Salem and Hudson Generating Stations in New Jersey, and discussed the efficacy of coupling predictive models.

John Veil of Argonne National Laboratory and Todd Petty of the University of West Virginia looked to the future of 316(a) assessments. Mr. Veil discussed environmental enhancements and restoration as part of thermal discharge mitigation strategies. Dr. Petty discussed the need to develop an ecological condition currency for cross contaminant trading. These speakers and others noted that the USEPA's current interest in integrated watershed management creates opportunities to use water quality trading and ecosystem enhancement strategies.

Future Research Needs

The workshop closed with many attendees sharing their perspectives on 316(a) issues and providing insights into future research needs. The research needs identified are:

1. Develop a procedure to set thermal standards when using multiple historical studies with conflicting data.

2. Laboratory and field studies often show differences in terms of thermal effects on aquatic organisms. Identify what the reasons are and develop ways to deal with these differences. Develop tests and data that create a better understanding of fish responses to actual temperature field exposures. There is a need to relate behavioral responses to physiological responses.

3. Develop a guide to sampling study design and sampling protocols that will show how sampling gear, locations and techniques can greatly influence what is caught or sampled, and how to deal with these issues.

4. Conduct a workshop on aspects of Balanced Indigenous Communities (BICs).

5. Perform research in the emerging area of restoration science and power plant impacts.

6. Examine and document situations where combined multiple stressors (such as from thermal effluents, chemicals, metals, and nutrients) are present, and determine what the combined impacts are.

7. Develop a review of recent case studies that demonstrate no appreciable harm (that is, no effects on the BIC), and how these determinations were made.

8. There is a need to infuse more science into the standards setting process. Evaluate state thermal water quality standards in terms of whether they are outdated, the reasons why, and provide suggestions for updating the standards with the best science possible.

9. Evaluate the importance of rare events versus average conditions on impairment. If rare events are important, determine how this can be reflected in water quality standards.

10. Evaluate the vulnerability of electrical utilities to climate change. For example, how would thermal effluents affect the BIC in the presence of higher background temperatures caused by global warming?

11. Evaluate whether 316(a) and 316(b) studies can be better integrated so that solutions address both issues at once better than at present. Determine how would this be done. A workshop to examine these questions would be useful.

12. A number of emerging issues relative to modeling were identified and include: effects of climate change on surface water temperatures and the more frequent occurrence of low flow rates associated with more frequent droughts; use of models in a forecasting mode to predict water temperatures a few days in advance (similar to weather forecasting); use of models to estimate the excess evaporation that may occur from thermal plumes discharged from once-through cooling systems; more realistic near-field turbulence simulations and dissolved gas issues to predict fish impacts. No specific recommendations on how to address these issues were provided.

13. Current evaluations of thermal discharges emphasize community and ecosystem responses. There is a need to develop and test indices to assess the health of biological communities in different types of ecosystems. Indices that work in one region and water body type are not necessarily successful in other regions or water body types, respectively.

14. There is growing interest in integrated watershed management of point and nonpoint sources. The USEPA desires to coordinate NPDES permitting within individual watersheds. The interest in watershed management creates opportunities to use water quality trading and ecosystem enhancement strategies to reduce pollution effects. Research is needed on both of these subjects; e.g., the creation and testing of ecological indices to allow cross pollutant trading.

15. There is a need to develop a framework for doing 316(a) assessments, and a review of tools that could be used in the framework.

16. Review 316(a) case studies (not necessarily to demonstrate "no appreciable harm" to BICs) to determine why they were done, what was done, and the results.

17. Determine how thermal TMDLs increased background water temperatures due to climate change, and threshold temperatures that affect balanced indigenous communities will affect future power plant siting, operations, and the 316(a) program.

Workshop Proceedings

Each of the 21 presenters at the workshop was asked to prepare a paper on their presentation. In total, 17 papers were received, and are contained in this EPRI report. The papers, together with the presentations and recommended future research needs voiced by the workshop attendees, are intended to help to advance both present and future needs of the 316(a) program.

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BEEN THERE, DONE THAT: A LEGAL PERSPECTIVE ON IMPLEMENTATION OF SECTION 316(A) OF THE CLEAN WATER ACT

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Introduction

Since the passage of the Federal Water Pollution Control Act (FWPCA or the Act) Amendments of 1972 (popularly known as the “Clean Water Act”), dischargers of heat have been allowed to seek a variance from otherwise applicable technology and water-quality based limits [1]. Under § 316(a) of the Act, a discharger is entitled to less restrictive permit limits if it can show that such limits will assure protection and propagation of a balanced, indigenous population of shellfish, fish and wildlife in on the body of water to which the discharge is made [2]. Because steam electric power plants discharge heated effluent from their cooling systems, many power plants have sought variances under this provision. This has given both the power industry and the permitting agencies over thirty years of experience implementing § 316(a).

Despite this long history, old questions are resurfacing and new questions arise as dischargers seek renewal of their variances. To resolve these questions efficiently and effectively, it is important to learn from our past experiences, whether those were successes or mistakes. Lawyers are keenly aware of the importance of precedent. When it comes to interpreting and applying § 316(a), it is no less important for regulators and the regulated community to recall the lessons of the past. As the philosopher George Santayana said, “Those who do not remember the past are condemned to repeat it [3].” When implementing § 316(a), we should not give in to the temptation to reinvent the wheel. Instead, we should use available data and information to carry us forward.

In some cases, moving forward may involve collecting new information. As we assess what data to collect and how to evaluate that data, it is important to bear in mind that the collection of even the most extensive empirical data will not lead to perfect certainty. As Santayana also famously said, “Experience seems to most of us to lead to conclusions, but empiricism has sworn never to draw them.” In crafting § 316(a) studies and interpreting the results, it is important to recognize that expert judgment plays an important role in the § 316(a) decision-making process, especially since many of the questions to be addressed by § 316(a) are, to some extent, subjective.

The purpose of this article is to briefly (1) summarize the applicable statutory and regulatory provisions governing § 316(a) variances, (2) discuss some of the intricacies of its application, and (3) identify some important legal issues going forward.

The Roots of Section 316(a)

The regulation of heated discharges dates back many years, to well before the passage of the 1972 Amendments. As early as 1948, the Federal Water Pollution Control Act required establishment of statewide water quality standards for pollutants, including waste heat [4]. The requirement for ambient water quality standards was maintained in the 1965 amendments to the FWPCA [5]. In 1966, the National Technical Advisory Committee (NTAC) convened to review the general scientific literature regarding the effects of pollutants on aquatic life and human health. The NTAC produced the "Green Book," which recommended generic temperature caps (e.g., 90°F) and limits on changes in temperature as compared to background or "natural" temperatures (e.g., 5°F) [6].

In 1971, the newly formed United States Environmental Protection Agency commissioned a more thorough study of water quality criteria. That study resulted in the 1972 "Blue Book," which stressed the need for site specificity [7]. That guidance document was revised, albeit with somewhat less emphasis on site specificity, and republished as the "Red Book" in 1976 [8].

In the FWPCA Amendments of 1972 (P.L. 92-500), Congress recognized that heat is different than other pollutants. Temperature is a natural attribute of all waters. While heat can have both direct and indirect effects on aquatic life, it is not "toxic," persistent, or accumulative. Effects, if any, of heat are transitory and aquatic organisms perceive and, in some cases, will avoid heated waters beyond their preference or tolerance.

Equally important, in some settings heated discharges have a beneficial effect on aquatic life, increasing survival, size, and fecundity. Appropriate instream temperatures vary by region and waterbody type, and may change over time in response to other factors. This variance is not easy to accommodate within state water quality standards, which consist of three components: designated uses, numeric or narrative pollutant criteria necessary to protect those uses, and antidegradation provisions. State standards tend to be generic, rather than site-specific. In many cases, they have been based on old data, which may not reflect biological responses such as avoidance or adaptation to higher temperatures. They may not reflect three-dimensional spatial patterns relevant to the preferred thermal niche of the organisms present in the waterbody. They may not include realistic averaging periods, or appropriate provisions governing allowable frequency and duration variation from the criteria. And they may be set based on consideration of potential effects individual organisms, rather than on the relevant aquatic population or community.

Thus, Congress recognized that a "one-size-fits-all" approach to heat would not be the most effective way to meet the broader goals of the Act.

Section 316(a) and The Companion Provisions of Section 303

Nevertheless, Congress wanted to ensure that variances would be granted only where doing so would provide an appropriate level of environmental protection. To that end, it crafted § 316(a), which provides as follows.

> [W]henever the owner or operator of any…source, after opportunity for public hearing, can demonstrate to the satisfaction of the [federal or state permit writer] that any effluent limitation proposed for the control of the thermal component of any discharge from such source will require effluent limitations more stringent than necessary to assure the protection and propagation of a balanced, indigenous population of shellfish, fish, and wildlife in and on the body of water into which the discharge is made, the [permit writer] may impose an effluent limitation under such sections [301 and 306] for such plant with respect to the thermal component of such discharge (taking into account the interaction of the thermal component with other pollutants) that will assure the protection and propagation of a balanced, indigenous population of shellfish, fish, and wildlife in and on that body of water [9].

Besides providing for a variance, Congress also required that state water quality standards for heat conform to § 316(a), and be implemented consistent with § 316(a). Section 303(g) of the Act provides that "[w]ater quality standards relating to heat shall be consistent with the requirements of section 1326 [CWA section 316] of this title [10]." In sections 303(d)(1)(B) and (D), Congress required listing and development of "total maximum daily loads" for heat using the § 316(a) standard [11]. Section 303(d)(1)(B) and (D) provide:

> (d) Identification of areas with insufficient controls; maximum daily load; certain effluent limitations revision.
>
> (1)(B) Each State shall identify those waters or parts thereof within its boundaries for which controls on thermal discharges under section 1311 of this title are not stringent enough to assure protection and propagation of a balanced indigenous population of shellfish, fish, and wildlife.
>
> (D) Each State shall estimate for the waters identified in paragraph (1)(B) of this subsection the total maximum daily thermal load required to assure protection and propagation of a balanced, indigenous population of shellfish, fish, and wildlife. Such estimates shall take into account the normal water temperatures, flow rates, seasonal variations, existing sources of heat input, and the dissipative capacity of the identified waters or parts thereof. Such estimates shall include a calculation of the maximum heat input that can be made into each such part and shall include a margin of safety which takes into account any lack of knowledge concerning the development of thermal water quality criteria for such protection and propagation in the identified waters or parts thereof.

Implementation of Section 316(a)

Early on, EPA confronted and addressed several questions about the scope of § 316(a) variances. In a 1973 memorandum, EPA's General Counsel opined that § 316(a) entitles permittees to seek a variance from either industry-specific, technology-based effluent limitations guidelines or from otherwise applicable state water quality standards [12]. In the same memorandum, EPA clarified that, in issuing NPDES permits or section 401 certifications, states may not refuse to implement § 316(a) [13]. And, with respect to the effect of a variance determination on an otherwise applicable state standard for heat, EPA found that a variance constitutes an *ad hoc* revision of the applicable standard for the particular segment of the waterbody during the permit term, so as not to run afoul of § 303(g) [14]. In addition, Delaware's water quality standards regulations provide that state antidegradation rules and implementation must be consistent with § 316(a) [15].

EPA's Rules and Guidance

The text of § 316(a) raises many questions, some of which are answered by the regulations or guidance and some of which are not. EPA's § 316(a) regulations, first published in 1979, have not changed in substance over the years [16]. Those regulations are helpful, but not terribly detailed. EPA provided more detail in a draft guidance document prepared earlier in 1977. That document was not finalized, but continues in use to this day. Collectively, here is a snapshot of what the rules and guidance suggest about how § 316(a) is to be implemented.

First, the rules clarify that the permittee bears the burden of proof and that the demonstration should focus not on the balance within a given "population" but instead on the balance within the aquatic community. Thus, the rules define the term "balance, indigenous community" (BIC) rather than "balanced indigenous population." Section 125.71(c) defines a BIC as "a biotic community typically characterized by diversity, the capacity to sustain itself through cyclic seasonal changes, presence of necessary food chain species and by a lack of domination by pollution tolerant species [17]." The rule says that a BIC may include historically non-native species introduced in connection with a program of wildlife management, as well as species whose presence or abundance results from substantial, irreversible environmental modifications [18]. A BIC normally does not include species whose presence or abundance is attributable to the introduction of pollutants that will be eliminated once technology-based requirements are met, or species whose presence or abundance is attributable to limits imposed under § 316(a) [19].

Obviously, then, defining the species that make up a "balanced indigenous community" is an important task. Because biological communities may consist of myriad species, and it would be virtually impossible to assess each in detail, the rules and guidance contemplate that § 316(a) demonstrations will focus on defining the biotic categories that comprise a "BIC" and, in appropriate cases, defining the "representative importation species" (RIS) within each category that deserve further study. According to the 1977 draft guidance biotic categories of potential concern include: phytoplankton (although experience indicates that this is virtually never an issue), zooplankton and meroplankton, habitat formers, shellfish and macroinvertebrates, fish, and other vertebrate wildlife. Section 125.71(b) of the rules defines RIS as "species that are representative, in terms of their biological needs, of a balanced indigenous community of shellfish, fish and wildlife in the body of water to which the discharge is made [20]." Section 125.72(b) provides a little more detail, saying that in specifying RIS, special consideration should be given to species mentioned in applicable water quality standards [21]. The 1977 draft guidance clarifies further that RIS may include one or more of the following categories of species:

- Commercially or recreationally valuable
- Threatened or endangered
- Critical to structure/function of ecosystem
- Capable of becoming a localized nuisance
- Necessary to the food chain
- Representative of thermal requirements of unselected species [22].

EPA's rules make the discharger responsible for developing a § 316(a) plan of study satisfactory to the permit issuer, although § 125.72(b) identifies the types of information a discharger may need to submit [23]. These include: biological, hydrographical and meteorological data; physical monitoring data; engineering or diffusion models; laboratory studies; representative important species; and other relevant information.

For existing dischargers, EPA's rules provide that § 316(a) demonstrations may be based on a showing of the absence of prior appreciable harm. Section 125.73 (c)(1) specifies that such a demonstration may show that (1) no prior appreciable harm has resulted from the normal component of the discharge (taking into account the interaction of such thermal component with other pollutants and the additive effects of other thermal sources), or (2) despite the occurrence of such previous harm, the desired alternative effluent limitations (or appropriate modifications thereof) will nevertheless assure protection and propagation of a BIC [24]. Section 125.73(c)(2) directs permit writers to consider the length of time in which the applicant has been discharging and the nature of the discharge, when determining whether or not prior appreciable harm has occurred [25]. EPA's 1977 draft guidance refers to "no prior appreciable harm" studies as "Type I studies [26]."

Under the 1977 draft guidance, facilities that cannot, or choose not to, do a Type I study may do a Type II or Type III study. Type II studies focus on RIS. Type III studies (a variant of the Type III study is the "Type III Low Potential Impact Determination" study) need not necessarily evaluate RIS, if there is adequate evidence that broader biotic categories are protected. Facilities that may pursue these types of studies include: (1) new sources not yet discharging; (2) facilities which have not been discharging heated effluent for a sufficient period of time to allow for evaluation of effects; (3) facilities discharging into previously despoiled waters which have begun to improve; and (4) facilities which have made or propose to make major changes in operations [27].

A § 316(a) variance terminates at the end of each NPDES permit term. This means the permittee must request renewal of any § 316(a) variance as part of its application for permit renewal. That said, EPA does not require that an applicant start its demonstration from scratch at each permit renewal. Instead, the permit writer has 60 days after receipt of the permit renewal application to request additional information [28]. Typically, permit writers will request additional information only where:

- the nature of discharge has changed
- the nature of the aquatic populations has changed, or
- there is information suggesting that the original variance was "improperly granted [29]."

Emerging Issues

Of course, despite this vast experience applying § 316(a), new issues continue to arise. The limited scope of this article allows me to name just a few, and to suggest what I believe to be, potentially, the correct resolutions. These include:

- Whether or not the listing requirements of § 303(d)(1)(B), (D) apply whenever heat is an issue, or only in situations in which the impairment is attributable to a point source discharge. (I believe a fair reading of the law and legislative history indicate the former is the case, although EPA has recently suggested it may change course and pursue the latter interpretation.)
- How to deal with situations in which a relationship exists between heat and other pollutants for which aquatic criteria are not attained, especially where a study shows a BIC is attained. (In my own view, this situation suggests that the criterion for other pollutants might also be unduly stringent.)
- Whether and how to take into account current or future environmental enhancements that protect a BIC in the waterbody overall, despite some small localized effects. (Because § 316(a) focuses on the end result for the BIC, it seems to me that a permit writer should take such enhancements into account in making a final § 316(a) determination.)
- Whether or not water quality "trading" (among sources of heat, or among sources of heat and other pollutants that could affect a BIC) is permissible. (Again, it seems to this author that the Act poses no impediment to such trading, as long as a BIC is protected (or, if there is no BIC because of the presence of the non-thermal pollutants to be traded, progress towards a BIC is made.)

Some Thoughts in Closing

No thumbnail sketch of § 316(a) and its implementing regulations can do justice to all of the legal, technical, and policy issues that can arise when the deceptively simple language of § 316(a) is applied in a specific case. Nevertheless, it demonstrates the need to build on existing information and experience to make good decisions about what appears to be a new wave § 316(a) variance assessments.

Based on that experience, here are some suggestions for addressing § 316(a) going forward.

- Start with what you know.
- If new studies are required, make sure to plan those studies carefully and do not skimp.
- Make sure everyone involved is on the same page about study objectives, schedule, and all other important elements of the inquiry.
- Do not go in search of perfect certainty–there is no such thing in environmental science.
- Focus on what counts–the overall health and sustainability of the relevant populations and community.

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8. *Quality Criteria for Water*, U.S. EPA (1976).

9. 33 U.S.C. §1326(a).

10. 33 U.S.C. § 1313(g).

11. 33 U.S.C. § 1313(d)(1)(B), (D).

12. Memorandum to EPA Deputy Assistant Administrator for Water Planning and Standards from Robert V. Zener, Acting Deputy General Counsel, dated December 28, 1973.

13. *Id.*

14. Id.

15. Delaware Department of Natural Resources and Environmental Control, Surface Water Quality Standards, § 3.4 (1999).

16. 40 C.F.R. §§ 125.70-.73 (2002).

17. 40 C.F.R. § 125.71(c) (2002).

18. Id.

19. Id.

20. 40 C.F.R. § 125.71(b) (2002).

21. 40 C.F.R. § 125.72(b) (2002).

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3
DEVELOPMENT OF THERMAL WATER QUALITY STANDARDS AND POINT SOURCE IMPLEMENTATION RULES IN WISCONSIN

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Introduction

With time, and for various reasons, rules need to be revised. Such is the case with Wisconsin's rules for regulating thermal discharges to water. Wisconsin's thermal rules have been under revision for a significant time. An advisory committee made up of a diverse group of professionals has been the primary vehicle for making the revisions. The author is chair of this advisory committee. The primary goal of the advisory committee is to produce a water quality-based thermal rule package that is environmentally protective, scientifically defensible, and reasonably implementable. Much effort has been made by the advisory committee members, as well as others, to assure this goal is met.

The purpose of this paper is to provide details of the process by which the Wisconsin Department of Natural Resources (DNR) is revising its water quality thermal rules. Some details of the actual revisions will be presented, but the focus of the paper is to explain who is involved in the rule revisions process, what the roles of the participants are, and what general strategies have been used for making revisions. A brief history of why Wisconsin DNR is revising its thermal rules is included to provide context to the revision process. A section providing suggestions and lessons learned is included, as well, in efforts to share some of the important aspects of our experience.

It is important to note that Wisconsin's thermal rule revisions process remains ongoing at this time. Thus all details in this paper regarding the rule revisions are to be considered "draft". The draft rule is expected to go out for public hearing and comment in spring 2004, and the final rule is expected to be promulgated late in 2004 or early in 2005.

Brief History

The State of Wisconsin established thermal standards for water quality in 1974 as part of its delegated responsibility of implementing the Federal Water Pollution Control Act Amendments of 1972 (i.e. the Clean Water Act or Public Law 92-500). These standards took effect in 1975, and stated that "the maximum temperature rise at the edge of the mixing zone above the existing natural temperature shall not exceed 5°F for streams and 3°F for lakes." Further, "the temperature shall not exceed 89°F for warm water fish." These standards are included in Chapter NR 102 of Wisconsin's Administrative Code. Additional thermal standard language, primarily narrative in nature, is also included in Chapter NR 102.

However, as a result of two lawsuits, the State Supreme Court of Wisconsin in 1979 ruled that the thermal standards established by the DNR were unconstitutional. The basis for the Court's ruling was that the application of the standards was not based on water quality parameters or conditions and thus should not be considered water quality-based. The court believed instead, that the application of the standards was categorically-based. A confounding problem for the Court was that although other states were using (and continue to use) similar standards as what Wisconsin had established, there were no federally recommended water quality-based or categorical treatment technology-based limits for thermal discharges. Since Wisconsin's statutes allow State standards to be more stringent than federal guidelines only when they are water quality-based, the Court ruled that Wisconsin's thermal standards were invalid and unconstitutional.

Following the State Supreme Court ruling in 1979, the DNR was unable to apply the established thermal standards in Wisconsin Pollution Discharge Elimination System (WPDES) permits. Instead the DNR applied a maximum end-of-pipe discharge temperature of 120°F, based on protection of incidental human contact, in most permits for dischargers of heated effluent. This approach was challenged in 1991 when the U.S. Environmental Protection Agency (EPA) Region 5 objected to two electric utility permits submitted by DNR on the basis that they did not include thermal limits. While the DNR recognized that discharges from the two facilities had resulted in fish kills, it was unable to act to remedy the situation because of the 1979 Supreme Court ruling.

Since the U.S. EPA Region 5 would not approve the two permits unless the thermal issues were addressed, an agreement was reached that the thermal standards of Chapter NR 102 needed to be revised. The resulting revised rule was to include scientifically sound and water quality-based standards that were protective of the biological, chemical, and physical components of receiving waters. Additionally, the revised rule was to enable enforceable water quality-based effluent limitations for thermal discharges. A U.S. EPA grant was used to support the thermal rule revision effort.

As a result, an advisory committee (AC) was formed in 1994. The AC met approximately 18 times between November 1994 and August of 1997, making significant progress in meeting the revision objectives noted above. The AC's work resulted in a draft proposed rule package which included significant revisions to both Chapter NR 102 and Chapter NR 106 (dealing with application of water quality standards to point source discharges) of Wisconsin's Administrative Code. The draft rule package went out for public hearing, review, and comment in the autumn of 1998. Many relevant comments were received by DNR into 1999.

Despite the progress that had been made, finalization of the thermal rule revisions was halted in 1999 due to external opposition, followed by internal DNR issues related to Department-wide reorganization, retirement, and staff reassignments. The thermal rule revision effort remained in hiatus until May 2001, when the author was hired and assigned the task of finalizing the rule revisions. The AC was reconvened (some members the same as the original, some new) and has met 11 times since October 2001. A DNR internal work group worked on making significant thermal standards revisions for the majority of 2002, and presented the results of its work as a proposal for the AC to consider. Overall, the focus of the renewed effort has been to make final revisions to the thermal rules, using the 1998 proposed draft rule as a starting point. Comments received in 1998 and 1999 have been considered. The AC and DNR internal staff are nearing the completion of a new draft rule package. As stated above, the draft rule is expected to go out for public hearing and comment in spring 2004, and the final rule is expected to be promulgated late in 2004 or early in 2005.

General Rule Revision Process

Primary Operating Principles

The AC has operated under four primary principles in revising Wisconsin's thermal rules (see Table 3-1). To address the primary argument of the State Supreme Court, the first principle is that the resulting rules must be water quality-based. Several things have been done to assure the revised rules will be considered water quality-based, and include the following:

- The thermal standards are developed for each particular water body use designation (i.e. cold water, warm water, Great Lakes, specific rivers, etc.).
- All data included in criteria development is based only on species known to exist in Wisconsin.
- Acute criteria are based on ambient temperature, rather than a static value (e.g. 89°F).
- Data from three different sub-lethal endpoints are included.
- Sub-lethal criteria are based on the month that a given endpoint is known to exist in Wisconsin.

Table 3-1
Primary Operating Principles

The Rules Must Be
1. Water Quality-Based
2. Environmentally Protective
3. Legally & Scientifically Defensible
4. Reasonable in Its Application

Second, the rule must be environmentally protective. This is being accomplished by including numerous and varied laboratory-derived and field observed biological effects data, as well as considering the comments of appropriate fisheries biologists and others. Third, the rule must be legally and scientifically defensible. The use of an extensive amount of detailed data is a primary way that this principle is being accomplished. Additionally, this has been accomplished by having a rationale for each decision made. Fourth, the rule must be reasonable in its application. The objective here is to assure that those who will be required to implement the rule can do so with a reasonable amount of effort and/or impact, and that the rule is not overly burdensome. Similar to this is what could be considered a fifth operating principle of the AC - that the rule revisions and the final rule should pass the “common sense” test. This “test” has been considered often throughout the rules revision process, and has helped the AC develop a much sounder rule that incorporates the four primary principles. The “common sense” test helps the AC to consider how to adjust good-intentioned initial revisions based on good data that just don’t “fit” a given scenario into much more relevant and reasonable proposals.

Participants

The thermal rules revision effort has included participation of three basic groups of people - an advisory committee (AC), an internal work group (IWG), and others. The following paragraphs describe the basic make-up and role of each of these groups.

From the beginning of the thermal rules revision effort the DNR as strived to be very inclusive in terms of external participation and input. Letters were sent out to acquire nominations and requests to participate as members of the AC. The DNR assured a working balance of government, industry, environmental advocacy, municipality, and academic representation on the AC. The current AC consists of 12 members, including only two DNR staff (the Chair and one legal staff representative). The other AC members represent the following: U.S. EPA Region 5 (1 person), power industry (2 people from two different companies), paper industry (1), food processor industry (1), aquaculture industry (1), environmental advocacy (2 people from two different organizations), municipal sewage treatment operation (1), and academia (1). Each member is asked to represent others in their “sector” as much as possible, as it is important to keep the size of the AC to approximately 12 in order to keep AC functions and meetings manageable.

The role of each of the AC members is to participate in the AC meetings and to review and comment on numerous documents and spreadsheets, including the draft rule. Participation includes reviewing and commenting on new proposals and providing requested information. Active participation by each member is very important for the AC process to be successful because it is this input that helps to assure the rule will meet the principles discussed above. The AC process provides an opportunity for potentially impacted entities to share their concerns, propose options, and suggest alternatives before a draft rule is prepared. All this information is considered as rule revisions are proposed. This process should eliminate most “problems” from the rule before a draft goes out for public review and comment.

The IWG is an informal, loosely organized group consisting of numerous central office and regional DNR staff. The IWG participants come from a wide variety of programs, such as Watershed Management, Fisheries Management and Habitat Protection, and Integrated Science Services (fisheries research). The IWG is loosely organized to allow for different staff with a particular interest and/or expertise to participate in those meetings that are most relevant to them or their program. The role of the IWG members is very similar to that of the AC members, and includes participation in the IWG and AC meetings and review and comment on numerous documents and spreadsheets, including the draft rule. The IWG held several of its own meetings to develop criteria which were presented to the AC to consider and comment on. Like the AC, participation includes reviewing and commenting on new proposals and providing requested information. Further, active participation by each IWG member is also very important for the AC process to be successful because their input also helps to assure the rule will meet the principles discussed above.

An ultimate goal of the participation of each AC and IWG member is to achieve member "buy-in" to the proposed draft rule before it goes out for public review. This will help assure the rule will meet the principles outlined above, and gives the DNR confidence that the rule as a whole is appropriate.

Other participants to the rules revision process include three primary types of people. One is those who participate in a public hearing or the public comment period by providing the DNR with oral or written comments. Two is those who wish to be periodically updated on the status of the rules revision process. This group is considered a distribution list group, and receives a brief update via email or written letter approximately once or twice per year. Three is a person who participates at one or two meetings at the request of the AC or the Chair to provide specific expertise or to answer specific questions. An additional person or two have attended some AC meetings. These people are allowed to attend, but not to participate unless asked to by one of the AC members. Finally, an additional DNR staff person participates in the AC meetings as a note taker, which helps the meetings run smoothly and helps to get meeting summaries out to meeting participants more quickly.

General Rules Revision Strategy

The following will focus on the general strategy used by the current AC, and reflects the rules revision effort since May 2001. Since the reconvened AC had a draft rule to start working with, a strategy was developed to take advantage of this. The strategy is intended to provide order and directed flow to the rules revision process. As mentioned in the "Brief History" section above, revisions are being made to two rules: Chapter NR 102 and Chapter NR 106 of Wisconsin's Administrative Code. Chapter NR 102 houses the water quality standards, including criteria. Chapter NR 106 houses the rules for how point source dischargers implement the water quality standards. Thus the general strategy has been to work through Chapter NR 102 revisions first, then work through Chapter NR 106 revisions, produce a new draft rule and package it for public review and comment, and then finalize and promulgate the rule. Figure 3-1 diagrams this strategy.

The basic idea is that the AC and IWG would consider the existing draft rule and the comments received on it and determine if revisions to the draft rule are needed. If it is determined that a portion of the rule needs to be revised, then the portion that needs to be revised is identified, revision suggestions are made, and then a particular revision option is selected and the revision is made. The process repeats itself until the AC and IWG feel sufficient revisions have been made and a draft rule should be drafted.

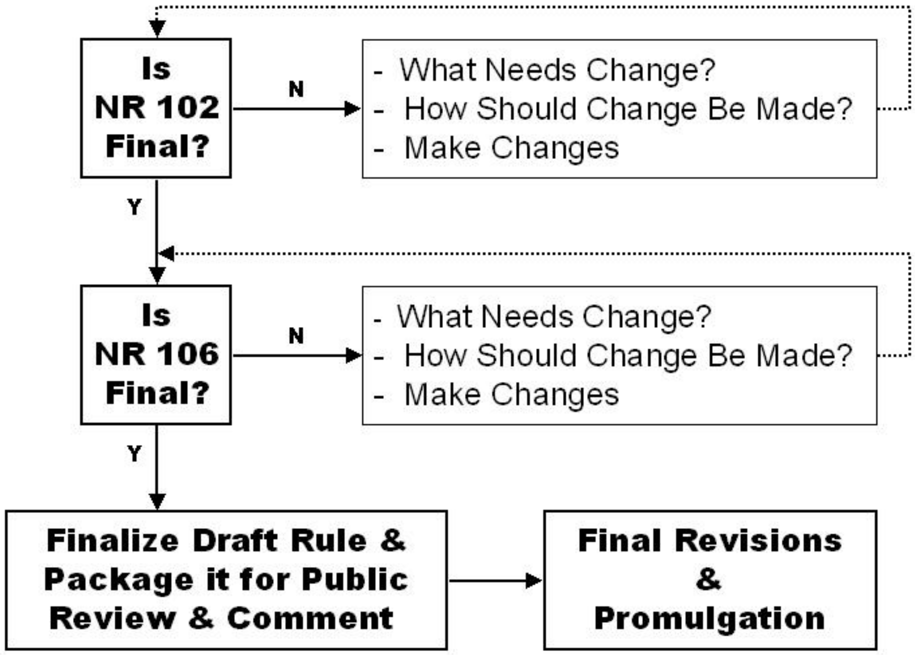

Figure 3-1
General Strategy for Finalizing Thermal Rules Revisions

There are two reasons for moving through this strategy from Chapter NR 102 revisions to Chapter NR 106 revisions. First is because it makes sense to start with the water quality standards and move to the implementation rules, since the implementation rules incorporate the water quality standards. Second is to attempt to assure the water quality standards are developed independently of point source discharge limit consideration influence. This is an important consideration for this effort because the AC membership from the beginning has consisted primarily of point source discharge interests and because most of the comments received regarding the draft rule were related to Chapter NR 106 issues. It is easy to understand that an AC with a point source discharge bias (what could be characterized as an innocent bias in our case) would be primarily focused on the end result of the rules revisions (i.e. the permit limit they will be issued). However, it is important that the water quality standards be valid, defensible, and protective for all conditions and uses.

The basis for this is that the water quality standards will have broad application and thus must be able to stand on their own. Chapter NR 106, as an implementation rule, uses the water quality standards of Chapter NR 102 in a specific way for a specific group of users. Other rules dealing with stormwater detention ponds, submerged heat exchangers, fisheries-related issues, and others could potentially (and likely will in the near future in Wisconsin) use the thermal water quality standards we are developing for these other applications within their own implementation rules. It is not to say that the standards could never be developed from a point source discharge perspective and be applicable to other uses, but it appeared that a bias did exist in our process. This issue was brought to light shortly after the AC reconvened. The reconvened AC was ready to move straight to the Chapter NR 106 revisions, feeling the draft water quality standards of Chapter NR 102 were acceptable. However, DNR fisheries staff did not accept that the draft water quality standards would be ecologically protective when applied in their implementation rules. This is why the IWG worked for 10 months making revisions to the water quality standards and then presented their proposed revisions to the water quality standards to the AC for review and comment. The result of this approach appears to be that the final Chapter NR 102 rule will be much more robust and be much broader in its application, which is important to prevent additional Chapter NR 102 revision efforts in the future.

The take home message of this section is to understand how different related rules fit and work together, and to know who is participating in your revision process and what bias's they may bring to the table, innocent as those bias's may be.

It should be noted that despite the Chair's best efforts, it was impossible to work through the strategy as depicted in Figure 3-1 perfectly because of the numerous interwoven issues between the two rules. However, the strategy has certainly aided the AC's and IWG's efforts, and thus has been a valuable tool for Wisconsin's thermal rules revision process.

Suggestions and Lessons Learned

Although our rules revision process is not yet complete, there are many things that we have learned that can be passed along for others to consider. It is the author's understanding that other States are considering making revisions to their thermal rules. Applying some of these suggestions and concepts would likely aid the rule revision efforts of these other States. The following suggestions are divided into three parts: those that are general in nature and apply to all participants; those that are addressed to the Chair or leader of the revision process (likely to be staff of a State agency or Department); and those that are addressed to the non-Chair, active participants of a rule revision process. The suggestions are intended to be self-explanatory and thus will include only limited narrative or examples.

Process Suggestions – General

The following should be considered and honored by all rule revision participants:

- Be willing to trust the other participants. This is critical to the group working together well and the process running smoothly.
- Realize and accept that no participant is likely to get everything they want. Like many processes, an open rule revision process is full of give and take discussions and suggestions, based on the prioritization of relevant issues to individual participants.
- Listen to and consider other participant's comments and suggestions. This is not to say that everyone's comments and suggestions need to be mandated and followed, but rather that everyone should be given an opportunity to be heard. It is in everyone's best interest to listen to what each participant has to say and to consider its relevance and applicability to the overall issue.
- Think "outside the box". Great ideas are not considered and creativity stifled when participants do not consider novel concepts for problem solving. Wisconsin's rules revision process has certainly considered options and ideas that are "outside the box" of historic regulatory approaches. This is considered to be beneficial to those involved in the process, as well as those potentially impacted by the rule.

- Remember that aquatic thermal issues are NOT trivial. Rather they include many interconnected related issues such as many individual and community-level biological parameters, many site-specific characteristics, weather, and influence on toxicity of other pollutants. The bottom line is that many issues, even seemingly “minor” issues, may take significantly more time or effort to work through than originally anticipated. Thus all participants should keep this in mind and be flexible when this occurs, remembering the reason is often that thermal issues are complicated.

Process Suggestions – Chair Perspective

As the Chair or leader of the rule revision process, it is important that the following are considered. Several of these were attempted from the beginning, but took time into the process to be realized. Others were learned after the process began. Several suggestions are related to or dependent on others. In any event, if followed from the beginning and carried through to the end, these suggestions will likely aid in leading the rule revision process effectively.

- Include as many appropriate participants as possible. This includes both internal and external participants if you are working for a State agency. The only reason to limit the number of participants is to keep the group at a manageable and effective size. If the group gets too large the process can get bogged down. However, if the group does not include significantly relevant participants, the process could get side tracked or derailed at some point, often near what was supposed to be the end.
- Know who the participants are and what their concerns are. Visiting facilities and learning about their operations is a great way to achieve this, as well as build trust.
- Remember that the other participants have other things to work on. For example, the rules revision effort is approximately 90% of the author’s workload. The author had to remember that this effort was not even 10% of most other participants’ workloads. This is important to remember when assigning tasks and asking for input. It is the participants responsibility to respond and participate, but it is the Chair’s responsibility to make every effort not to ask too much of each participant.
- Keep the process moving forward. It was difficult to get momentum rolling after an approximate three-year hiatus, followed by a 10-month change of plan. Scheduling meetings every four to six weeks has helped keep the process moving and has seemed to be a reasonable pace for the AC meeting participants.
- Provide summaries of decisions made. It is encouraging for the group to see/remember that decisions are being made and that the process is moving forward.
- Document everything and organize the information well. It is inevitable that much material will be accumulated during a rule revision effort, including many versions of drafts. Keeping good notes with dates, and keeping this information organized and accessible is very important as the Chair.
- Provide interactive tools that allow participants to “play” with relevant, real-world data. In our case a significant example of this is a spreadsheet that provides input of default or site-specific data for each of the variables in the limit calculation, as well as a plot of the resulting monthly ambient temperature, acute and sub-lethal permit limits, and discharge temperatures for a year. Participants and the author have used this tool very extensively to “view” what proposed revisions mean in the big picture.

- Provide opportunity for participants to provide input as much as possible. For example, participants are given an open time to voice general comments at the beginning of every AC meeting. Additionally, every email and letter sent to participants includes invitation to contact the Chair to ask questions or provide comments.
- Understand that the interaction of various rules may be confusing to others. It is not uncommon for portions of the rule being revised to influence aspects of other rules, and vice versa. Also, participants may have to consider the resulting interaction of the rule being revised with multiple other rules. It is important to consider these possibilities and the resulting concerns participants may have. And quite frankly, rule language is often not easy to read or understand.
- Provide periodic updates to interested parties. As mentioned above, emails have been sent to those who wish to be updated periodically. These emails are short summaries intended to let recipients know the rules revision process is moving, know the very basic details of what has been accomplished, and the latest approximate timeline. Additional means of providing periodic updates is to make site visits or provide summaries at interest group meetings or conferences.

Process Suggestions – Participant Perspective

The following suggestions are intended for the active participants of the rules revision process. Each of these suggestions can help the rules revision process run more smoothly.

- Don't assume the rule revisions are being "designed" to be against you or your interest. Holding to this assumption makes it impossible for trust to occur in the process, and ultimately leads to a lot of wasted time and effort.
- Consider participation in the rules revision process as a commitment to honor and contribute to. Participation is an opportunity to provide direct input and influence on the revised rule. This opportunity should be treated with respect.
- Provide comments and suggestions throughout the rules revision process, especially when prompted. Participant comments and suggestions are vital to the success of the process. Timely submittal of comments and suggestions is also very important. As mentioned above, it is the Chair's responsibility to not expect too much from the participants, but it is the responsibility of the participants to provide requested responses, as well as miscellaneous comments and suggestions.
- Don't wait until the public comment period to make comments. For reasons mentioned in the above bullets, it would be very damaging to the rules revision process for an active participant to hold comments and suggestions until a public comment period. This action would certainly break trust with other active participants.
- Especially for those who represent an industry, provide real-world examples and data of your operations. In our case, the author was not familiar with the details of each industry's operations as they influence different aspects of the rule revisions. Significant in our case, and likely in most cases, is that several different industries will have interest in and be potentially influenced by the rule revisions. Thus, it is not easy for the Chair to know how the rule will impact specific aspects of a given industry. This is a significant reason why it is important for all appropriate parties to participate in the process from the beginning.

- Share revision updates and ideas with cohorts in your field. This will help to let others not actively involved, but ultimately interested, in the rule revisions to have a heads up on what is being proposed. Additionally, it provides opportunity for these other interested parties to voice concerns or suggestions that may differ from those previously discussed at meetings. These new concerns or suggestions can be shared with other participants at future meetings.

Summary Details of the Rules Revisions

Some of the more significant details of the revisions currently being proposed for Chapter NR 102 and Chapter NR 106 of Wisconsin Administrative Code are highlighted in this section. The purpose is to provide a glimpse of the types of revisions being made, as well as an idea of the types of issues that have been worked through to consider these revisions. Consider what was discussed in the previous sections to get an appreciation of the effort and time needed to reach these proposals, and to consider how the revisions are meeting the goals and principles laid out for our effort. Without question the most current proposed revisions are much more detailed and comprehensive than the original 1998 draft rule. Please remember that all things listed in this section are the current state of our work, but are to be considered draft until a final rule is promulgated.

Chapter NR 102

Two primary components of Chapter NR 102 are being revised-the water quality criteria and the ambient (background) temperature. Revisions pertaining to each are presented below.

Both acute and sub-lethal water quality criteria have been developed. All criteria are based on fish data as the original AC found 1) fish data to be protective of other species and 2) insufficient data existed on thermal impacts to other aquatic organisms. In all, data and information from 692 references, representing the years 1874-2003, have been used and cited throughout the rules revision process, with most of these references pertaining to criteria development. The vast majority of references are from peer-reviewed publications or agency reports, etc., while a small number are from personal communications or other sources.

Acute criteria are based on 360 Upper Incipient Lethal Temperature (UILT) - acclimation temperature data pairs, and will be applied as an absolute daily maximum. Using the UILT-acclimation temperature data in concert with regression and analysis of covariance statistical analyses has enabled the development of acute criteria that work on a “sliding scale” with ambient temperature, thus eliminating the use of static number criteria (such as 89°F). This sliding scale considers the receiving water’s ability to assimilate heat (more when water temperatures are cooler and less when water temperatures are warmer). Separate criteria have been developed for each water body use designation by considering only data from fish that exist in each given water body classification for that classification’s criteria.

The sub-lethal criteria are based on three different life stages-gametogenesis, spawning, and growth. Twelve data points were used to develop criteria to protect gametogenesis, 444 data points were used to develop maximum spawning temperature criteria, and 124 data points were used to develop maximum no growth temperature criteria. A matrix considering water body use designation (and the fish that exist in each) and the months each life stage occurs in Wisconsin, in combination with a polynomial regression, was used to develop a single sub-lethal criterion for each month. The sub-lethal criteria are to be applied as a seven-day average.

Unlike for other pollutants, both acute and sub-lethal criteria will be applied at the edge of a mixing zone. An acute mixing zone is allowed because heat is a non-conservative pollutant that dissipates. The acute and sub-lethal criteria are applied at the edge of the same mixing zone to streamline and simplify the application of the criteria. This is possible since the acute criteria are applied as absolute daily maximums and the sub-lethal criteria are applied as seven-day averages. Yet the two criteria are applied simultaneously.

Ambient temperatures were developed using data from as many stations in Wisconsin between October 1987 and December 2002 as possible. The time range is to capture data from relatively warm, cold, and average temperature years. Approximately 4070 monthly average temperature values were used to develop ambient temperatures for all waters including the Wisconsin portions of the Mississippi River, except for the Great Lakes. Data sources included U.S. Geological Survey (USGS) Wisconsin Water Year Books, other USGS data, DNR Wisconsin River data, and Fox River data from the Green Bay Metropolitan Sewerage District (which keeps water quality data for many locations in the lower Fox River and lower Green Bay). This data is derived from continuous 15-minute temperature readings. Approximately 938 monthly average temperature values were used to develop ambient temperatures for Great Lakes waters of Wisconsin. Sources of the Great Lakes data were intakes at numerous water treatment facilities along Lake Michigan and Lake Superior. This data is derived from daily temperature readings. All data are organized into appropriate water body use designations by month. Final ambient temperatures are geometric means of all monthly averages for a given month and water body use designation.

Chapter NR 106

The primary component of Chapter NR 106 being revised is the limit calculation procedure. A modification of the mass balance equation for toxics has been developed for use in calculating limits for temperature. The equation integrates the appropriate water quality criterion, ambient temperature, effluent flow, and mixing zone considerations (stream flow and fraction of effluent from the receiving stream for flowing waters, and mixing zone area and an empirical factor for lakes) to calculate an end-of-the-pipe temperature limit that assures the water quality criterion will be met at the edge of the mixing zone. The same equation is used for both the acute and sub-lethal criteria.

Site-Specific Options

The details of Chapter NR 102 and Chapter NR 106 provided above describe the development of default criteria, default ambient temperatures, and default limit calculations. These are the default regulations in the rules. Despite making significant effort to produce defaults that are specific to given water body use designations, the DNR realizes that some discharge site conditions may significantly vary from the rule defaults. Thus, those who wish to develop alternative, site-specific water quality criteria, ambient temperatures, or limit calculations may do so. However, the alternatives must be approved by the DNR before being used.

Concluding Remarks

It is the author's intent and hope that details shared in this paper regarding the process by which the Wisconsin Department of Natural Resources (DNR) is revising its water quality thermal rules will be useful to those who will be making similar revisions to thermal or other water quality rules in the future. The rules revision process is not easy, but it can be done as effectively and efficiently as possible if all participants of the process work together in trust towards one another and commitment to completing the process. The outcome can be rules that meet objectives, and are environmentally protective, legally and scientifically defensible, and reasonably implementable. DNR is looking forward to this outcome.

4
UPDATING A TEMPERATURE CRITERIA METHODOLOGY FOR THE OHIO RIVER MAINSTEM

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Introduction

Temperature criteria provide an important basis for evaluating and regulating effects from cooling water and other thermal discharges in order to protect fish and other aquatic life. The technical justification for the temperature criteria in the ORSANCO Pollution Control Standards was originally developed by Ohio EPA [1] using a standardized methodology for calculating seasonal and monthly average and daily maximum temperature criteria. This approach used data from the thermal effects literature to create a thermal effects database for fishes. This data was then used within a procedure that calculates various behavioral and physiological thresholds for a list of representative fish species that are intended to represent the fish assemblage of a particular river. Ohio EPA took this same approach in setting temperature criteria for inland waters and Lake Erie in the 1978 revisions to the Ohio water quality standards (WQS). The temperature criteria derivation process was later incorporated within the Fish Temperature Modeling system that is part of the Ohio ECOS data management system developed and operated by Ohio EPA. Much of the literature upon which the thermal effects database is based dates from before the late 1970s with some sources dating from the 1940s and 1950s. Because the literature database exceeds 30-40 years of age and newer sources have become available, concerns have been raised about the contemporary applicability of the existing Ohio River temperature criteria. The incorporation of more recent information is seen as being needed to determine the relevancy and appropriateness of the current temperature criteria. Other considerations, including the use of various thermal thresholds (e.g., incipient lethal temperature, critical thermal maxima) and of protecting 100% of the representative species vs. 95%, etc. have also been raised.

The current temperature criteria were derived using a methodology developed by Ohio EPA [1] to calculate seasonal temperature criteria for the Ohio River mainstem and the other major mainstem rivers of Ohio. The original Fish Temperature Modeling system was developed as a mainframe routine, but presently exists in FoxPro as part of the Ohio ECOS data management system. MBI is presently developing an update to this system for ORSANCO. This will include updating the thermal effects database and reviewing the applicability of other criteria derivation methods that have been developed since the inception of the Ohio EPA methodology in 1978. The key variables that determine the outputs of the model are the list of representative fish species and the temperature tolerance endpoints used for each. The temperature tolerance endpoints used in the model were derived from the extensive literature database that was assembled in support of the existing methodology prior to 1978 [1].

Methods

The primary input variables to the Fish Temperature Model are four thermal parameters for each representative fish species; a physiological optimum temperature, a maximum weekly average temperature for growth, an upper avoidance temperature, and an upper incipient lethal temperature. These were derived from an extensive literature review and were assigned to each Ohio River basin fish species for which sufficient thermal data could be found. When multiple thresholds were available for a particular species, the most ecologically and geographically relevant data was used.

Thermal Parameters

Four thermal input variables are used in the Fish Temperature Model to determine the summer (June 16–September 15) average and daily maximum temperature criteria. However, in developing these baseline input variables, six thermal parameters were first considered by Ohio EPA [1]. General concepts of thermal responsiveness (e.g., acclimation) were considered and are discussed in more detail elsewhere [2]. Of the six thermal parameters that were inventoried for each fish species, the upper incipient lethal temperature (UILT) and the critical thermal maximum (CTM) are considered lethal thresholds and the remaining four (optimum, final preferendum, growth, and upper avoidance) are considered sublethal parameters. At the time the Ohio EPA [1] methodology was developed, the rapid transfer method (from which the UILT is derived) was viewed as providing a firmer basis for physiological response than does the slow heating method on which the CTM is based [2]. Each of the six thermal parameters are defined as follows:

Upper Incipient Lethal Temperature–at a given acclimation temperature this is the maximum temperature beyond which an organism cannot survive for an indefinite period of time;

Critical Thermal Maximum–the temperature at which a test organism experiences equilibrium loss resulting from a steady increase in temperature (approximately 0.5°C/hr.);

Optimum–the temperature at which an organism can most efficiently perform a specific physiological or ecological function;

Final Preferendum–the temperature at which a fish population will ultimately congregate regardless of previous thermal experience [3];

Upper Avoidance Temperature–a sharply defined upper temperature at which an organism that a given acclimation temperature will avoid [4];

Growth–the mean weekly average temperature for acceptable growth [5].

A fish species was included in the database when a minimum of three of the six parameters was available. MBI is in the process of updating the literature database to include sources available since 1978. We expect that this will not only add new species to the thermal database, but also provide a wider availability of thermal parameters for each species.

Thermal Input Variables

The analysis used four thermal input parameters that included: 1) the optimum or final preferendum; 2) the mean weekly average temperature (MWAT) for growth as described by Brungs and Jones [5]; 3) the upper avoidance temperature as described by Coutant [4]; and, 4) the upper incipient lethal temperature (UILT) at acclimation temperatures of 27-30°C. Thermal parameters compiled from various literature sources for 84 freshwater fish species were used as the primary database for the model. Missing parameters were estimated by calculating relationships between some of the six thermal parameters that were gleaned from the literature for each species—at least three of the six had to be available for a species before this procedure could be used. In order to estimate the missing thermal parameters, calculation of the differences between the; 1) optimum and UAT, 2) optimum and UILT, 3) optimum and critical thermal maximum (CTM), 4) UAT and UILT, 5) UAT and CTM, and 6) UILT and CTM were made [1]. Extrapolations were then made in a stepwise procedure as follows:

1. based on the species family relationships (e.g., longnose gar, Lepisosteidae); or
2. based on the next closest family if information for a parameter did not exist within the species family; or,
3. based on the average of all families as a last choice.

The four primary thermal parameters are stored by species and accessed by the model when that species is designated as being representative.

Representative Fish Species

The derivation of temperature criteria is also dependent on the development of a list of representative fish species, which is the primary input variable for the model. Representative species constitute a *subset* of the assemblage that have sufficient thermal tolerance data upon which temperature criteria can be derived. There is a tendency for species regarded as being tolerant to a wide variety of environmental impacts to be included in these databases, which is similar to other water quality criteria databases. As such, there will likely be species present in the potential assemblage that are more sensitive to the parameter that is being considered. This approach is simply a best attempt to represent the entirety of the assemblage and it is limited by the extant tolerance databases. As such, the model output will propagate a degree of uncertainty, which can be considered in the eventual derivation and application of the temperature criteria. In developing a list of representative fish species for a particular water body or area, the following criteria for membership were used:

- species that represent the full range of response and sensitivity to environmental stressors;
- species that are commercially and/or recreationally important;
- species that are representative of the different trophic levels;

- rare, threatened, endangered, and special status species;
- species that are numerically abundant or prominent in the system;
- potential nuisance species; and,
- species that are indicative of the ecological and physiological requirements of representative species that lack thermal data.

In addition to these conceptual guidelines, the historical occurrence of fish species in a particular water body is also considered.

Temperature Criteria Derivation Process

Average and daily maximum summer temperature criteria were determined via an analytical process similar to that developed by Bush et al. [6]. Temperature tolerance values for 69 Ohio River basin fish species are presently contained in the Ohio EPA database (Table 4-1). These values include the four primary thermal parameters described previously; optimum, mean weekly average for growth, upper avoidance, and upper incipient lethal temperatures. The model permits alternative values to be substituted and these can be maintained as alternate databases to be used for computing the effect of any species-specific differences on the derivation of summer season thresholds. The tolerance values in the existing database [1] were used in the derivation of the summer average and maxima for the Ohio River mainstem. The procedure is simply one of listing each representative species under each thermal parameter adjacent to the whole Fahrenheit temperature when it is exceeded. The cumulative effect of increasing temperature is readily apparent as each species thermal criteria are exceeded. This process indicates where the various species occur (with respect to increasing temperature) relative to each other and does not indicate exact thresholds or limits. The temperatures at which 100%, 90%, 75% and 50% of the representative fish species for the four thermal thresholds are then derived to show what proportion of the representative assemblage is protected at a given temperature. The long-term survival temperature is calculated from the short-term survival (i.e., the UILT) as UILT minus 2°C. The following guidelines are used to derive summer average and maximum temperature criteria.

Averages should be consistent with:

- 100% long-term survival of all representative fish species;
- growth of commercially or recreationally important fish species;
- growth of at least 50% of the non-game fish species;
- 100% long-term survival of all endangered fish species; and
- the observed historical ambient temperature record.

Daily maxima should be consistent with:

- 100% short-term survival of all representative fish species; and
- the observed historical ambient temperature record.

Non-summer season temperature criteria are derived from the historical temperature record and considering other species-dependent criteria such as spawning periods.

Table 4-1
Summary of Behavioral, Physiological, and Reproductive Temperature (°C) Thresholds for 69 Species of Freshwater Fishes Known to Occur in Ohio Rivers and Streams [1]. Species Considered Representative of the Ohio River Mainstem Fish Assemblage Appear in Boldface Type. Thresholds Denoted by an Asterisk were Estimated when Data was not Available. Each is Applicable to All Waters of the State Excepting Lake Erie and Adjacent Embayments. All Behavioral and Physiological Criteria Apply to the Summer Season (June 16 – September 15) and Assume Acclimation to Ambient Field Temperatures of 27-30°C

Species	Optimum or Final Preferendum	MWAT for Growth[1]	Upper Avoid Temp.	Upper Incipient Lethal Temp.[2]	Spawning Temperatures									
					Mar.	April	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
Shortnose Gar	33.0*	34.8	35.0	38.3*			19.0		30.0	30.0				
Longnose Gar	34.0*	35.4	35.0	38.3*			18.0		30.0					
Goldeye	28.0*	29.5	29.0	32.6*			12.8							
Mooneye	27.5*	29.2	29.0	32.6*										
Skipjack Herring	27.0*	29.4	30.5	34.1*										
Gizzard Shad	29.0	31.3	34.0	36.0			19.5	23.1/26.7	29.0					
Brown Trout	13.8	17.0	20.0	23.4								12.8	8.9/6.7	
Rainbow Trout	18.5	20.7	24.5	26.5			10.0	15.5						
Coho Salmon	16.6	19.4	23.5	25.0										
Chinook Salmon	17.3	19.9	24.1	25.0										
Brook Trout	18.0	20.4	23.0	25.3										
Smelt (spring)	8.3	13.7	16.0	24.4			14.5							
Redfin Pickerel	26.0	28.8	30.1*	34.3*										

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Species	Optimum or Final Preferendum	MWAT for Growth[1]	Upper Avoid Temp.	Upper Incipient Lethal Temp.[2]	Spawning Temperatures									
					Mar.	April	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
Northern Pike	26.1	28.4	29.8*	33.3	4.4/10.0	11.1/14.0	17.2/20.0							
Muskellunge	24.0	27.1	28.8*	33.3		12.8/15.0	17.2							
Bigmouth Buffalo	32 9*	34.1	35.0	38.3*		15.5	18.3	27.0						
Black Buffalo	32 9*	34.1	35.0	38.3*										
Smallmouth Buffalo	33.0	35.1	36.0	39.3 *			16.0/20.0	27.0/28.0						
Quillback	29.5	32.4	35.0	38.3 *		19.0		28.0	28.0/28.0	28.0/28.0	28.0			
River Carpsucker	32.0	34.1	35 9	38.9 *		18.0	21.0/24.0							
Highfin Carpsucker	30.2*	32.6	34.0	37.3*			19.0	28.0	28.0/28.0	28.0/28.0	28.0			
Golden Redhorse	26.0	27.9	28.5	31.8 *			15.0							
Shorthead Redhorse	27 9*	28.6	28.5	31.8 *		11.0	16.0							
Hog Sucker	27.2	29.5	31.7	34.2 *	15.6									
Common White Sucker	23.9	26.4	30.6	31.4			20.0	23.3						
Spotted Sucker	24.0	26.1	27.0	30.3 *		12.0/14.5	17.8/19.0							

Table 4-1
Summary of Behavioral, Physiological, and Reproductive Temperature (°C) Thresholds for 69 Species of Freshwater Fishes Known to Occur in Ohio Rivers and Streams [1]. Species Considered Representative of the Ohio River Mainstem Fish Assemblage Appear in Boldface Type. Thresholds Denoted by an Asterisk were Estimated when Data was not Available. Each is Applicable to All Waters of the State Excepting Lake Erie and Adjacent Embayments. All Behavioral and Physiological Criteria Apply to the Summer Season (June 16 – September 15) and Assume Acclimation to Ambient Field Temperatures of 27-30°C (Continued)

Species	Optimum or Final Preferendum	MWAT for Growth[1]	Upper Avoid Temp.	Upper Incipient Lethal Temp.[2]	Spawning Temperatures									
					Mar.	April	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
Common Carp	33.0	35.7	36.0	41.0			17.0/19.0	26.0	28.0/28.0	28.0/28.0				
Goldfish	28.1	30.9	33.0	36.6*			16.0		30.0/30.0	30.0/30.0				
Golden shiner	27.2	29.6	33.5	34.5										
Blacknose Dace	23.9	25.8	27.2	29.5			15.0	22.0						
Longnose Dace	27.3*	29.2	31.0	33.1			11.1	23.3						
Creek Chub	23.9	26.5	29.4	31.6		12.8			26.7					
Emerald Shiner	27.0*	29.0	31.1	33.0*			20.0	27.0	27.0/27.0	27.0				
Silver Shiner	23.1*	25.1	27.2	29.1*										
Rosyface Shiner	26.8	28.8	31.0	32.9			/17.8	21.1/26.7	28.9/28.9					
Striped Shiner	27.9*	29.8	31.2*	33.5										
Common Shiner	25.4*	27.3	28.7*	31.0			15.6		28.3/28.3					
Spottail Shiner	27.2	29.3	31.7	33.4*			20.0							

Table 4-1
Summary of Behavioral, Physiological, and Reproductive Temperature (°C) Thresholds for 69 Species of Freshwater Fishes Known to Occur in Ohio Rivers and Streams [1]. Species Considered Representative of the Ohio River Mainstem Fish Assemblage Appear in Boldface Type. Thresholds Denoted by an Asterisk were Estimated when Data was not Available. Each is Applicable to All Waters of the State Excepting Lake Erie and Adjacent Embayments. All Behavioral and Physiological Criteria Apply to the Summer Season (June 16 – September 15) and Assume Acclimation to Ambient Field Temperatures of 27-30°C (Continued)

Species	Optimum or Final Preferendum	MWAT for Growth[1]	Upper Avoid Temp.	Upper Incipient Lethal Temp.[2]	Spawning Temperatures									
					Mar.	April	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
Spotfin Shiner	29.7	31.9	35.0	36.3*				25.0	29.0/29.0	29.0				
Silverjaw Minnow	27.0*	29.1	31.1	33.4*										
Fathead Minnow	28.9	30.3	32.0	33.2			15.0	23.5/26.8						
Bluntnose Minnow	28.9	30.4	31.1	33.3			20.0/21.1	26.1						
Stoneroller	28.6	30.8	33.8	35.2*		14.4/18.3		24.0/27.0	27.0/27.0					
Channel Catfish	30.5	32.8	35.0	37.3				23.9/26.7	27.8/29.5					
Yellow Bullhead	28.0	30.6	31.0*	35.8*										
Brown Bullhead	31.1	33.2	36.1	37.5			21.0	25.0/27.0						
Flathead Catfish	32 9*	33.9	34.5	37.8*		22.0		30.0						
Mosquitofish	35.3	36.5	39.0	39.0										
White Bass	29.0	31.4	32.0	36.1*	12.0	14.4/17.8		24.0						
White Crappie	29.0	30.9	32.0	33.0		14.0/16.0	20.0	23.0						

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Species	Optimum or Final Preferendum	MWAT for Growth[1]	Upper Avoid Temp.	Upper Incipient Lethal Temp.[2]	Spawning Temperatures									
					Mar.	April	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
Black Crappie	28.3	29.9	30.2	33.0		19.0								
Rockbass	28.2	30.4	29.6	33.7*			15.6	21.1						
Smallmouth Bass	28.0	30.3	31.0	35.0		15.0/18.3	23.9							
Spotted Bass	28.5	31.1	31.0	35.1*		15.0/18.3	23.9							
Largemouth Bass	28.0	30.8	31.5	36.5		18.9	21.0/23.9							
Green Sunfish	30.6	33.7	33.0	40.0			20.0		28.0/28.0	28.0/28.0				
Bluegill	31.8	33.5	33.6	36.8			16.0/23.9	26.0/27.8		32.0/32.0				
Longear Sunfish	30.4*	33.0	34.1*	38.2			20.0	25.0						
Pumpkinseed	28.5	30.7	32.0	35.0			20.0		29.0	29.0/29.0				
Sauger	27 9*	28.1	29.0	30.4		3.9	9.0/12.0	15.0						
Walleye	25.0	27.2	29.5	31.6		5.6	8.9/11.1	15.0						
Yellow Perch	27.1	28.8	31.0	32.3		8.5	14	16.1						
Dusky Darter	25.0	27.8	30.8*	33.3*										
Greenside Darter	26.7*	30.6	35.0*	38.3*				18.3						

Table 4-1
Summary of Behavioral, Physiological, and Reproductive Temperature (°C) Thresholds for 69 Species of Freshwater Fishes Known to Occur in Ohio Rivers and Streams [1]. Species Considered Representative of the Ohio River Mainstem Fish Assemblage Appear in Boldface Type. Thresholds Denoted by an Asterisk were Estimated when Data was not Available. Each is Applicable to All Waters of the State Excepting Lake Erie and Adjacent Embayments. All Behavioral and Physiological Criteria Apply to the Summer Season (June 16 – September 15) and Assume Acclimation to Ambient Field Temperatures of 27-30°C (Continued)

Species	Optimum or Final Preferendum	MWAT for Growth[1]	Upper Avoid Temp.	Upper Incipient Lethal Temp.[2]	Spawning Temperatures									
					Mar.	April	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
Orangethroat Darter	26.0	28.8	31.8*	34.3*		13.0	25.0							
Fantail Darter	23.9	26.4	27.2	31.4*		18.9		24.4						
Freshwater Drum	29 9*	30.9	31.5	34.8*		/18		24.5/						
Mottled Sculpin	16.5	19.3	23.3*	25.0*			5.0	16.1						

1 - MWAT for growth calculated as: optimum + 0.333 (UUILT – optimum; Brungs and Jones 1976).

2 - Upper Lethal Temperature; 50% survival at 27-30°C acclimation.

* - Estimate based on conversion factors in Ohio EPA [1].

Derivation of Ohio River Temperature Criteria

The derivation of seasonal temperature criteria for the Ohio River mainstem included summer average and daily maximum values based on the output of the Fish Temperature Model, consideration of species-specific spawning thresholds compiled in Ohio EPA [1], and consistency with the historical ambient temperature record. Twenty-five (25) species were considered representative of the Ohio River mainstem fish assemblage (Table 4-1). The list is most relevant to the upper and middle portions of the Ohio River mainstem as Trautman (1957) was the principle source of distribution data used when the list was developed by Ohio EPA [1].

Summer Average and Maximum Criteria

Summer average and maximum criteria were calculated in accordance with the outputs of the Fish Temperature Model (Table 4-2). These apply during the defined summer period of June 16-September 15 as daily maxima and a *period* average. The rationale for the period average as opposed to a daily or weekly average is in recognition of the realities of electric power generation and the thermal requirements of fish. Neither is a “smooth” function with power generation being driven by periodic short-term peak demand and fish being able to avoid short-term exceedences of the long-term survival thresholds. Meeting the long-term period average also requires equivalent “cool down” periods when temperatures are below the survival thresholds and closer to the equally important physiological thresholds for growth and maintenance. The results of the Fish Temperature Model outputs for the Ohio River mainstem appear in Table 4-2 (summer season thresholds). The results indicate that an average temperature of 28.4°C (83.1°F) and a daily maximum of 30.4°C (86.7) will protect 100% of the representative species during the summer period. The period average of 28.4°C exceeds the upper avoidance temperature for 10-25% and the growth temperature for 10-25% of the representative species. Sixteen (16) species are considered to be either commercially or recreationally important. Of these, the 28.4°C average exceeds the growth temperatures for one species. No rare, threatened, or endangered species are among the representative fish species chosen by Ohio EPA for this analysis.

Seasonal Average and Daily Maximum Criteria

Establishing seasonal temperature criteria includes using not only the results in Table 4-2 for the summer period (June 16-September 15), but additional information for the remaining months (Table 4-3). These are set primarily in accordance with the historical ambient record, but also include an assessment of any exceedences of spawning temperature thresholds for each representative fish species (Table 4-2). Temperature duration analyses performed by USGS were used to determine the historical seasonal average and daily maximum temperatures. Averages were computed by averaging the daily maximums. Daily maximum temperatures were determined by examining the period of record and selecting the highest value that occurred at least three times during any one year and/or at least 10 times in a 10-year dataset. Any decimals derived from arithmetic transformation from degrees Centigrade to degrees Fahrenheit were rounded to the next highest whole number. Casual observations of summaries of the highest average and daily maximum ambient temperature record for any one year showed that the bi-weekly or monthly average was usually 6-10°F lower than the daily maximum.

Table 4-2
Temperatures at which 100%, 90%, 75%, and 50% of the Representative Fish Species for the Ohio River Mainstem are Within each of Five Thermal Tolerance Categories During the Summer Season Index Period (June 16–September 15). The Long-Term and Short-Term Survival Temperatures in °C (°F) Represent Summer Season Average and Maxima

Thermal Category	Proportion of Representative Fish Species			
	100%	90%	75%	50%
Optimum	26.0 (78.8)	27.0 (80.6)	27.5 (81.5)	29.0 (84.2)
Growth (MWAT)	27.9 (82.2)	28.1 (82.6)	29.5 (85.1)	30.9 (87.6)
Avoidance (UAT)	28.5 (83.3)	28.5 (83.3)	30.2 (86.4)	31.5 (88.7)
Survival (Long-term)	28.4 (83.1)	29.8 (85.6)	31.0 (87.8)	33.1 (91.6)
Survival (Short-term)	30.4 (86.7)	31.8 (89.2)	33.0 (91.4)	35.1 (95.2)

Source: Ohio EPA [1]

Table 4-3
Seasonal Monthly/Bi-Monthly Average and Daily Maximum Temperature Criteria (°F) for the Mainstem Ohio River as Originally Derived by Ohio EPA [1]

Month – Inclusive Dates	Monthly/Bimonthly Average	Daily Maximum	Criteria Rationale
January 1-31 February 1-29 March 1-15	45 45 51	50 50 56	Average and daily maximum criteria based on the historical temperature record at the Wheeling, Willow Island, and New Haven monitoring locations.
March 16-31	54	59	White bass initiate spawning.
April 1-15	58	64	White bass spawning; exceeds sauger spawning.
April 16-30	64	69	White bass, river carpsucker, smallmouth bass spawning.
May 1-15	68	73	Gizzard shad, river carpsucker, emerald shiner spawning; exceeds highfin carpsucker, sauger spawning criteria.
May 16-31	75	80	Smallmouth bass, river carpsucker spawning; exceeds sauger spawning criteria.
June 1-15	80	85	Exceeds spawning criteria for all species.
June 16-30 July 1-31 August 1-31 September 1-15	83 84 84 84	87 89 89 89	Long-term survival threshold exceeded by average July 1–September 15; short-term survival exceeded July1–August 31; growth of sport/commercial species exceeded July 1–September 15.
September 16-30 October 1-15 October 16-31 November 1-30 December 1-31	82 77 72 67 52	86 82 77 72 57	Average and daily maximum criteria based on the historical temperature record at the Wheeling, Willow Island, and New Haven monitoring locations.

Review of ORSANCO Temperature Criteria

Ohio EPA used the same approach described here to derive temperature criteria for the inland rivers and streams of Ohio and the near shore and open waters of Lake Erie. These were adopted in the Ohio WQS (Ohio Administrative Code chapter 3745-1) in 1978 and remain unchanged today. ORSANCO adopted the temperature criteria shown in Table 4-3 in 1984 for the Ohio River mainstem using the criteria originally developed by Ohio EPA [1]. In 1995 questions were raised about the relevancy of those criteria, specifically regarding the age of the underlying database and the availability of a significant body of more recent thermal effects literature. To this end, ORSANCO commissioned a review of the existing methodology and the underlying thermal effects database. This study is currently in progress and the results will be forthcoming in 2004. In addition to updating the available thermal effects literature database, an effort will be made to characterize other criteria derivation methodologies that have emerged during the past 25 years. We expect that the products of this review will be useful elsewhere as thermal effects assessments under the Clean Water Act re-emerge, after a nearly 25 year period of comparative dormancy, as a priority for U.S. EPA and the states. We also expect that it will be relevant to other issues, particularly those that pertain to TMDLs and the potential for changing ambient conditions related to climate change assessment and research.

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5

OVERVIEW OF CWA SECTION 316(A) EVALUATIONS OF POWER PLANTS WITH THERMAL DISCHARGES IN MARYLAND

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Abstract

As an EPA-delegated state, in the late 1970s Maryland developed and implemented regulations of thermal discharges and mixing zones in accordance with EPA guidance on implementation of Clean Water Act Section 316a provided at that time. Maryland regulations (Maryland Code of Maryland Regulations (COMAR) 26.08.03.03) established procedures for determining thermal impacts to biota in the receiving water relative to determination of necessary changes in facility processes or operations to minimize these impacts. Maryland has applied these regulations to all power plants in Maryland with thermal discharges, including facilities located on both freshwater and estuarine waters. Over the past 30 years, Maryland's Power Plant Research Program (PPRP) participated in and/or conducted studies of a wide range of thermal impacts in various habitats. These evaluations resulted in a range of determinations, from a decision that the existing discharges met the mixing zone limits to requiring further studies, e.g. long-term fishery studies at Chalk Point and Dickerson power plants. These studies, some lasting over 20 years, ultimately showed no long-term impact from the thermal discharges.

Introduction

Maryland facilities with thermal discharges are regulated by the Maryland Department of the Environment (MDE), the state agency with authority and responsibility for NPDES permitting. Maryland regulations relating to thermal discharges were developed based on EPA guidance on implementation of Clean Water Act Section 316a when that legislation was enacted, and are documented in Maryland Code of Maryland Regulations (COMAR) 26.08.03.03 (available online at http://www.dsd.state.md.us/comar/26/26.08.03.03.htm). These regulations address thermal impacts and mixing zone specifications for tidal and non-tidal waters of the state.

While MDE is responsible for regulation of thermal discharges, a sister agency, the Maryland Department of Natural Resources (MdDNR), provides the technical support employed to address these issues at power plants through its Power Plant Research Program (PPRP).

PPRP was established in 1971 to ensure that Maryland meets its electricity demands at reasonable costs while protecting the State's valuable natural resources. It provides a continuing program for evaluating electric generation issues and recommending responsible, long-term solutions. The Maryland legislature created the Power Plant Siting Program, precursor to the current PPRP, in 1971 as a result of extensive public debate regarding the potential effects on the Chesapeake Bay from the Calvert Cliffs Nuclear Power Plant. Calvert Cliffs was a source of concern because the plant uses a once-through cooling system that withdraws over 3.5 billion gallons of water per day from the Bay and discharges the water back to the Bay with a temperature elevation of about 12°F. The controversy over potential environmental impacts during the licensing of Calvert Cliffs prompted the creation of PPRP to ensure a comprehensive, technically based evaluation and resolution of environmental and economic issues before decisions were made regarding whether and where to build other generating facilities. Today, PPRP maintains this role by providing a comprehensive set of technically based licensing recommendations for proposed generating facilities. PPRP also conducts research on power plant impacts to the Chesapeake Bay, one of Maryland's greatest natural resources, and provides technical support to MDE regarding all power plant NPDES permits and variances associated with those permits. In addition to surface water concerns, PPRP's evaluations consider impacts to Maryland's ground water, air, land, and human resources. PPRP examines all of these areas in its review of effects on Maryland's natural resources, especially the Chesapeake Bay and its ecosystems.

PPRP operates with a small administrative and technical staff, supported by "integrator contractors" with special expertise in engineering, economics, biology/ecology, and atmospheric sciences. The program is funded from an Environmental Trust Fund that is maintained through a surcharge on users of electricity. The surcharge amounts to about 20 cents per month for average residential customers, but has provided a relatively stable source of funding to address the State's power plant assessment needs for nearly three decades. The manner in which PPRP carries out its responsibilities with regard to thermal discharge assessments are varied and customized to issues and circumstances specific to individual facilities and impacts. As a result of review of a permit or variance application from a given facility, PPRP may recommend studies be performed by the applicant. In such instances, PPRP's integrator contractor will be assigned responsibility for technical reviews of applicant's study plans and the findings of the studies. A final review of findings would be prepared for PPRP, and upon its review and concurrence would then be incorporated into recommendations from PPRP to MDE concerning disposition of the applicant's application. In cases where an issue may be relatively generic and findings may be relevant to broader statewide issues, PPRP may develop cooperative studies with an applicant, with PPRP contractors working with the applicant and their consultants to develop and implement studies. In cases where potential impacts are of concern, or where the efficacy of new technologies may be of interest, PPRP may conduct independent studies. Since inception of the program, PPRP has carried out all of these modes of study at all power plants in Maryland with regard to cooling water discharge impacts. Findings from a number of these studies are the basis for the remainder of this presentation and for the State's perspective on thermal impact assessment methodologies, significance and solutions.

Maryland Thermal Regulations

COMAR section 26.08.03.03 describes the factors, criteria, and standards for thermal effluent limitations, including definitions of regulatory mixing zones that apply to cooling water discharges from power plants and other large industrial facilities. Dischargers unable to meet mixing zone criteria can request alternative effluent limitations (AELs) which "assure the protection and propagation of a balanced, indigenous community [BIC] of shellfish, fish and wildlife in and on the body of water into which the discharge is made." In making such a request, dischargers are required to show that the thermal discharge limitations that would otherwise apply to them are more stringent than necessary to protect the BIC. The regulations also require AELs to consider: 1) cumulative impacts of the thermal discharge together with all other significant impacts on the species affected, including impingement and entrainment impacts; 2) a significant increase in abundance or distribution of any species considered to be nuisance species; 3) a significant change in biological productivity; 4) a significant elimination or impairment of economic or recreational resources; and 5) a significant reduction in the successful completion of the life cycle of Representative Important Species (RIS) (defined according to COMAR 26.08.03.04).

Existing dischargers at the time the regulations were issued (1974), were allowed to base their demonstration of AELs on the absence of prior appreciable harm instead of predictive studies. These demonstrations had to show that: 1) appreciable harm has not resulted from the thermal component of the discharge, taking into account the interaction of the thermal component with other pollutants and the additive effect of other thermal sources, to a BIC of shellfish, fish and wildlife in and on the body of water into which the discharge is made; or, 2) despite the occurrence of the previous harm, the desired AELs, or appropriate modifications to them, will nevertheless assure the protection and propagation of a BIC of shellfish, fish and wildlife in and on the body of water into which the discharge is made.

In determining whether prior appreciable harm has occurred, MDE is to consider the length of time an applicant has been discharging, and the nature of the discharge. If the discharger fails to demonstrate that existing facilities, or AELs together with all other impacts, will assure the protection and propagation of a BIC of shellfish, fish, other aquatic life, or wildlife in and on the receiving water, then the discharger is to make changes in the facility processes or operations, or both, sufficient to assure the protection and propagation of a balanced indigenous population of shellfish, fish, other aquatic life, or wildlife in and on the receiving water.

Mixing Zone Regulations

Maryland's thermal mixing zone regulations are diagrammed in summary form in Figure 5-1. There are 3 sets of mixing zone definitions laid out in the first part of the regulations (paragraph C, numbers 1, 2, 3): 1) a 50 foot mixing zone, meant to screen out small dischargers from further analysis; 2) a case-by-case mixing zone which may be requested when the detailed analysis required for tidal and non-tidal waters would not be applicable for some reason; and 3) compliance with maximum thermal limits and with specific mixing zone sizes depending on the type of receiving water. The maximum thermal limit criteria vary with the Use type definition as listed in COMAR 26.08.02.02B; however, all existing and proposed facilities in the state are located on waters defined as Use I or II, for which the thermal limit is 90ºF (32ºC). [The basis for selection of this value is not known to us; however, the 1974 draft EPA 316(a) technical guidance lists that value as a short-term maximum temperature for Bluegill survival for June through September, and 32.2°C as an allowable summer maximum for tropical regions and for the east coast of the U.S. as far north as Cape Hatteras, NC.] If this criterion is not met, regardless of other aspects of the mixing zone criteria, AELs would have to be requested; to our knowledge, no discharger has applied for AELs based solely on this criterion.

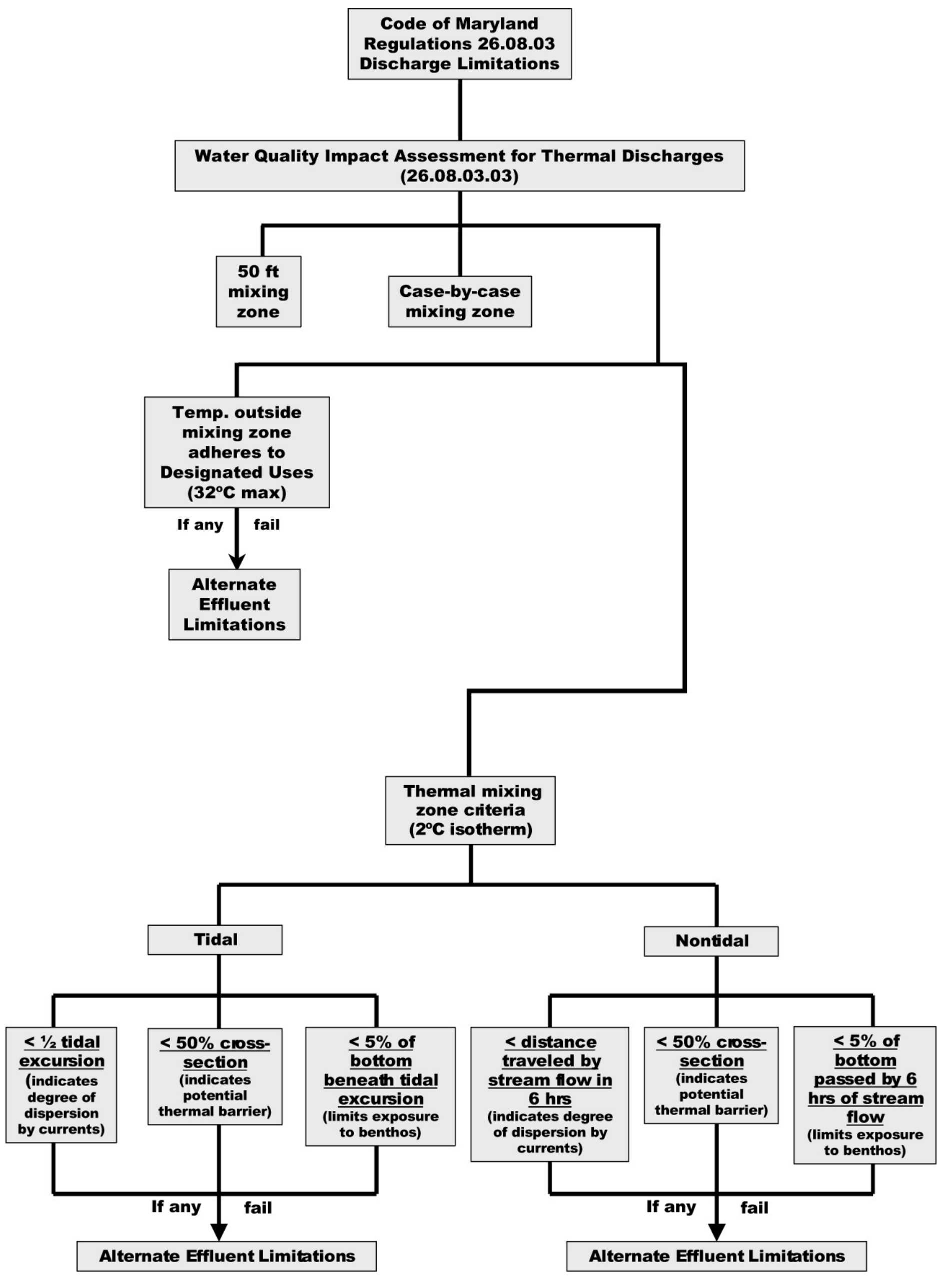

Figure 5-1
Diagram of Regulations for Thermal Discharges in the State of Maryland

Dischargers whose thermal plumes do not meet the 50-foot mixing zone limit are then required to evaluate their facility for compliance with specific regulatory size limits as summarized in the receiving water body. For tidal and non-tidal waters, the 24-hour average of the 2°C above ambient isotherm may not exceed 50 percent of the accessible cross section of the receiving water body. The third mixing zone limit is intended to limit exposure of bottom dwelling organisms in the receiving water body. For tidal waters, the 24-hour average of the bottom touched by waters heated 2°C or more above ambient isotherm may not exceed 5 percent of the bottom beneath the ebb tidal excursion multiplied by the width of the receiving water body. For nontidal waters, the same criterion applies except that the bottom area is defined by the stream bottom passed over by the stream flowing for 6 hours (as measured during critical periods).

Power Plants in Maryland

Figure 5-2 shows the locations of power plants in Maryland; those with once-through cooling are highlighted. Table 5-1 lists facilities in the state for which 316(a) studies were conducted and whether these facilities passed or failed the mixing zone criteria. (One of these, Westport, has subsequently retired the once-through cooling portion of the facility.) In summary, there were 5 facilities that passed all of the thermal mixing zone criteria, 4 facilities which failed, and 2 which failed under some flow conditions (both riverine facilities). One of the facilities (Wagner) which failed, subsequently applied for and ultimately received a case-by-case mixing zone, since there is an unusual flow pattern in its receiving water (Baltimore Harbor) which precludes easily calculating the standard mixing zone criteria.

Maryland Case Studies

In the remainder of this paper, we describe three mixing zone case studies of Maryland facilities in a variety of tidal and non-tidal waters and results of additional studies that were required to support alternate effluent limitations. These case studies were selected to illustrate a variety of facilities across the range of habitats in the state and how the mixing zone regulations applied to them. Calvert Cliffs was selected as a facility that passed the criteria and is located on the mainstem of the Chesapeake Bay, a large estuarine water body. Chalk Point was chosen as an estuarine facility on a relative small waterbody, the Patuxent River estuary, and does not pass the mixing zone criteria. Dickerson was chosen as a freshwater riverine facility on the Potomac River and does not pass mixing zone criteria under some flow conditions.

Calvert Cliffs Nuclear Power Station

Calvert Cliffs is owned by Constellation Nuclear, a member of Constellation Power Source, Inc. (formerly BGE). Maryland's only nuclear power plant, it is located on the Chesapeake Bay mainstem in Calvert County. It has generating capacity of 1,675 MW, and employs a once-through cooling system utilizing 3600 million gallons per day (mgd). It has a shoreline intake embayment with curtain wall that extends 8.5 m below the surface, and a high velocity discharge orifice which is 4 meters high, 3 meters deep and extends 268 meters offshore in the main channel of the Bay. Units 1 and 2 began operating in May 1975 and April 1977, respectively.

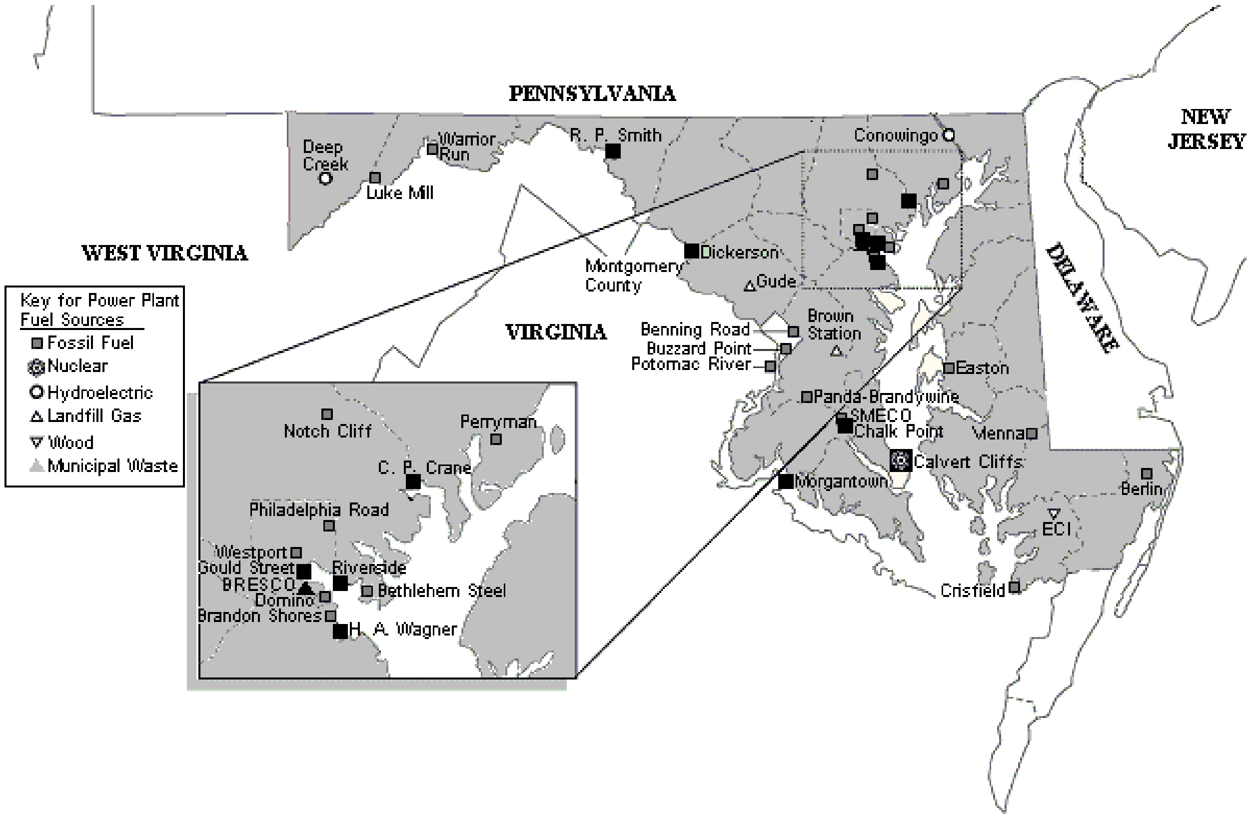

Figure 5-2
Locations of Power Plants in Maryland. Once-through Cooling Power Plants Indicated by Black Squares

Table 5-1
Status of Power Plants Under Maryland Thermal Mixing Zone Criteria

Plant	Mixing Zone Regulatory Status	Water Body
BRESCO	Fails	Patapsco (Baltimore Harbor)
Calvert Cliffs	Passes	Chesapeake Bay mainstem
Chalk Point	Fails	Patuxent River estuary
Crane	Fails	Gunpowder River (tidal)
Dickerson	Fails*	Potomac River
Gould Street	Passes	Patapsco (Baltimore Harbor)
Morgantown	Passes	Potomac River estuary
Riverside	Passes	Patapsco (Baltimore Harbor)
R.P. Smith	Fails*	Potomac River
Wagner	Fails	Patapsco (Baltimore Harbor)
Westport	Passes	Patapsco (Baltimore Harbor)

* under some flow conditions

Because of its size and the extent of controversy surroundings its placement and construction, Calvert Cliffs was the subject of intense study. Utility contractors conducted a wide range of intense environmental studies to satisfy Nuclear Regulatory Commission license technical specifications. These utility studies were augmented by extensive PPRP-funded studies. All of these studies and their findings are described in detail in [1], which summarized PPRP's conclusions regarding biological impacts of Calvert Cliffs.

Figure 5-3 illustrates a plan view of the Calvert Cliffs discharge, showing an example of a thermal plume from one of the original studies as described in [2] and [3]. The figure also illustrates the surface dimensions of two of the mixing zone criteria in relation to the point of discharge and a sample discharge plume. These plume dimensions are based on estimates made in [4]. The figure shows that the discharge plume is well within the regulatory limits for the maximum radial extent and bottom area. Figure 5-4 illustrates a cross-section of the Chesapeake Bay in the vicinity of the Calvert Cliffs discharge, along with the allowable limit (50% of the cross-section) and the estimated maximum distance that the plume extended. This figure also shows that the discharge plume is well within regulatory limits, not an unexpected result since the discharge is located in a large open waterbody with plenty of room for dilution of the plume without impacting a large area.

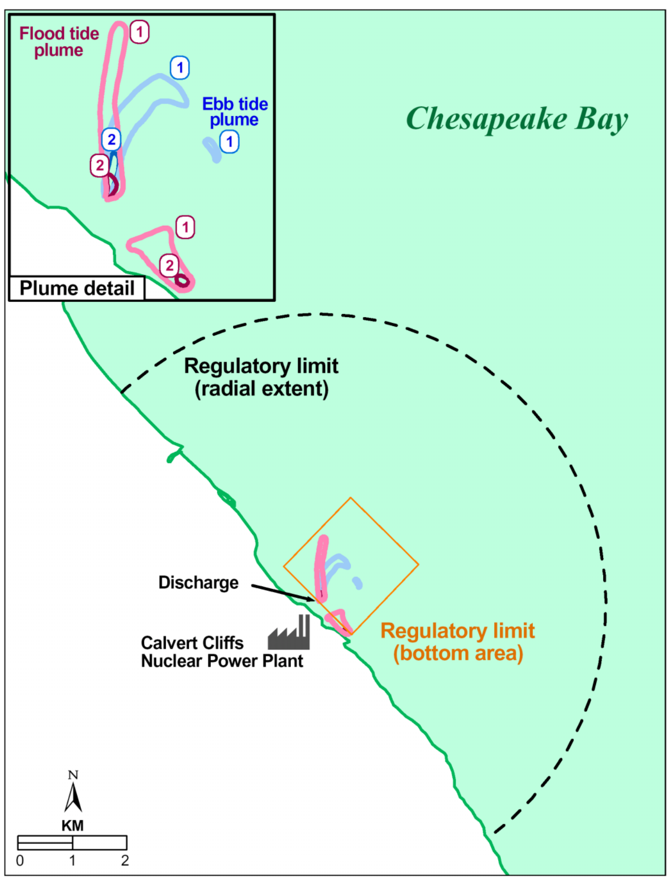

Figure 5-3
Limits of the Regulatory Mixing Zone (Radial Extent and Bottom Area) in the Vicinity of the Calvert Cliffs NPP Discharge in Comparison with Sample Flood Tide and Ebb Tide Thermal Plumes

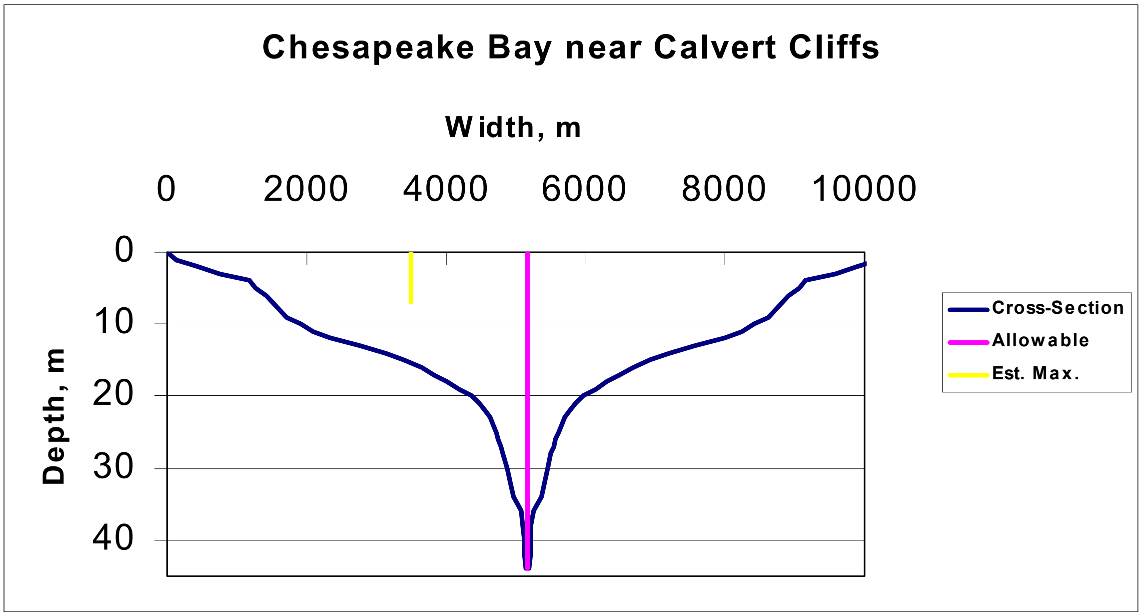

Figure 5-4
Limits of the Regulatory Mixing Zone (Cross-Sectional Area) in the Vicinity of the Calvert Cliffs NPP Discharge in Comparison with an Estimate of the Maximum Plume Extent

Table 5-2 summarizes the results illustrated in the figures, providing a list of allowed dimensions for each of the three mixing zone criteria, in comparison with estimated actual dimensions of the thermal plume. The ratios of actual to allowed dimensions are all well less than 100%, indicating that the mixing zone criteria are easily passed. Thus, no further 316(a) studies were required to be performed for this facility.

Table 5-2
Calvert Cliffs Nuclear Power Plant Mixing Zone Dimensions and Compliance with Maryland Regulations

Mixing Zone Specification	Allowed Dimensions	Estimate of Actual Dimensions	Ratio of Actual to Allowed Dimension
Maximum radial extent of 2EC-above ambient isotherm, 24-hour average (km)	5.3	1.8	34%
2EC-above ambient isotherm thermal barrier, 24-hr average (% of cross-section) (km)	9.1–14.3	3.5	25–38%
Area of bottom touched by waters heated 2EC or more above ambient (km^2)	3.1	.34	11%

Chalk Point Steam Electric Station

The Chalk Point Steam Electric Station, owned by Mirant Energy (formerly PEPCO), is located on the estuarine portion of the Patuxent River in Prince George's County. It is the largest generating facility in Maryland, with a total generation capacity of 2,415 MW provided by a mix of oil, coal and gas generating facilities. Units 1 and 2 utilize a once-through cooling system, withdrawing a maximum of 360 mgd per unit and discharging the heated water into the Patuxent River. Units 3 and 4 have closed-cycle cooling, using cooling towers and re-circulating water at a rate of 374 mgd per unit, with make-up and blow-down taken from and discharged into the intake and discharge streams of the once-through cooling system. Seven combustion turbine generators are also located on the site. The plant has dredged intake and discharge canals, as seen in Figure 5-5. One feature of the cooling water system to note in Figure 5-5 is the location of what are termed auxiliary cooling pumps. These pumps shunted water from the intake canal directly to the discharge canal as a means of ensuring compliance with a 100°F maximum temperature of waters discharged to the Patuxent River.

Figure 5-6 illustrates a plan view of the Patuxent River estuary in the vicinity of the Chalk Point discharge, showing an example of thermal plumes from one of the original studies as described in [2] and [5]. The figure also illustrates the surface dimensions of two of the mixing zone criteria in relation to the point of discharge and sample discharge plumes. These plume dimensions are based on estimates made in [6]. The figure shows that these sample discharge plumes are just within the regulatory limits for the maximum radial extent but well exceed the bottom area limit. (Note that the plumes shown here are not necessarily representative of the 24-hour average plume dimension but simply illustrate an example of a flood and ebb tide plume from one measurement). Figure 5-7 illustrates a cross-section of the Patuxent estuary in the vicinity of the Chalk Point discharge, along with the allowable limit (50% of the cross-section) and the estimated range that the plume extends. This figure also shows that the discharge plume always exceeded the regulatory limits, sometimes extending all the way to the opposite shore. Table 5-3 summarizes the results illustrated in the figures, providing a list of allowed dimensions for each of the three mixing zone criteria, in comparison with estimated actual dimensions of the thermal plume. The ratios of actual to allowed dimensions are usually all greater than 100%, indicating that the mixing zone criteria are not met for this facility.

Tempering pump entrainment–Auxiliary cooling water pumps, also called tempering pumps, were not screened. Thus, when operated, all ages and sizes of fish and crabs could be passed through the pumps and suffer physical damage from striking pump impellors and experiencing pressure changes. Large concentrations of fish and crabs were present in the intake canal, most likely because the intake flows and configuration of the canal were attractive to these organisms, which resulted in large numbers of organisms being entrained through the pumps. PPRP carried out a detailed assessment of the effectiveness of the tempering pumps for reducing plant-induced mortality of aquatic biota, using data collected by the facility owner and their contractors [7]. Several Representative Important Species (RIS) and dominant benthic and zooplankton species were used in the evaluation as indicators of overall system-wide responses. Expected mortality with and without auxiliary pump operation was estimated using thermal tolerance data available from the literature for blue crabs, white perch, striped bass, spot, *Macoma balthica* (a shellfish), and *Acartia tonsa* (a zooplanktor). PPRP concluded that the operation of the pumps increased plant-induced mortality of spot, white perch, striped bass, and zooplankton, but could reduce blue crab mortality slightly under some circumstances. *Macoma* mortality was largely unaffected by their operation. The overall conclusion was that cessation of use of the tempering pumps would result in a 50% decline in losses of fish and crabs from CWIS operations. A sensitivity analysis confirmed that the conclusions drawn were not significantly affected by uncertainties in the input data used. As a result of this evaluation, PPRP recommended to MDE that the Chalk Point NPDES permit be modified to eliminate the requirement for use of auxiliary pumps. Thermal criteria in the permit were later changed to a thermal loading cap rather than a specific discharge temperature cap.

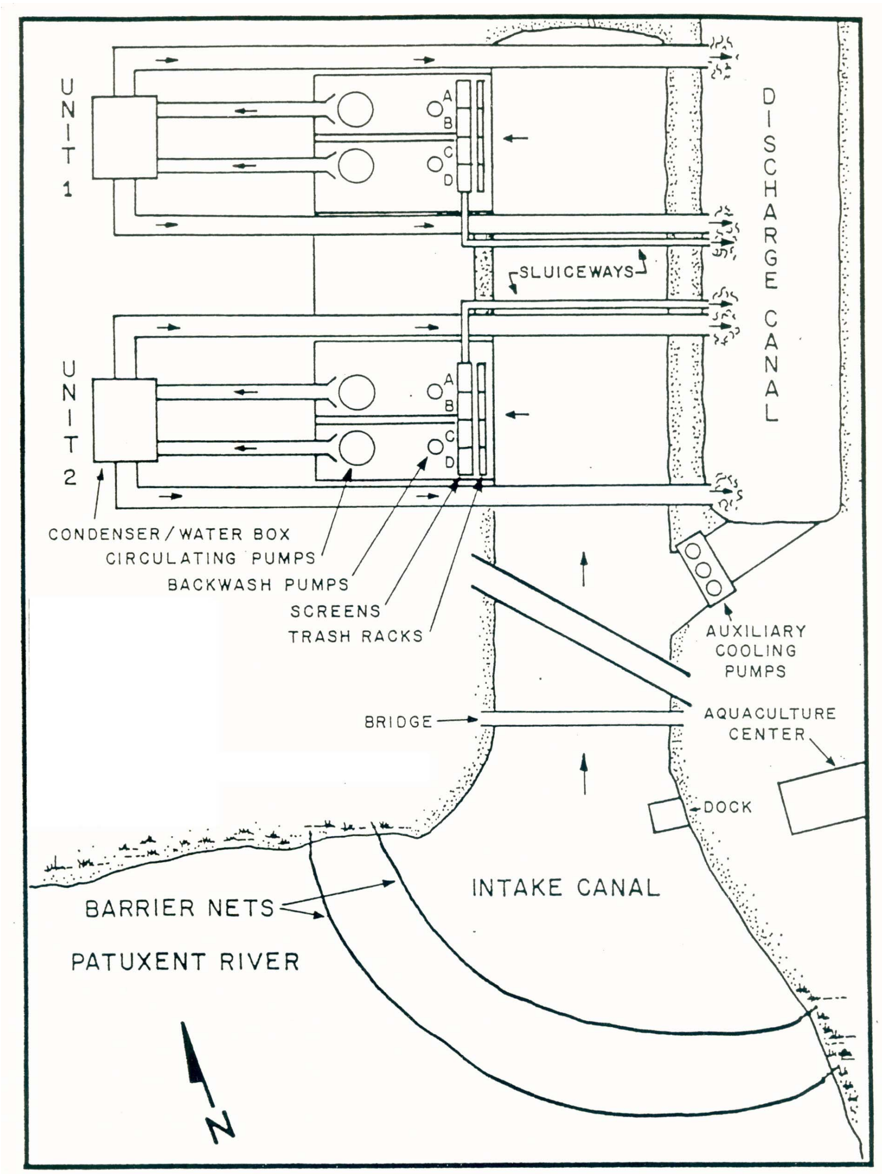

Figure 5-5
Chalk Point Steam Electric Station (SES) Intake and Discharge Canals Showing Points of Discharge from Units 1 and 2 and Auxiliary Cooling Pumps

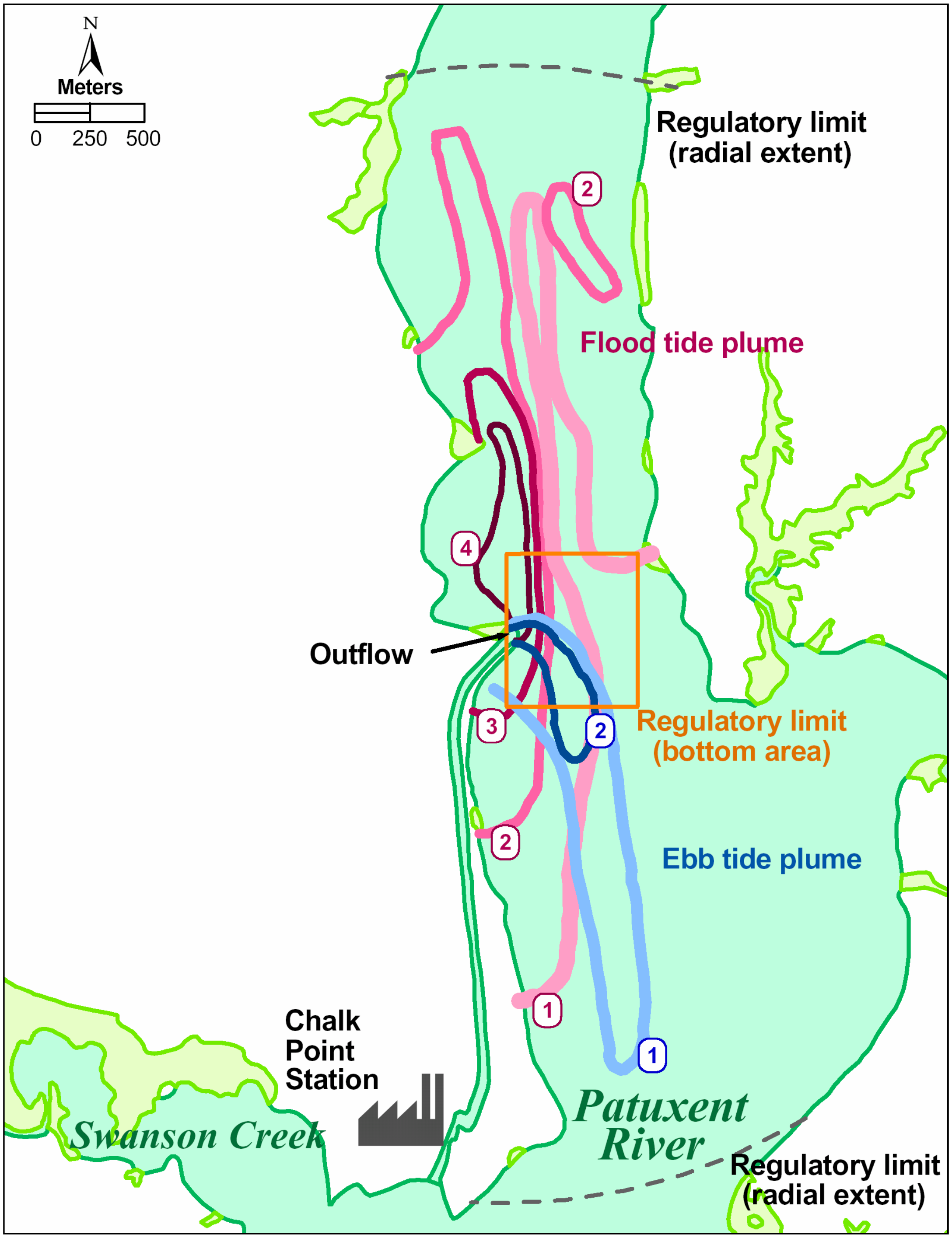

Figure 5-6
Limits of the Regulatory Mixing Zone (Radial Extent and Bottom Area) in the Vicinity of the Chalk Point SES Discharge in Comparison with Sample Flood Tide and Ebb Tide Thermal Plumes

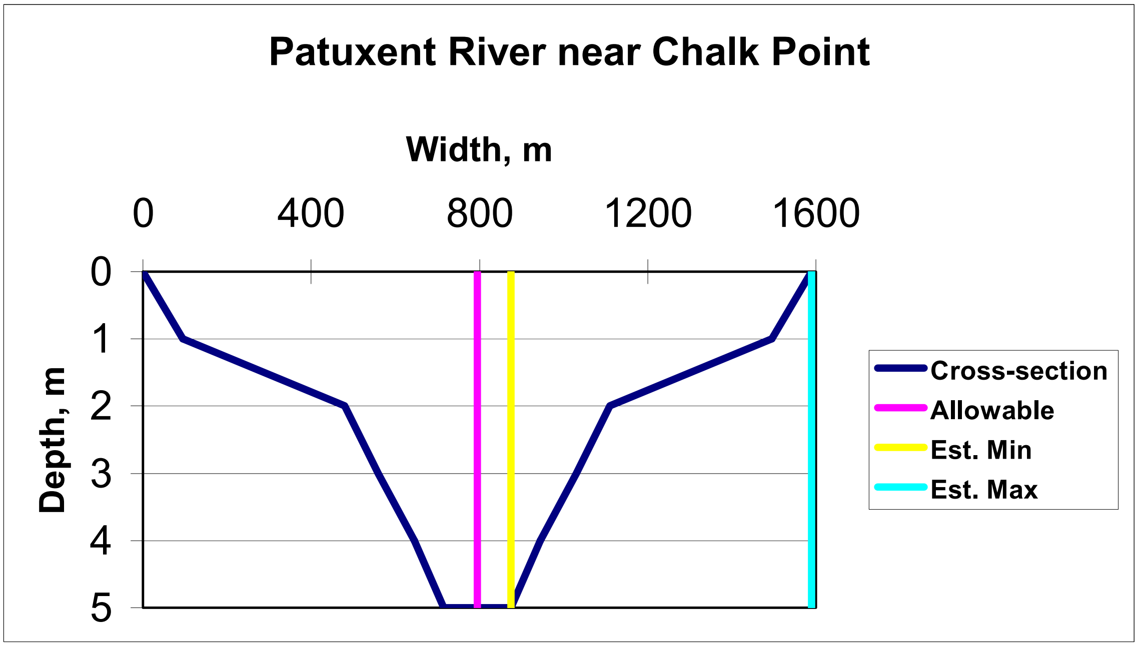

Figure 5-7
Limits of the Regulatory Mixing Zone (Cross-Sectional Area) in the Vicinity of the Chalk Point SES Discharge in Comparison with an Estimate of the Minimum and Maximum Plume Extent

Table 5-3
Chalk Point Steam Electric Station Mixing Zone Dimensions and Compliance with Maryland Regulations

Mixing Zone Specification	Allowed Dimensions	Estimate of Actual Dimensions	Ratio of Actual to Allowed Dimension
Maximum radial extent of 2EC-above ambient isotherm, 24-hour average (m)	2,500–2,650	2,500–4,600	94–184%
2EC-above ambient isotherm thermal barrier, 24-hr average (% of cross-section) (m)	50	55–100	110–200%
Area of bottom touched by waters heated 2EC or more above ambient (ha)	33–49	62–96	127–291%

As a result of failure to pass the thermal mixing zone criteria, Chalk Point was required to demonstrate that AELs should be granted and further studies were required on the thermal impacts from the discharge. Loos and Perry conducted a study to determine the abundance and species composition from 1991-2000 and compared results to a 1985-1990 study to indicate any thermal effects of discharges [8]. The study is based on a time series of fish abundance data by species from otter trawl samples, as well as data for chemical and physical parameters at fixed stations in the mainstem of the Patuxent estuary (n=22), the Chalk Point discharge canal (n=1), and in Swanson Creek (n=1). The sampling stations were generally allocated along transects across the estuary, and covered the shoal and the channel in control and nearfield regions. Trawling at each station was conducted with the tide. The abundances of 13 common species and the ratio of abundance by nearfield and control regions were summarized in tables, and compared through visual inspection of box and whisker plots. Spatial distribution was evaluated from box and whisker plots based on monthly data with annual variation being removed. The species composition in the 1991-2000 period was compared to results from earlier studies to evaluate whether the fish community had changed or been negatively impacted as compared to the previous study period (1985-1990). The study concluded that the species composition of the river has remained constant between the two study periods, with the exception for baywide changes for some species (e.g. an increase in striped bass abundance resulting from stocking). AELs have been granted at each 5-year permit cycle as these long-term studies continued.

Dickerson Steam Electric Station

The Dickerson SES, located on the freshwater portion of the Potomac River in Montgomery County, is owned by Mirant Energy (formerly PEPCO). It has total generating capacity of 556 MW, and utilizes a once-through cooling system with a capacity of 400 mgd. As with all other power plants in Maryland, Dickerson was the subject of intensive PPRP study and evaluation, as is summarized in [9].

Figure 5-8 illustrates a plan view of the Potomac River in the vicinity of the Dickerson discharge, showing an example of thermal plumes from one of the original studies as described in [2] and [10]. The figure also illustrates the surface dimensions of one of the mixing zone criteria in relation to the point of discharge and a sample discharge plume. These plume dimensions are based on estimates made in [9]. The figure shows that these sample discharge plumes exceed the regulatory limits for the maximum downstream extent. (Note that the plumes shown here are not necessarily representative of the 24-hour average plume dimension but simply illustrate an example of a plume from one measurement). Figure 5-9 illustrates a cross-section of the Potomac River in the vicinity of the Dickerson discharge, along with the allowable limit (50% of the cross-section) and the estimated range that the plume extends. This figure also shows that the discharge plume always exceeded the regulatory limits. Table 5-4 summarizes the results illustrated in the figures, providing a list of allowed dimensions for each of the three mixing zone criteria, in comparison with estimated actual dimensions of the thermal plume. The ratios of actual to allowed dimensions are often greater than 100%, indicating that the mixing zone criteria are not met for this facility.

As a result of failure to pass the thermal mixing zone criteria, Dickerson was also required to demonstrate that AELs should be granted and further studies were required on the thermal impacts from the discharge. Two studies were recently concluded, one on the overall fishery in the receiving water near the discharge [11] and the other focusing on smallmouth bass [12].

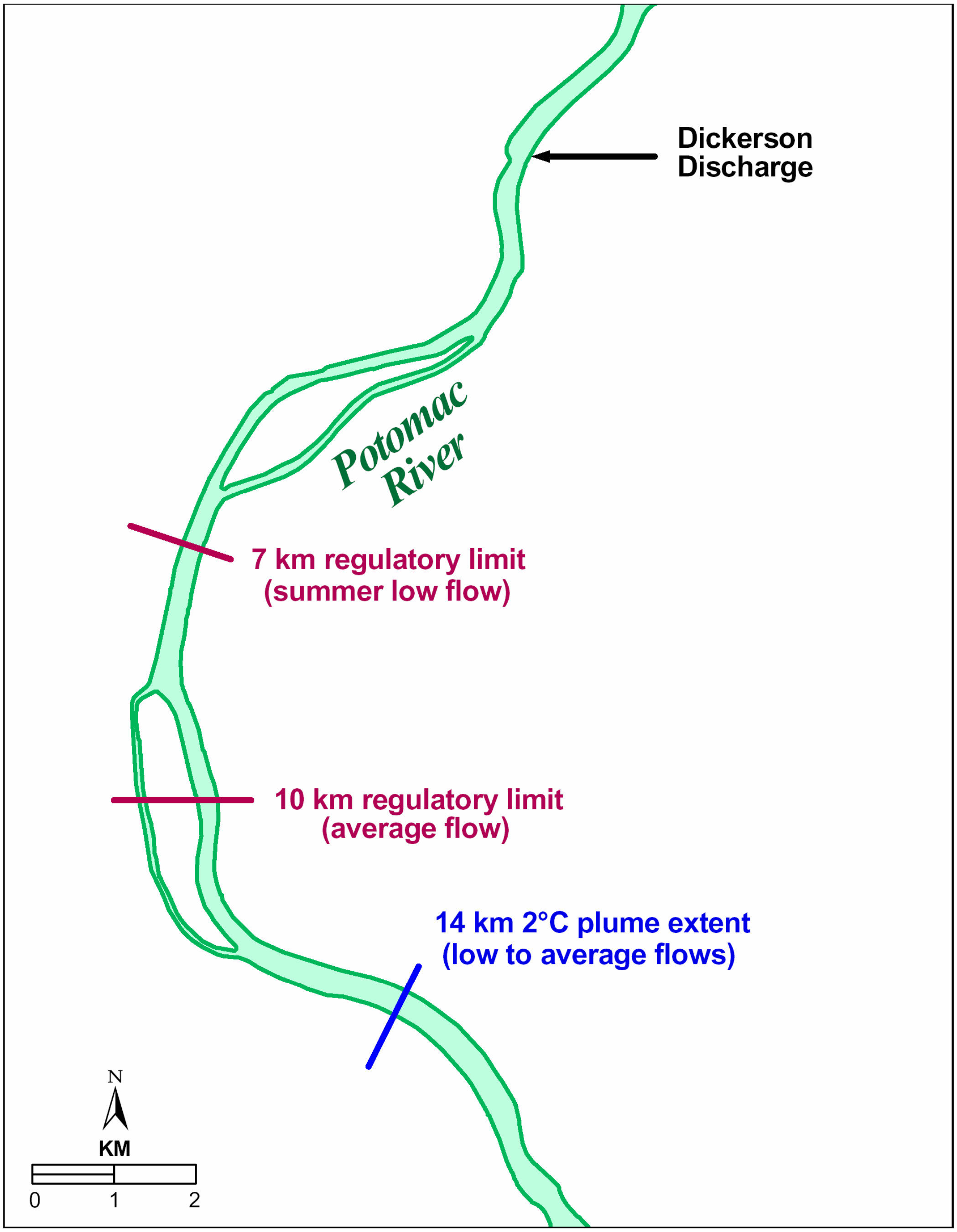

Figure 5-8
Limits of the Regulatory Mixing Zone (Downstream Extent) in the Vicinity of the Dickerson SES Discharge in Comparison with Sample Thermal Plumes for the Summer Low Flow and Average Flow Conditions

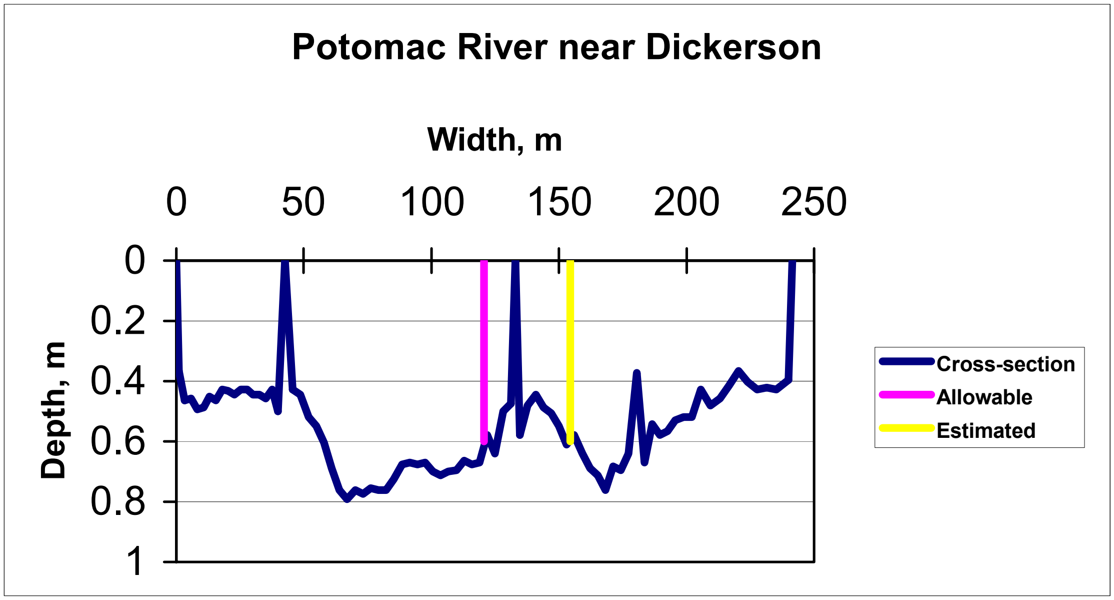

Figure 5-9
Limits of the Regulatory Mixing Zone (Cross-Sectional Area) in the Vicinity of the Dickerson SES Discharge in Comparison with an Estimate of the Maximum Plume Extent. River Islands are Present at Two Locations as Shown Where the Bottom Profile Reaches the Surface

Table 5-4
Dickerson Steam Electric Station Mixing Zone Dimensions and Compliance with Maryland Regulations (Low to High Summer Flows)

Mixing Zone Specification	Allowed Dimensions	Estimate of Actual Dimensions	Ratio of Actual to Allowed Dimension
Maximum downstream extent of 2EC-above ambient isotherm, 6-hour travel time (km)	7.3–19.6	2.5–14	34–192%
2EC-above ambient isotherm thermal barrier, average low-flow (% of cross-section) (m)	140–155	192 (maximum extent)	123–137%
Area of bottom touched by waters heated 2EC or more above ambient, 6-hour travel time (10^3 m^2)	110–295	45–1400	41–1,273%

The general study was based on a time series of fish abundance data from electrofishing samples at fixed stations around and within the Dickerson Station thermal influence. Electrofishing collections were made at 43 stations from 1979 to 1989, and at a subset of 14 of the original stations, plus one additional station, from 1990 onwards. The electrofishing was conducted in each season, with repeat sampling of stations within season when logistically feasible. Only two electrofishing collections were made during winter from 1990 to 2000. The abundance (log-transformed) of fish by species or functional groups at impacted and control sites was compared through exploratory graphical analysis. Abundance patterns and species composition were also compared to expected results based on published studies of fish distributions. Results indicated that species in the sunfish and catfish family are neutral or attracted to the thermal plume, while minnows, suckers, and darters have the strongest avoidance reaction. These results are in agreement with the literature. The study concluded that the heated discharges have only a minor seasonal effect on fish distributions, and that no adverse long-term impacts have occurred.

- The smallmouth bass study was based on length and scale/otolith samples collected from the Dickerson discharge and Point of Rocks (control site) in 1998 and 1999. A SAS clustering procedure was used to group the individuals into age classes based on scale/otolith readings.

Analysis of variance was conducted to test for differences in mean length at age and overall mean length between the control and impacted areas.

Smallmouth bass near the discharge was found to have significantly larger mean length across age groups than bass collected at Point of Rocks. The comparison of mean length by age group was inconclusive. The study concluded that the discharge does not have an adverse impact on the growth of smallmouth bass.

Conclusions

This brief overview provides several diverse examples of the process employed by Maryland in making power plant thermal mixing zone and impact determinations under Maryland's thermal regulations. Based on 30 years of PPRP experience, the major points we wish to convey include:

- All studies confirmed that thermal mixing zone criteria are protective of the biotic community in the vicinity of power plant thermal discharges, since these thermal criteria have been used in identifying facilities with a potential for impacts.
- Detailed assessments that were required to demonstrate AELs then served as a foundation for technically-based regulatory decisions.

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6
ASSESSING BIOLOGICAL RESPONSE OF FISH TO EXTREME TEMPERATURES: AN UPDATE ON THE SCIENCE

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Introduction

"The use of physiological criteria is justified by the premise that temperature limitations for water bodies should be designed to protect the desirable aquatic species found there. Thus, the physiological requirements of those species must be known and accommodated."

- C. C. Coutant [1]

Understanding the thermal requirements of fish has a long history in both the regulatory and scientific arenas. In this paper we review the history of environmental regulation related to thermal discharges, discuss the basics of thermal exposure and response by fish, describe standard methods that have been used to determine thermal tolerance, and discuss recent advances in thermal tolerance assessment.

Historical Perspective

Incorporation of biological responses to temperature into the criteria for environmentally benign location, design and operation of thermal power stations has been a progressive process over several decades. Independent science and applied research on temperature effects on aquatic life have been blended with concerns over thermal discharges and power station engineering to yield the regulatory and scientific framework we know now (and love?). This historical perspective sets the stage for current re-evaluation of recent approaches and emergence of some new enhancements and new directions.

The 1960s

The decade of the 1960s has been considered by many to be the decade when concern for the health of the environment became a national passion. At that time, "thermal pollution" became a "hot topic." Those concerned over pollution of rivers and lakes from all causes discovered that thermal power stations discharge large amounts of cooling water at elevated temperatures. Early scientific studies showed these discharges often had negative effects on aquatic life [2, 3, 4, 5, 6].

Concurrently, there were predictions that the thermal generating capacity in the U.S. would expand tremendously. A large amount of the added capacity was to be supplied by nuclear power stations that had lower heat efficiency (release more heat per amount of electricity generated) than conventional fossil-fueled plants [7]. Visions of rivers boiling away were raised. Immediately, the earlier work of academic scientists who studied the effects of temperature on fish and other aquatic life attained elevated importance [8, 9, 10, 11, 12]. Senator Edmond S. Muskie, a champion of environmental protection, organized widely publicized hearings on thermal pollution before the Subcommittee on Air and Water Pollution of the Committee on Public Works of the U.S. Senate in February 1968. The Federal Water Pollution Control Administration (precursor of the present Environmental Protection Agency; EPA) held two national symposia on thermal pollution in 1968 on engineering and biological aspects [7, 6]. Ironically, the world's largest thermal discharges were from the government's own cold-war plutonium-production reactor facilities at Hanford, Washington (Columbia River) and Savannah River, S. Carolina, where intensive thermal effects studies were under way [13].

The need for biological criteria to protect aquatic life from overly elevated temperatures was recognized early but not formalized in national criteria (the EPA "green book") until 1967 [4, 14, 15]. These criteria were fairly general and restricted thermal additions to amounts that would raise temperatures above ambient by only a few degrees (e.g., ~2°C or 5°F). Difficulties defining the environmental "ambient" temperature and the understanding that thermal requirements of various species differ led to realization that criteria for a water body needed to be more organism based and focused on the desirable species present. A rich literature had already accumulated that allowed some species-specific criteria to be developed. New thermal-effects literature began to be summarized annually in reviews for the Water Pollution Control Federation (published in the annual literature review issue), a series that continued until the mid 1980s.

The Early 1970s

Both the conceptual view of how aquatic organisms are exposed to elevated temperatures in power station discharges and the dose-response physiological framework for assessing biological effects from those exposures were clarified in early 1970s. The thermal pollution issue and approaches for resolving it with biological data were analyzed [16, 17]. The EPA commissioned the National Academies in early 1970s to develop water quality criteria for all water pollutants, including heat [18]. The academy report (the "blue book") summarized biological thermal-effects information current at the time and gave direction to new biological and physical studies as regulatory guidance for states.

These approaches became the modus operandi for evaluating thermal effects of nuclear power stations under the new National Environmental Policy Act of 1969. That Act and the environmental impact statements it mandated were implemented by the then Atomic Energy Commission (AEC) in the early 1970s. That was a time when the forecasts of massive nuclear power plant development were being realized.

In 1972, the landscape for evaluating thermal effects under water pollution control laws changed radically. The 1965 Water Pollution Control Act was amended (Clean Water Act) to focus on technological fixes for pollutants as well as water quality criteria. Cooling towers became the intended "best available technology" for treating heated discharges. Nonetheless, the Act included a Section 316(a) that allowed variances to these restrictions (both technology and water temperature criteria) if studies demonstrated that a "balanced indigenous population" (population equals community in this context) was supported in the vicinity of the thermal discharge. The EPA and the Nuclear Regulatory Commission (successor to the AEC) commissioned guidelines for conducting such demonstrations (culminating in the last "draft" guidelines in 1977, which is still in effect today). An important part of the guidelines was selection of "representative and important species" (RIS) whose biological responses to temperature would be indicative of a whole ecosystem in which they were found.

Decade of 1975 to 1985

The decade roughly 1975 to 1985 saw a flurry of laboratory research on biological effects of temperature and field studies at power stations leading to 316(a) demonstrations. The laboratory research concentrated on better defining the thermal requirements of the RIS. The number of publications reporting original dynamic temperature tolerance data for fishes took off in the 1970's. However, many of these studies were only reported in the gray literature. The demonstrations generally consisted of field research on aquatic community composition near power plants to document "no prior harm," augmented by laboratory-derived thermal tolerance limits of the RIS species. Although a number of power stations converted to cooling towers, the majority was able to successfully demonstrate lack of harm and was allowed the 316(a) variances by the responsible state agency.

Decade of 1985 to 1995

Relative quiet prevailed on the thermal effects scene at this time. The 316(a) field crews were disbanded and laboratory programs by industry and academia were shifted from thermal effects to other topics. Research publications continued, many formalizing earlier gray-literature reports into the peer-reviewed literature. Although the 316(a) demonstrations were to be revisited every 5 years, the variances were usually extended without debate until the mid 1990s.

1995 to Present

In the background of 316(a) activity related to thermal discharges there lurked another part of the Clean Water Act, Section 316(b) related to water intakes. This section required that the best technology available be used to minimize entrainment and impingement at the intakes. No formal guidelines for implementation were issued (actually, draft ones were remanded by the courts in the 1970s). A do-the-best-you-can attitude prevailed for intakes until the EPA was again taken to court in 1995. EPA was eventually required to fully implement a strict interpretation of the water-intake section. The 316(b) saga is another story for another day, except as it renewed interest in the thermal side.

Concurrent with the 316(b) intake discussions has been a renewed interest in 316(a) thermal issues. Regulators began to look at deferred thermal renewals as well as new intake-related regulatory actions. Because 316(a) requires a balanced indigenous community, the effects of intakes are built in, so to speak (ecologically, if not legally). Although research on biological thermal effects had slowed to a near stop in the late 1980s, some new work suggested value in looking at new or updated approaches. This is where we are today.

Thermal Exposure

Whether in laboratory experiments or natural settings, the amount of thermal stress experienced by a fish is a function of several factors in addition to the temperature to which it is exposed. Thermal stress accumulation is conceptually similar to a contaminant dose-response relationship and many of the underlying factors have a significant temporal component. For fish residing at a low or moderate temperature, the rate of change to a higher temperature and the duration of exposure are crucial factors in the level of stress experienced. In addition, the temperature regime to which a fish is acclimated for several days to weeks prior to an exposure of concern affects the degree of stress experienced. In temperate regions the relationship between acclimation and thermal tolerance mimic the natural seasonal variation in temperature. Typically fish in the northern hemisphere have greater tolerance for high temperatures in the summer and low temperatures in the winter because of acclimation to seasonal temperature regimes. Not as well understood is the relationship between thermal effects and large (i.e., 6-10°C) daily variations in temperature that are characteristic of some systems [19, 20].

In addition to temporal considerations there are also spatial aspects of the environment that affect exposure and should be considered. For example, plume dynamics at powerplant outfalls affect the spatial extent of exposure and the sharpness of the gradient from background temperatures to those at the mouth of the outfall pipe [17]. Typical 316(a) demonstrations often include the designation of a mixing zone for regulatory and compliance purposes. The mixing zone is generally defined as a limited area or volume of the receiving water where the initial dilution of a discharge is allowed to occur and such that 1) it does not impair the integrity of the water body as a whole; 2) there is no lethality to organisms passing through the mixing; and 3) there are no significant health risks, considering likely pathways of exposure [21].

Rivers typically have mixing zones that extend downstream from the facility outfall and may or may not extend across the entire river. Because of tidal flow in estuaries and bays, the designation of a mixing zone around an outfall must account for daily movements of the thermal plume. Any assessment of powerplant effects should also consider natural characteristics of the thermal environment such as vertical stratification in lakes and bays and longitudinal variation in rivers and streams.

Fish Responses to Stressful Temperatures

Thermal effects occur at several levels of organization from subcellular to community. In general there is a relationship between the response time to some environmental stressor at a particular level of biological organization and the ecological relevance of that response (Figure 6-1) [22]. We believe this also holds true for exposure to extreme temperatures. Short-time responses, such as at the biochemical or physiological levels, can have a significant impact at the individual level (i.e., mortality). However, the loss of an individual is usually not considered ecologically relevant. On the other hand, if a large number of individuals experience the same fate resulting in a population-level response, the ecological relevance would likely be significant. Although the first level of thermal effects assessment is often to understand the biological response, the ultimate question as to the severity of the impact is whether or not the impact has ecological relevance (i.e., long term effects on a population or community).

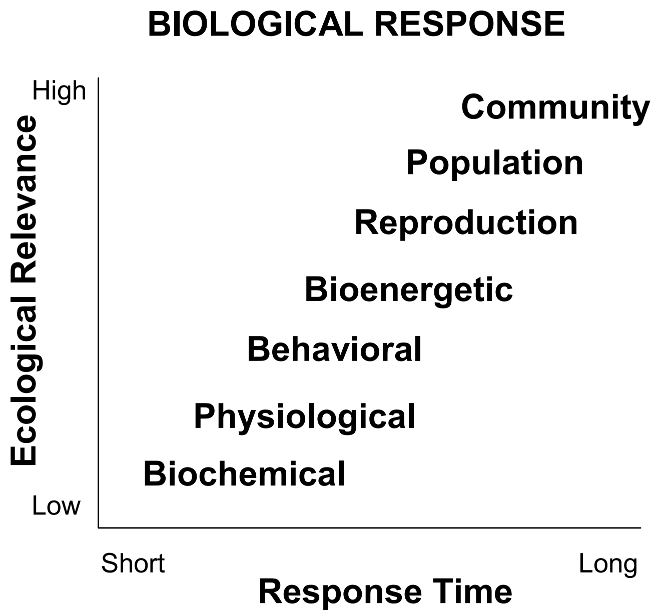

Figure 6-1
Generalized Relationship between Response Time and Ecological Relevance of Biological Response of Fish to Extreme Temperatures

Biochemical and physiological responses to extreme temperatures are often not obvious until a fish exhibits a loss of equilibrium or death. Biochemical responses include the development of alternate enzyme systems, protein denaturing, and changes in blood chemistry. Examples of physiological responses are changes in respiration, ventilation, and heart rate. Longer term physiological responses include reduced growth and delays in maturation and reproduction. Behavioral responses include thermoregulation (e.g., moving to areas with less stressful temperatures), reducing activity to minimize metabolic costs, and a cessation of feeding. The severity of these responses is ultimately a factor of the magnitude and duration of the thermal stress. Population- and community-level responses occur when the lower level responses are severe.

Thermal Effects Assessment

A variety of tools are available for assessing the impact of extreme hot and cold temperatures on fish. These include a mixture of laboratory experimentation, field observation, and mathematical models. In this paper we present several of these methods to illustrate the types of approaches available. We present these methods starting with those that assess biological response at the lowest level (i.e., biochemical and physiological), then those at a mid-level (i.e., behavioral and energetic), and lastly those at the highest level of response (i.e., population and community). As discussed above, there is a general relationship among the level of biological response, the response time, and the ecological relevance.

Assessment of Low-Level Biological Responses—Physiological/Biochemical

Researchers through the years have developed several laboratory methods that assess a fish's tolerance to extreme warm and cold temperatures. Although the exposure being tested usually results in significant biochemical and physiological responses, the response that is most easily measured is death or some near-death indicator, such as loss of equilibrium. Several types of thermal tolerance tests have been developed over the years, and two of the most popular are described here. For a more detailed discussion see a recent article by Beitinger et al. (2000) who provide an excellent review of thermal tolerance methodology and a summary of results for 116 species [23].

Incipient Lethal Temperature (ILT)—The ILT test measures the response of fish to an immediate change in temperature [8, 10]. All test fish are acclimated to a particular temperature. Subsets of fish are transferred without additional acclimation to several exposure temperatures (that is, a subset of fish for each exposure test temperature). After a set period (typically 24 h), the proportion of affected fish (typically measured as death or loss of equilibrium) is noted for each test temperature. The ILT is the exposure temperature that results in 50% mortality (or loss of equilibrium) over the test period. Both upper (UILT) and lower (LILT) lethal temperatures can be determined by this methodology.

Critical Thermal Methodology (CTM)—The CTM measures a fish's response to a rapid but not immediate change in temperature [24]. Experiments can be designed to test for a critical maximum (CTMax) or critical minimum (CTMin) temperature. Subsets of test fish are acclimated at different temperatures for several days or weeks. Temperature for each group is then raised (or lowered) at a constant rate (0.2-1.0°C/min typically) and the time at which each fish displays a loss of equilibrium is noted. At loss of equilibrium fish are typically placed in a recovery tank so that they can be used in future experiments or released to the wild. Some investigators use death or other endpoints instead of loss of equilibrium. The critical temperature, CTMax (or CTMin), for each acclimation group is calculated as the mean (or median) temperature to loss of equilibrium.

By combining CTMax and CTMin results or UILT and LILT results one can create thermal tolerance polygons (Figure 6-2) which describes a tolerance range of temperatures for any acclimation temperature. The two methods do not produce identical polygons as they are designed to measure different responses. A tolerance polygon defined in this way does not define the bounds of absolute thermal tolerance, however, and interpretation of the polygons needs to include a full understanding of the prior acclimation and rate of increase for specific experiments.

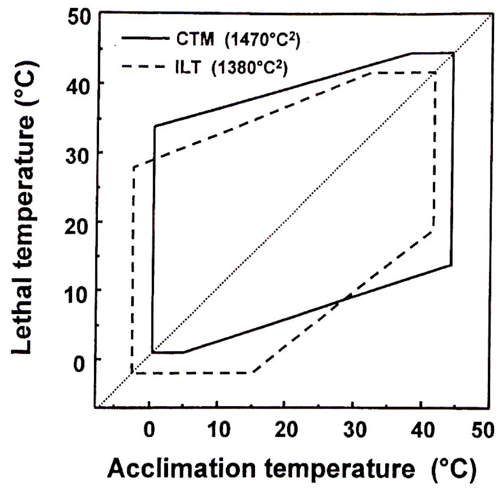

Figure 6-2
Temperature Tolerance Polygons Generated by the CTM and ILT Methods for the Sheepshead Minnow [23]

Results from ILT, CTM, and similar methods have been used to assess possible thermal effects at heated discharges. Thermal tolerance data for temperate bass (striped bass *Morone saxatilis* and white perch *M. americana*) were used to assess potential impacts at the thermal discharge at Salem nuclear plant on the Delaware Estuary (Figure 6-3) [25]. Upper limits of tolerance were compared to the maximum temperatures at the edge of the mixing zone. To account for seasonal differences, tolerance results were matched by their acclimation temperature to the ambient river temperature for different times of the year. Maximum temperatures at the edge of the mixing zone approached upper tolerance limits in the summer, but no effect was predicted as this is a time when neither species is in this part of the river.

Assessment of Mid-Level Biological Responses—Behavior/Growth/Reproduction

In addition to causing mortality, exposure to extreme temperatures can also have more subtle effects on behavior, growth, and reproduction. Measures of changes in behavior or reduction in growth or reproduction provide other means to assess thermal exposure.

Prior to experiencing some debilitating effect of exposure to extreme temperature, fish often respond behaviorally in an effort to avoid or minimize the exposure. Studies on behavioral thermoregulation both in the laboratory and field have led to the concept of preferred temperatures. Numerous studies have been performed in laboratory settings where fish were presented the choice of a range of temperatures sometimes even including confounding factors such as availability of food and cover [26, 27]. Preferred temperatures determined in this way are usually similar to the optimal temperature for food consumption and growth determined in other studies. Similar investigations have been carried out in the field using telemetry and other methods to follow habitat choice by fish under heterogonous environmental conditions [26, 28]. Environmental management should consider thermal preference behavior when evaluating habitat suitability, manage for needed temperature and thermal structure, and give priority attention to minimizing degradation of preferred thermal habitat, especially when it is in short supply [29].

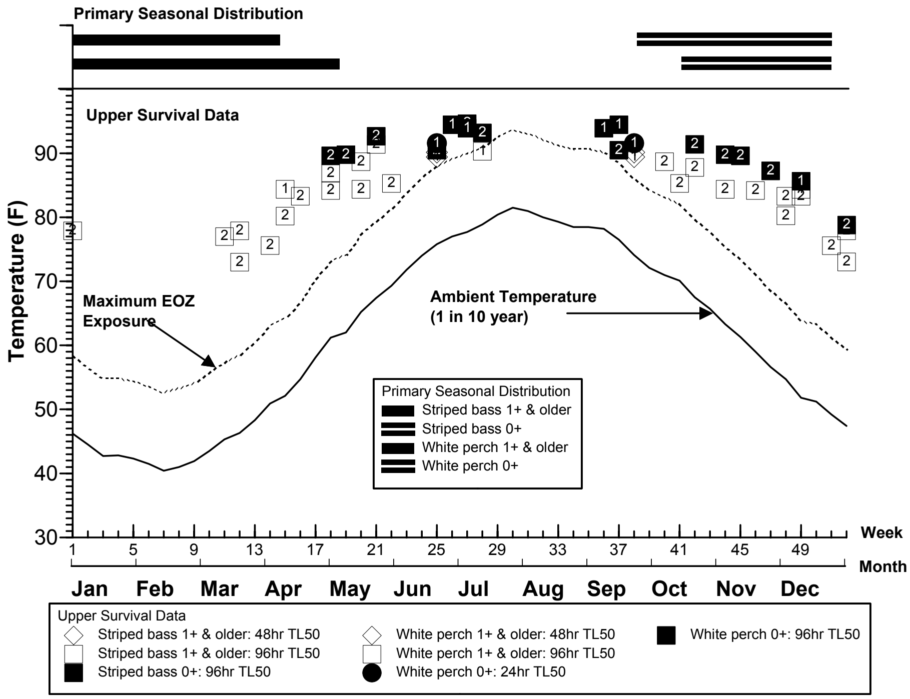

Figure 6-3
Upper Survival Data for Temperate Bass Relative to their Primary Seasonal Occurrence near Salem Nuclear Plant and the Estimated Ambient and Maximum Plume Temperatures at Edge of Zone of Initial Mixing During a Typical Warm Year [25]

Although determination of a fish's preferred temperature tells us where no adverse effects occur, it unfortunately tells us little about thermal tolerance at the extremes. However, Eaton et al. have developed a method that uses field observations of inhabited temperatures to estimate thermal tolerance values for 30 species [30]. They accumulated data on the temperatures at which fish were collected (or observed) throughout their historical range along with the dates when collected (Figure 6-4). They then calculated the 95th percentile of the top 5% of observed occupied temperatures as an estimate of an upper tolerance limit. They contend that this method is useful for developing thermal criteria because their results were similar to (but not greater than) existing EPA criteria. The EPA typically uses critical thermal maxima minus 2°C to derive short-term exposure limits for their water quality criteria [18].

Where thermoregulation represents a behavioral response to the thermal environment, there are other behaviors that are affected as a result of extreme temperature exposure. For example the ability of a fish to escape a predator can be slowed by warm or cold temperatures. Predator avoidance studies were designed to quantitatively measure the impact of thermal exposure on the ability of exposed fish to escape a predator [31]. Groups of fish exposed to different temperatures are presented as separate groups to predators. After a set time (e.g., 0.5 hr), the proportion of each group consumed is noted to determine the temperature exposure at which escape behavior is affected. One study found that the duration of exposure time necessary to result in a significant decline in predator avoidance ability was 10-20% less than that necessary to cause visible equilibrium loss [31]. Other investigators have used high speed photography to investigate the effects of environmental stressors on the initial startle response that occurs at the outset of an escape movement [32, 33].

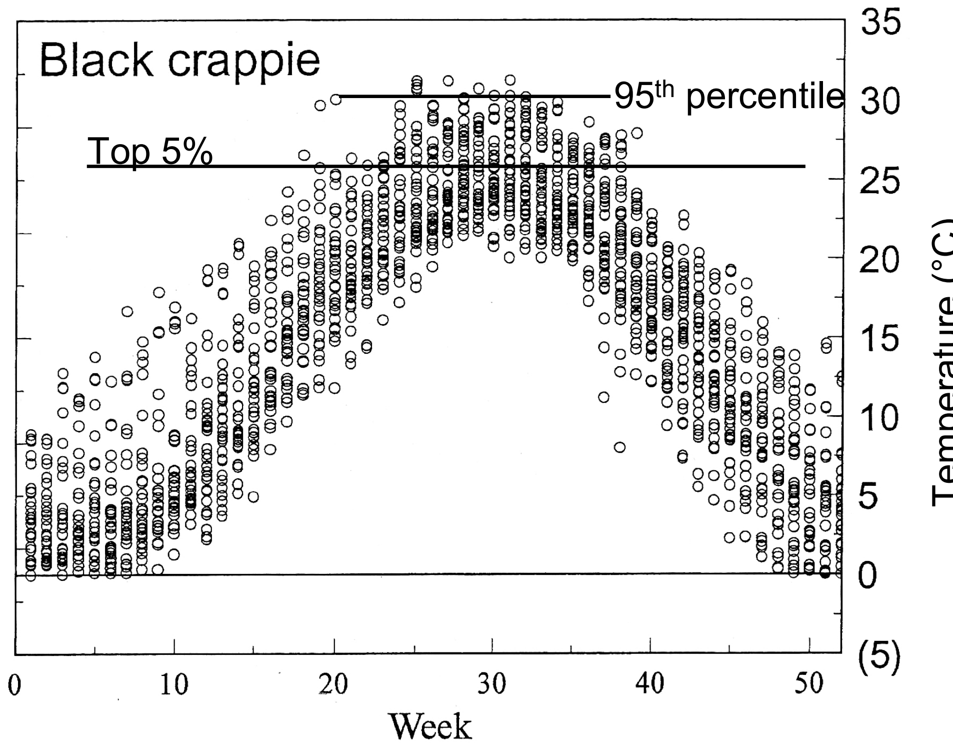

Figure 6-4
Temperatures Occupied by White Crappie throughout the United States by Week of the Year [30]. Upper Temperature Tolerance is Defined as the 95th Percentile of the Top 5% of observations

Further up the hierarchy of biological responses to temperature is the effect on growth. In the laboratory, growth rates can be measured at different temperature regimes [34]. Adverse effects on growth can be measured in the field over time using standard methods such as repeated capture and measurement of tagged fish or by analysis of scales and otoliths. The effect of different temperature regimes on growth can also be investigated by simulation with a bioenergetics model which tracks the distribution of consumed calories to various metabolic costs and growth [35, 36]. Because metabolism and food consumption rate (and many other physiological functions) are directly related to temperature, fish growth is also temperature dependent. These models have been developed for a number of species (ref) and allow investigators to easily test response to alternate temperature regimes. For example, long-term growth simulations of a representative female white sturgeon under normal (i.e., baseline) conditions reveal small within year fluctuations in weight resulting from slow (or negative) growth during periods of high and low temperatures (Figure 6-5) [36]. Large declines in weight occur during spawning events when eggs are released. When the entire temperature regime is increased by 1°C, growth is reduced and spawning events occur less often. When the temperature regime is increased by 2°C, growth is reduced further and no spawning occurs at all because of the sturgeon's poor condition.

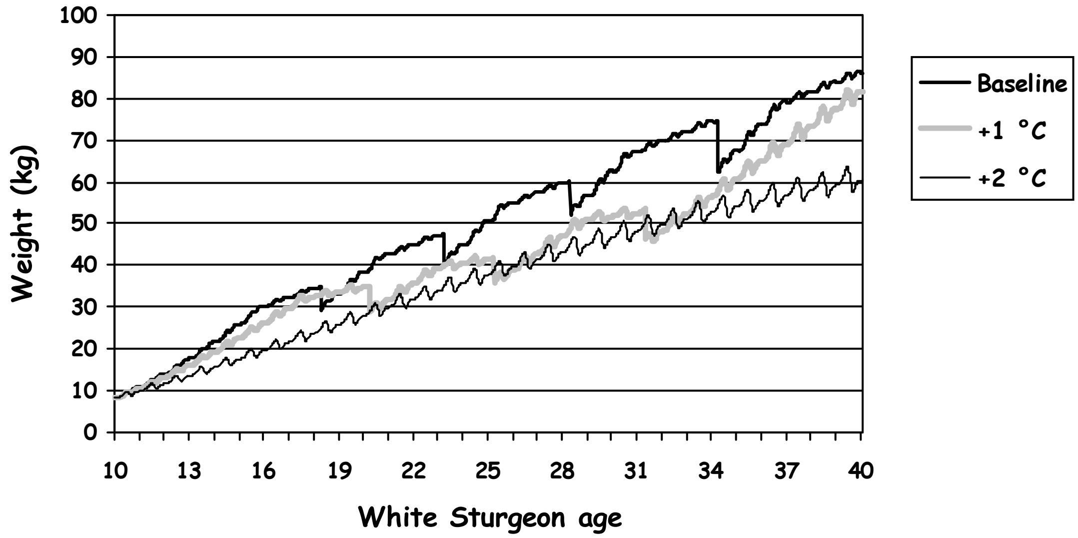

Figure 6-5
The Total Weight of a White Sturgeon Female during a 30-Year Simulation with Three Temperature Regimes

Results from laboratory, field, and computer simulations, like those described above, can be combined to generate thermal effects polygons describing the thermal limits of multiple activities or life stages. In Figure 6-6 the upper and lower lethal temperatures for young sockeye salmon (*Oncorhynchus nerka*) are plotted to describe the zone of tolerance. Within this zone, two other zones are represented to illustrate (1) an area beyond which growth would be poor to none at all under the influence of the loading effect of metabolic demand, and (2) an area beyond which temperature is likely to inhibit normal production [1].

Assessment of High-Level Biological Responses—Populations/Community

Ultimately, thermal effects need to be assessed at the population or community level. Although the unnecessary death of individual fish is not desirable, the effect on a population or community may be inconsequential. Standard fish sampling methods can be used to assess changes in fish abundance over time. Where field methods are not practical, many investigators have turned to computer models (e.g., individual-based population models and age-based matrix models) to evaluate potential population level effects [37]. Community-level effects such as a change in biodiversity can be assessed using standard measures of biodiversity and species richness or newer methods such as the Index of Biotic Integrity recently developed for many aquatic systems [38, 39, 40, 41].

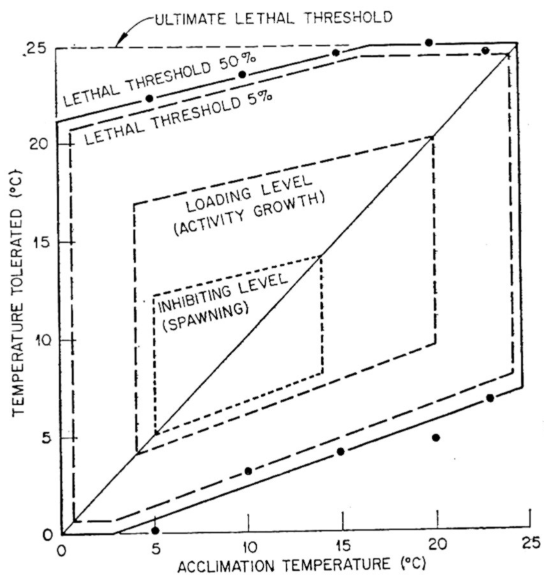

Figure 6-6
Thermal Effects Polygons for Young Sockeye Salmon [1]

Recent Advances in Thermal Assessment

As biologists develop experiments to better understand the effects of exposure to extreme temperatures and as others try to use those results to assess impacts and develop regulatory criteria, it quickly becomes apparent that the interactions are complex and that existing information is rarely adequate to address all the questions. Recent research on several fronts provides examples of methods that are being developed to better understand thermal response in fish.

On a biochemical level, several researchers have been investigating the utility of using intercellular heat shock proteins (HSP) as indicators of thermal exposure and stress [42, 43, 44]. These proteins are expressed above baseline levels as part of an organism's response to protein denaturing caused by a variety of potential stressors. It is believed that different stressors result in the expression of a different complex of HSP. Like other thermal response endpoints, HSP expression is a function of acclimation history, exposure duration, and other exposure dynamics. An understanding of the circumstances that result in elevated HSP expression will provide a better understanding of extreme temperature effects and provide an additional tool for detecting when sublethal thermal stress is present.

The thermal tolerance tests described above (CTM and ILT methodologies) are useful for predicting the effects of a sudden exposure to high (or low) temperature like that that might be experienced as a fish passes through a heated plume from a powerplant. However, this type of rapid exposure is not particularly representative of the long-term or chronic exposure that is often experienced in more natural systems. Recent studies have investigated thermal regimes that are less dramatic and more natural in terms of exposure duration and temporal dynamics. Zale and Gregory developed the Acclimated Chronic Exposure (ACE) method for cultured tilapia *Tilapia aurea* that was recently applied by Selong et al. to bull trout *Salvelinus confluentus* [45, 46]. This method is a hybrid of the ILT and CTM that uses a much slower increase from acclimation temperature to test temperature (about 1°C/d). Fish are then held at the test temperature for 60 days or until death. Lethal temperature is defined as that at which 50% survive for 60 days. The authors concluded that the ACE method offers a better tool for defining temperature criteria and assessing fish responses to thermal change in natural systems. Calibration tests to relate these ACE results to the more prevalent CTM. ILT, growth, and other methods would be useful.

Johnstone and Rahel took a different approach to assess thermal tolerance under natural (but stressful) thermal regimes [47]. They modified the ILT method to assess the influence of daily fluctuations of 10°C on the thermal tolerance of cutthroat trout *Oncorhynchus clarki*. They tested a 3-week exposure to a gradually warming cyclic regime and found 100% survival at 10-20°C followed by 16-26°C (17 days total). As the experiment progressed, all survived for 1 d at a 17-27°C cycle, 22% died after 1 day at 18-28°C, and the remaining died after 1 day at 19-29°C. By this method they found that fish could survive short periods of exposure above temperatures that would otherwise have been lethal under a long-term chronic exposure. Similar results were observed in a companion field study [48].

An alternative to assessing the effects of chronic exposure and fluctuating temperatures in the laboratory would be to develop a computer model of thermal stress response. Bevelhimer and Bennett present a modeling framework for assessing cumulative thermal stress in fish that assumes that stress accumulation occurs above a species-specific threshold temperature at a rate dependent on the degree to which the threshold is exceeded [49]. The model includes short-term and long-term acclimation and the rate of temperature increase as factors in determining the rate at which thermal stress accrues. The model also includes a mechanism for stress recovery (or alleviation) when temperature drops below the threshold temperature as in systems where anthropogenic heat sources are intermittent and in unaffected systems with large daily variation.

Model such as this one can be a useful tool for reducing the impact of power plant operations on fish populations. One way for mitigative measures to be both protective and cost efficient is to become more responsive to short-term changes in plant operations and environmental conditions. This model can be used as part of a real-time monitoring system to provide adequate warning for mitigative measures to be invoked prior to actual environmental impact.

Summary

The development of regulatory criteria for thermal exposure that began in the 1960s and continued at a relatively high level of activity to the mid-1980s appears to be heading toward a bit of a resurgence as the power industry evolves, thermal effects become better understood, and the possibility of global climate change becomes more likely.

Hanging over all predictions of the effects of thermal additions to the environment by power stations or other anthropogenic sources is the apparent reality of gradual climate change and global warming [50]. This warming, coupled with oceanic cycles that affect continental temperatures in roughly 7- and 20-year periodicities is likely to create stressfully high ambient temperatures for aquatic life [51]. Thus, thermal-effects assessments based on historical temperature regimes may underestimate the detrimental effects of other thermal additions. The exact adjustments that must be made are not known, partly because the degree of warming is likely to vary regionally within a continent [50]. Allowances for temperature elevations of ambient by at least 1-2°C would seem prudent.

Because a fish's response to extreme temperatures occurs at several levels of biological organization and over different time scales, a variety of methods have been developed to assess thermal effects. Early techniques (ILT & CTM) were designed primarily to measure response to rapid temperature change and were primarily suitable for evaluating the effects on fish passing through power plant thermal plumes. These methods have recently been modified to assess less dramatic temperature changes such as those that might be experienced in a more natural system with only a moderate change in temperature.

Other methods have also been developed to investigate responses other than direct mortality. These other methods include behavioral tests, measures of biochemical response, and models of growth and thermal stress. Ultimately, resource managers should be most concerned about long-term responses that affect population dynamics and community structure.

Future needs in thermal assessment include:

1. continued development of new techniques that can assess the effects of chronic exposure to temperatures that are not immediately lethal but still stressful,
2. the development of bioindicators of past thermal exposure (e.g., stress proteins, RNA/DNA ratios) that can be used to assess the state of fish health in the field,
3. a better understanding of relationships between physiological and behavioral responses,
4. the development of more accurate computer models of both exposure and thermal stress accumulation, and
5. the development of techniques for real time monitoring and mitigation.

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7
MEASURING COMMUNITY BALANCE AND THERMAL EFFECTS IN FRESHWATER

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Introduction

In this paper, I discuss how to: design and conduct 316(a) studies, measure community balance, and assess community structure using multimetric indices. The approaches described herein are based on conducting such studies for 25+ years, as well as a review of the literature [1, 2, 3]. Because once-through power plants are necessarily located on large waterbodies, the discussion herein is limited to such waterbodies, i.e., medium to large rivers as well as lakes and reservoirs. Most of my studies have centered on freshwater systems so my approaches and examples are based on studies of freshwater aquatic communities. However, many of the approaches I describe apply equally well to marine or estuarine systems. All that will be needed are changes in the sampling gears and some of the assessment techniques.

How to Sample Large Freshwater Systems

Approximately half the nation's steam-electric generating capacity is located on large rivers. Thus, it is particularly important that they be sampled effectively. EPA presently provides no guidance for sampling large rivers. To fill this void, EPRI recently funded a study in which I reviewed, summarized, and assessed numerous methods for sampling freshwater fish in large waterbodies [1]. In the late 90's, EPA provided a guidance document for conducting bioassessments of lakes and reservoirs [2], and around the same time, under this sponsorship of EPRI, EA Engineering developed a "Catalog of Methods" in which they reviewed and summarized numerous descriptive and predictive methods for evaluating the effects of power plant operations on aquatic communities [3]. These three reports, among others, should be consulted before any 316(a) studies are designed or implemented.

What to Sample

During the peak of the 316(a) Demonstrations in the 1960s/70s, it was common to collect data on all trophic levels; phytoplankton, rooted macrophytes, zooplankton, macroinvertebrates, shellfish, and fish. In recognition of their short generation times and broad sources for recolonization, phytoplankton and zooplankton are rarely assessed any more in freshwater systems. Although state-specific concerns may dictate otherwise, there also seems to be a general consensus that if fish are protected, then macroinvertebrates do not need to be studied. In practice, the typical freshwater 316(a) demonstration now focuses on fish. In marine or estuarine systems, shellfish would also require close scrutiny. Because fish will be the target group at most freshwater sites, the remainder of this paper will be devoted to discussing the methods for collecting and assessing fish communities.

What Gears To Use

It is often stated that electrofishing is the best single gear to collect fish [4, 5]. However, it is also clear that in large waterbodies, no single gear captures all the species [6]. The improved coverage provided by multiple gears is demonstrated in Table 7-1 for three areas of the Wabash River and one area on the lower White River in Indiana.

Table 7-1
Percent of Total Species Collected According to Gear Type

Power Plant	Total spp. Collected	% by Gear	
		Shock	Seine
Cayuga	71	66	70
Wabash River	74	76	86
Breed	73	75	89
White River (Ratts)	76	69	76
	All sites combined	69	76

During these studies, which each lasted several months, >70 species were collected from each of the four areas (EA field data). However, even when the data for the entire study period and all the areas were combined, an average of only 69% of the species encountered were collected by electrofishing, whereas seining yielded 76% of the total species encountered. When these data were broken down by month, but with the locations combined, the percentage that any one gear contributed ranged from 37 to 90%. At a single location and on a single date, the percentage that any one gear contributed ranged from 33 to 94%. Thus, it is clear that in a large river with a diverse fish fauna, shocking or seining alone will, on average, only collect about 2/3 of the species actually present, even with repeated sampling. And, on individual dates, <50% of the species present are often collected by one of these gears.

Because you want to get a broad cross-section of the species present including both large and small species, I recommend a combination of electrofishing and seining at large river sites. Based on my experience, these two gears, when used at the appropriate frequency should collect a high (80-90%) percentage of the fishes present. However, even these gears tend to overlook certain species. For example, sturgeon are greatly under-sampled by electrofishing [6], so gill netting would need to be added if one of the sturgeon sp. was a target species. Similarly, deep, mid-channel areas are very difficult to sample. Thus, if certain species (e.g., some of the big river chubs, *Macrhybopsis meeki, M. gelida*) are of concern then fine-mesh trawling may be required. But in most situations, electrofishing and seining should suffice. Electrofishing is much more effective at night [5, 7, 8], so I recommend that it be conducted at night. Other details regarding how to sample large rivers effectively can be found elsewhere [9].

In lakes and reservoirs, electrofishing should always be one of the gears of choice. Because these waterbodies often contain large populations of benthic species (e.g., catfish) and pelagic/profundal species (e.g., *Morone* spp., *Pomoxis* spp.), gill netting with multimesh nets is also recommended.

Study Design

In assessing possible thermal impacts, we need to know how fish are distributed relative to thermal inputs and how these distributions change seasonally.

Sampling Frequency

I recommend that sampling should be conducted at least seasonally (including the winter), with monthly sampling during the summer, if possible.

Sampling Locations and Effort

At a minimum there should be three locations; in rivers this would include one upstream of the plant's discharge, one near field area, and one far field area. In practice, you'll usually need more than three locations. In large rivers, it is often a good idea to match each location on the plant side with one on the opposite shore to help separate normal longitudinal or habitat effects from those caused by the thermal discharge. For electrofishing, the effort at each location should consist of a 500-1000 m zone, or 20-60 minutes per zone. To the greatest extent possible, zones should have similar habitats.

Other Things to Measure or Consider

Measure basic water chemistry, water temperature and dissolved oxygen at all locations, and secchi depth and specific conductance at all electrofishing locations. Because temperature is the main constituent of concern, additional effort is often required to describe its magnitude and distribution correctly. Typically, this would involve taking measurements at various points along each electrofishing zone (or at least at the beginning and end of each zone) and, because of the buoyancy of heated water, taking periodic vertical temperature profiles.

Besides the spot measurements to be conducted in conjunction with the biological sampling, detailed studies need to be done to describe the behavior of the thermal plume under varying seasonal and flow conditions. Often these studies will involve preparation of a thermal model. In all cases, various model predictions should be "ground-truthed" under a variety of conditions, then the model recalibrated as necessary.

Habitat affects the abundance, quality, and distribution of fishes [10]. Thus, it should be evaluated and measured in conjunction with the initial selection of sampling locations. As much as possible, habitat should be standardized among locations to reduce its influence.

How to Determine Whether a Balanced Indigenous Community (BIC) is Present

A successful 316(a) demonstration is based on the maintenance of a BIC. For existing facilities, you must be able to demonstrate that there is no "appreciable harm" under current operating conditions or, if such harm does exist, that it is not the result of the thermal discharge (i.e., the thermal discharge does not prevent establishment of a BIC). To measure/assess the BIC you must:

- Look at the community holistically.
- Demonstrate that key species, often referred to as Representative Important Species (RIS) are doing well ***and*** that the community is doing well.
- Statistically compare various measures of community health (e.g., total CPEs, CPE's of RIS, status of thermally sensitive species, etc.) in thermally affected zones with those same measures in thermally unaffected areas. Typically, you will also have to extrapolate your findings in order to predict what will happen under worst case conditions (i.e., high air temperatures and low river flows). But your study plan should be flexible enough so that you can sample if true worst case conditions do develop during the monitoring period, thus eliminating the need for predictions of what might happen. In a traditional 316(a) Demonstration, a Representative Important Species List is developed in conjunction with the appropriate regulatory agency, then predictions are made as to how the various RIS will react to various thermal loadings. This species by species approach, though still useful, doesn't really address the issue of a BIC. Because the establishment of a BIC is central to a successful 316(a) Demonstration, it seems reasonable to try to measure/evaluate the BIC directly, rather than relying solely on the RIS approach. This is where multimetric indices, especially the Index of Biotic Integrity (IBI) [11] come into play. Because the IBI is based on a composite of community based measures (usually referred to as metrics), it is very useful for 316(a) purposes. The absence of depressed IBIs is proof of the absence of adverse thermal effects since any such effects should be reflected in the various IBI values. Similarly, because the IBI measures community health, the presence of high IBI scores indicates that a BIC is present. However, because the IBI responds to multiple perturbations, depressed IBIs establish only that impacts are occurring, but not whether those impacts are thermally related.

Unfortunately, IBIs have been established for only a few large rivers or areas:

- Ohio River mainstem [12]
- Large rivers in Ohio [4]
- Large rivers in Wisconsin [13]

A couple of years ago, I discussed the difficulties associated with developing IBIs in large rivers [9]. Little progress has been made since that time and this remains an area where further research is needed.

Examples of How the IBI Can Be Used

Provided below are four examples where the IBI has been used successfully to assess thermal impacts.

1. In 1999, based on six electrofishing passes on the Muskingum River in Ohio, EA Engineering [14] found that the mean IBI score downstream of the Muskingum River Plant (MRP) during the summer (late June thru Sept) was identical to the mean score upstream of it *and* that scores in both areas met the expected biocriteria for this area [14]. These good downstream IBI scores occurred despite the fact that downstream water temperatures during the July sampling events ranged from 32-35C. These results indicated that the thermal plume from the MRP was having little or no impact on fishes downstream of it. Partly as a result of these findings, American Electric Power, the owner of the MRP, was able to obtain slightly less restrictive thermal limits for this plant.
2. Since 1991, IBIs have been calculated near several Ohio River power plants that have been studied as part of the Ohio River Ecological Research Program (ORERP). These studies have shown that typically there is no upstream/downstream difference in mean IBI scores [15].
3. Monitoring conducted near several power plants on the White River in Indiana has also revealed few upstream/downstream differences even though short-term avoidance has clearly been evident at some of these plants [16].
4. In 2001, EA Engineering [17] prepared a 316(a) Demonstration for a paper mill on the Pigeon River in North Carolina (NC). Fish were sampled once in July and once in August. The August results were:

	Control Area	Most Thermally Affected Area
Water temp (C)	19.8	31.3
IBI	52	52
Species Richness	16	18

What was particularly interesting about this site was that the composition of the fish community was quite different downstream of the mill, and, in my opinion, that it was not balanced due to a reduction in darter numbers and diversity. But in this case, different was not worse as measured by the IBI. I concluded that the lack of darters was partly the result of blockage from a dam downstream, thereby preventing the more thermally tolerant darter species found further downstream from recolonizing the area below the mill. Thus, it was not temperature that was causing the downstream community to remain unbalanced, rather it was a lack of recolonization sources. Both the state of NC and Region IV EPA accepted this interpretation and the mill's thermal variance, which was originally granted in the mid-80's, was continued as part of the mill's new NPDES permit.

Problems with Current Lake and Large River IBIs, and the Use of RIS

1. As mentioned previously, there are very few IBIs applicable to large waterbodies.
2. As pointed out by Jim Karr who developed the IBI, productivity measures (e.g., CPE) vary widely, often unpredictably, yet most IBIs include at least one CPE metric [18].
3. Changes in downstream IBIs may be measuring short-term avoidance, not long-term impacts.
4. Intolerant lists used in most IBIs are dominated by small stream forms that are naturally rare in lakes and large rivers [9]. Intolerance rankings usually do not take into account thermal tolerance. For example, blue suckers are sensitive to habitat alterations but, based on my observations, do not appear to be particularly thermally sensitive.

I suggest that an IBI that includes metrics more responsive to thermal enrichment should be developed. Possible metrics could include:

Metric	What it Would Measure
(1) Species richness	Overall impacts, Avoidance
(2) % round-bodied suckers	Response of a thermally sensitive group
% thermally sensitive spp. (e.g., white sucker, yellow perch, N. pike)	Response of all thermally sensitive spp.
% thermally tolerant spp. (e.g., common carp, flathead catfish, bluegill, river carpsucker)	Species that should increase in response to excessive thermal enrichment
(5) % DELT anomalies	A measure of thermal stress
(6) Wr (Relative Weight)	A measure of effects on growth
(7a) CPE (total spp.) (7b) CPE (RIS)	Avoidance, Effects on overall production Avoidance, Effects on overall production
% YOYs or % of spp. with multiple year classes	Effects on reproduction
(9) % omnivores	In thermally stressed environments, omnivores are likely to increase
% change between summer and other seasons (as measured by total CPE, RIS CPE, or richness)	Indication and measure of avoidance

I stress that the appropriateness of these metrics remains to be demonstrated. Nonetheless, I believe that at least some of them will work and there undoubtedly are others that can be developed. The important point is that if we are going to use the IBI to assess thermal impacts then we need metrics that are responsive to thermal inputs.

What About "Worst Case" Assessments?

In my experience, most worst case assessments based on predicted outcomes tend to exaggerate the sensitivities of the local aquatic community, probably because they don't take avoidance into account. Therefore, if possible, it is best to collect field data during and following worst case conditions and examine those data, rather than relying on predictive exercises. This requires flexibility on the part of those conducting the field studies, so that collections can be made if worst case conditions develop.

Avoidance

In EPA's 1986 WQ Criteria Summary document (i.e., the gold book [19]), the section on temperature included the following statements:

> *"juvenile and adult fish usually thermoregulate behaviorally by moving to water having the temperature closest to their thermal preference" (emphasis added). The EPA report goes on to note that "this response (avoidance) precludes problems of heat stress by juvenile and adult fishes during the summer".*

So it is well established that fish respond in a predictable manner to high temperatures. One of the challenges for both the utility industry and the regulatory agencies is how this known behavioral response can be factored into the standard-setting process.

Some states (e.g., Ohio) make a distinction between short-term avoidance, which is a good thing (because it allows fish to avoid lethal conditions) vs. long-term avoidance, which is probably a bad thing because it may exclude fish from favored feeding or nursery areas for long periods or may prevent access to spawning areas at key times. In considering short term vs. long term avoidance and assessing the magnitude of any impacts associated with avoidance, here are some things to consider:

- the duration of avoidance
- how many species are affected
- are critical habitats affected
- are all age classes of fishes being affected

Examples of Avoidance Working

Turtle Creek Reservoir is a 1550 acre reservoir in west central Indiana that I have been working at annually since it was created in 1980. For a considerable period, Turtle Creek supported one of the best trophy largemouth bass fisheries in the Midwest despite the fact that temperatures in the discharge cove of the lake occasionally exceed 40 C almost every summer [20]. Our long-term monitoring shows that when temperatures are excessive, most species simply avoid this area, return when temperatures are more favorable, and are attracted to this area during the winter [20].

On the Muskingum River in Ohio, it was documented that fish moved into two tributary streams when temperatures in the mainstem became excessive [14]. Any biologist who has worked near power plants for any period realizes that avoidance is a predictable, well-known phenomenon. The challenge is how to use it as part of a 316(a) Demonstration and how to incorporate it into state water quality standards.

A Current Case Study Example

In conjunction with a Use Attainability Study being conducted on the Upper Illinois Waterway, EA Engineering is assisting Midwest Generation determine the appropriateness of Illinois' thermal standards. The area focused on was Dresden Pool, a 14-mile reach of the Des Plaines River formed behind Dresden Island Lock and Dam (Figure 7-1). The portion of the pool that is below the I-55 bridge (Lower Dresden Pool) is classified as General Use, whereas the area upstream of the bridge is classified as Secondary Use and, as a result, Upper Dresden Pool has considerably less restrictive thermal limits.

To assess the protectiveness of the current thermal standards, I established a list of RIS. Based on thermal tolerances of these species as reported in the literature AND taking short-term avoidance into account, I concluded that the secondary thermal standard would be protective of most species, but probably not for redhorse.

To quantify impacts to redhorse and possibly other species, I used the long-term biological data base that had been collected from Dresden Pool [21]. Since Lower Dresden Pool is already under the more stringent General Use Thermal Standard, I theorized that the fish community in Lower Dresden Pool should be a good predictor of what the one in Upper Dresden Pool would look like if the more stringent General Use thermal limits were applied there. For this approach to be valid, habitat quality in the two areas needed to be comparable. Biologists from EA Engineering calculated Qualitative Habitat Evaluation Index (QHEI) scores at 0.5-mile increments throughout the Pool and confirmed that habitats were similar upstream and downstream of the I-55 bridge. They also determined that habitat was poor and likely be limiting throughout the pool. I then compared IBI scores between the two areas and found that they averaged 2-3 IBI units lower in Upper Dresden Pool, not enough to have any biological significance. More importantly, I found that mean scores in both areas were in the low to mid 20s, far below those expected for an area classified as General Use. Expected IBI scores should be around 40 [4]. Based on poor IBI scores throughout the pool, as well as the lack of difference in IBI scores between Upper and Lower Dresden Pool, I concluded that the entire pool is primarily habitat, not temperature, limited.

Because redhorse were one of our main groups of concern, I compared redhorse CPEs between the two areas. I found that CPEs in the warmer portion of the Pool (i.e., Upper Dresden Pool) were lower than in the cooler portion, but the difference was very small, about 1 redhorse/km vs 2/km, respectively. I also found, using data from other large rivers in northern Illinois, that redhorse numbers in both areas were far below expectations for a river of this size. I again concluded that habitat (specifically, probably a lack of spawning habitat) was the principal factor determining redhorse populations in both portions of the Dresden Pool and that changing the thermal regime would likely result in only a minor improvement in redhorse populations in Upper Dresden Pool.

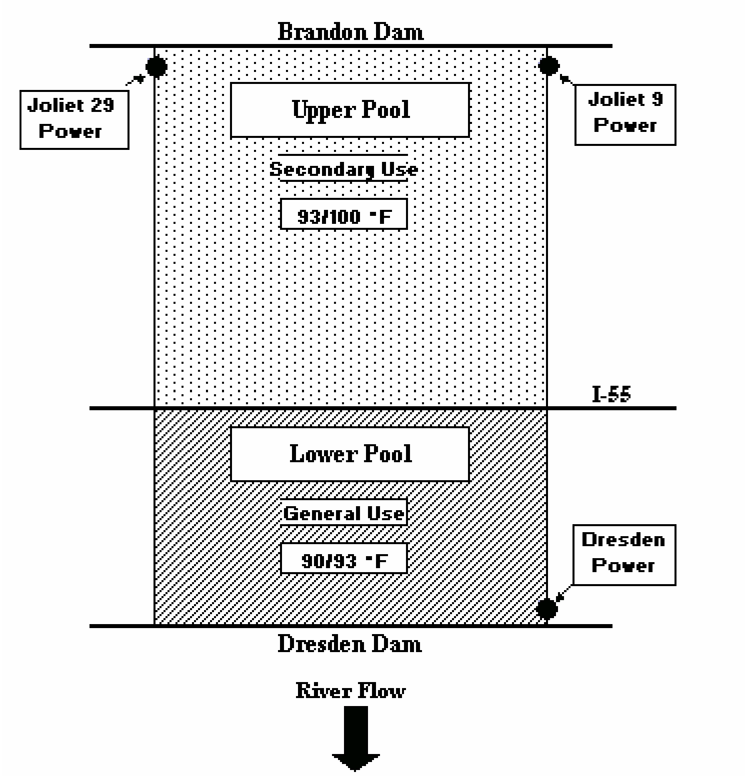

Figure 7-1
Diagram of Dresden Pool on the Upper Illinois Waterway Showing the Dual Aquatic Life Uses and the Associated Temperature Criteria

Lastly, EPA Region V suggested that perhaps the RIS list for Dresden Pool needed to be changed by adding northern pike, walleye, and yellow perch to it. However, a review of the literature indicated that the habitat in this area was unsuitable for northern pike and yellow perch. This assessment was supported by EA field data that showed that perch and northern pike are rare not only in Upper Dresden Pool (the warmer area) but also in lower Dresden Pool, which is cooler because of the General Use thermal standard. EA field data also showed that redhorse and smallmouth bass, species that are as or more thermally sensitive than walleye (the third species suggested by Region V), were much more common in Upper Dresden Pool (the area with the less stringent thermal standards) than walleye. Again, this suggests that factors other than temperature, most likely habitat quality, are responsible for the lack of walleye in Upper Dresden Pool, not the current, less restrictive thermal limits. Given that all three species are habitat limited, I concluded that there was no need to add them to the RIS list originally developed. This last example demonstrates that it is important to develop the RIS carefully and that the availability of corroborating good field data to support your choices can not be over emphasized.

In summary, because every case will be somewhat different, a flexible approach that combines a RIS with a holistic, community-level approach offers the best and most flexible opportunity for developing a successful 316(a) Demonstration.

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8

IN A NUTSHELL: 15 YEARS OF BIOMONITORING AND 316(A) PERMITTING AT TWO AMERICAN ELECTRIC POWER FACILITIES ON THE MUSKINGUM RIVER

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Introduction

Adverse effects of heated water at once-through cooled power plants are episodic, site-specific, and dependent on plant operating characteristics and existing hydrothermal conditions [1, 2]. Significant, prolonged adverse thermal effects can occur when fish and other aquatic life have no refuge from constant elevated temperatures [3]. While environmental permitting of power plant thermal discharges is typically based on "worse case" steady-state design conditions, knowledge of the long-term responses of receiving stream biota to varying elevated temperature exposure would be extremely helpful to allow consideration of risk-based permitting. Such permitting could be based on specified parameters of allowable heat loadings considering the observed severity of biota response (or lack of response) to varying exposure regimes. The maintenance of certain biological endpoints over a specified time (e.g., no significant lowering of community index scores during a 4-year period) could be incorporated into NPDES permits as an alternative to end-of-pipe limitations. In most instances, however, the availability of a long-term water quality and biological database (using standardized methodologies) is lacking. In lieu of a reasonable "dose-response" relationship for a particular water body, permitting agencies may choose to regulate heat loadings conservatively.

American Electric Power (AEP) operates two coal-fired, once-through cooled power plants on the Muskingum River, in eastern Ohio (Figure 8-1). Conesville Station (located in the free-flowing portion of the Muskingum River about 20 km downstream from Coshocton, OH) has three once-through cooled generating units with a combined generating capacity of 415 MW. Just upstream of Zanesville, OH, the Muskingum River becomes impounded by the first of a series of low head dams. Muskingum River Plant is located between the Luke Chute and Beverly dams, about 48 km upstream of the Muskingum River/Ohio River confluence. This plant has four once-through cooled units (combined generating capacity of 840 MW). Figure 8-1 also indicates the 7 day once-in-ten-year low flow statistic for USGS gauging stations just upstream of the facilities.

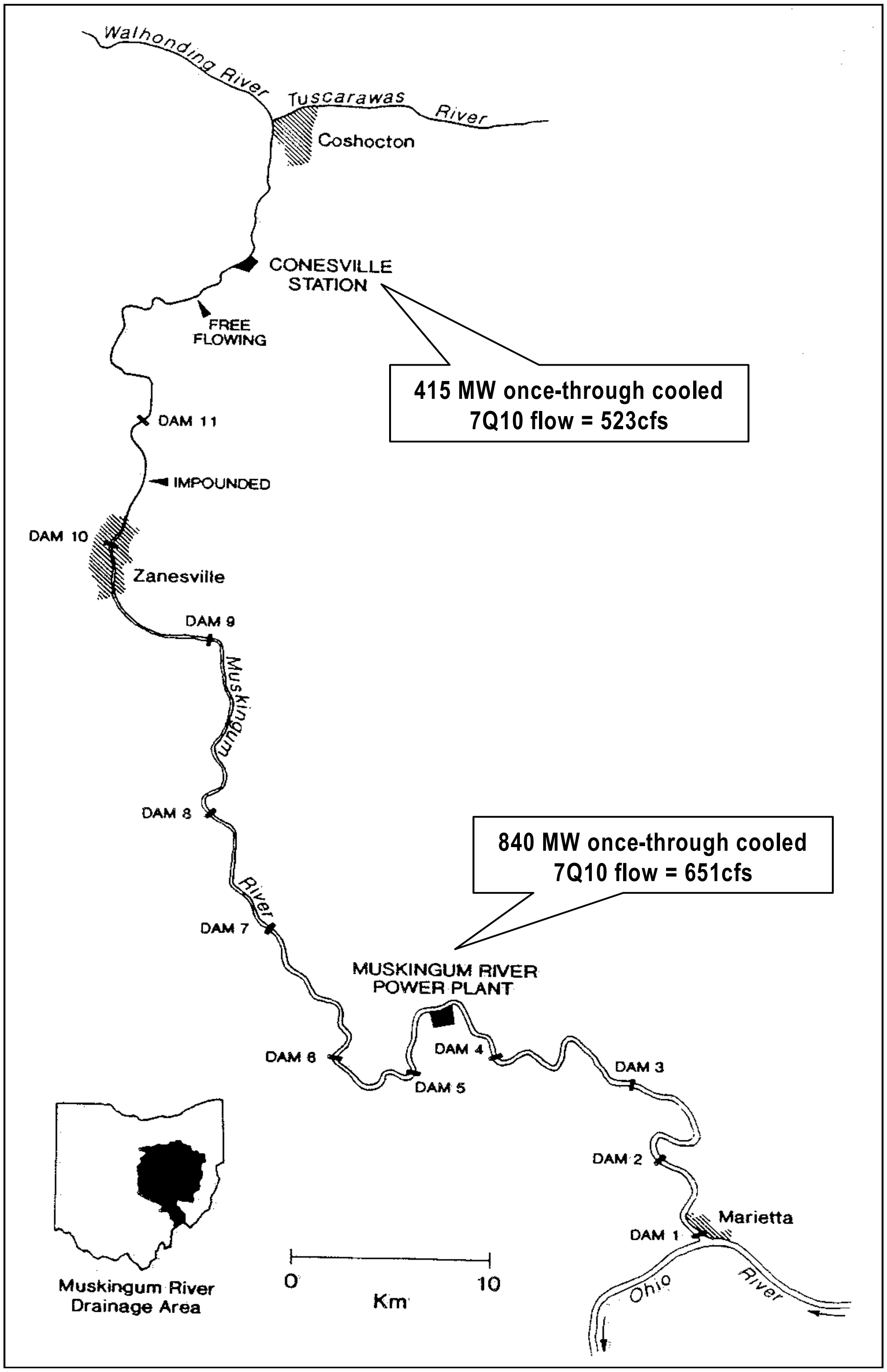

Figure 8-1
Map of Muskingum River Drainage Basin, with Relative Locations of Conesville and Muskingum River Plant. Combined Generating Capacity of Once-Through Cooled Units and Low Flow Statistic (of Muskingum River at Plant Location) are also Indicated

The purpose of this study was to evaluate long-term biological sampling results obtained over several years during varying thermal exposure conditions. For each plant site, the goal was to define specific hydrothermal conditions that were associated with adverse biological effects, if such effects were suggested by monitoring results. A long-term perspective was chosen for effect assessment because: 1) variation in reference "unimpacted" biological community parameters is normal and may be significant between consecutive years; 2) the longevity of important aquatic faunal groups (fish and mussels) exceeds three years; and 3) the exposure of aquatic biota to elevated temperature is typically intermittent and/or cyclical in time.

Thermal Loading Limitations

Both facilities are required to regulate combined unit generation so that certain fully mixed river temperatures are not exceeded during summer months (June 15 through September 30). The specific temperature limitations constitute the approved Section 316(a) variance for each facility, since the temperatures are greater than applicable temperature criteria. Figure 8-2 indicates the permit requirements for fully mixed river temperatures. Once mixed river temperatures reach 91°F (32.7°C), the number of hours at this temperature becomes limited on a cumulative and 24-hour basis. The maximum allowable mixed river temperature is 93°F (33.9°C). One of the purposes of biological monitoring on a long term basis was to evaluate the protectiveness of the thermal loading limitations. Thus, we attempted to determine the biological response to varying *in situ* temperatures, which could be tracked and recorded continuously.

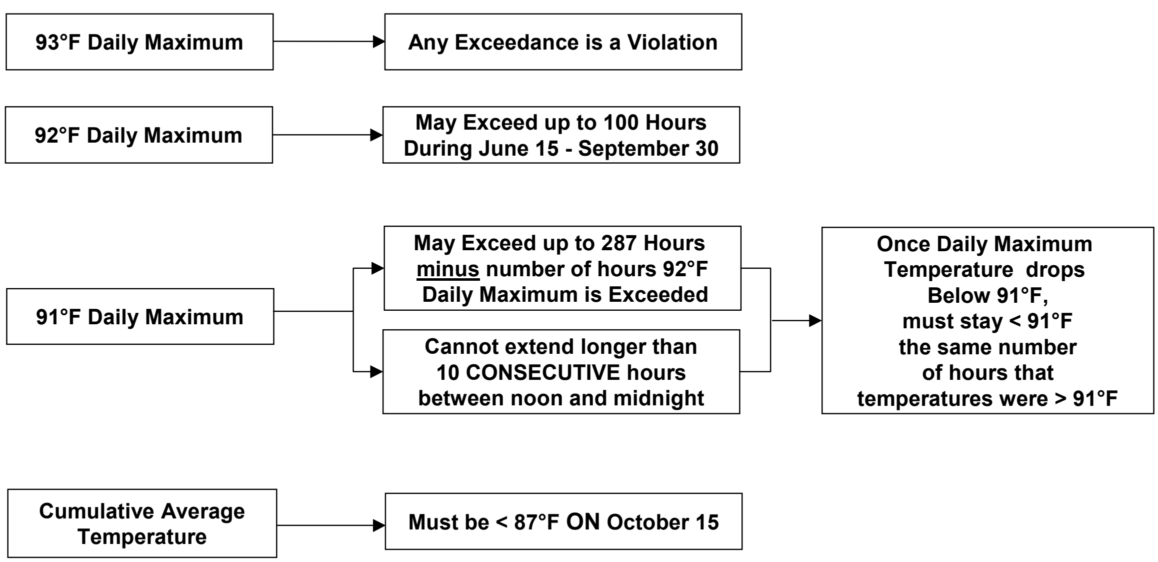

Figure 8-2
Currently Effective Mixed River Temperature Requirements for Conesville and Muskingum River Plants. Downstream Temperatures are either Modeled or Measured with *In Situ* Temperature Probes. The Temperature Requirements are Effective from June 1 through September 30

Field Procedures and Data Analysis

At each plant site, five electrofishing zones were established: two zones upstream of the once-through cooling discharge (Zones 1 and 2), and three zones downstream of the once-through cooling discharge (Zones 3, 4, and 5). At both plant sites, Zone 3 was located within the immediate mixing zone. Electrofishing was conducted near shoreline areas at night, as previous studies in the Muskingum River had indicated that catch rates (especially species richness) were higher during night sampling compared to day [4]. Electrofishing procedures followed Ohio EPA guidelines [5]. A distance of 500 m was sampled in a downstream direction. The shocker was powered by a 240 V, 3,500 W generator to yield 60 pulses per second and 6-10 A. Zones were electrofished for 20–30 minutes with elapsed time being dependent on current velocity and the presence of physical habitat features (e.g., snag piles, emergent aquatic vegetation).

Except for voucher specimens, all fish were identified in the field. Each fish was individually weighed, up to a maximum of 20 individuals for each species. After 20 individuals were weighed, all others were batched weighed and counted. All minnows (except carp) were counted and batch weighed only. Electrofishing sampling was conducted by biologists employed by EA Engineering, Science, & Technology (Deerfield, IL).

At the time of electrofishing, certain water quality analyses were obtained at each zone: water temperature (°C), dissolved oxygen (mg/L), conductivity (µhmos), and water clarity (secchi disk depth in cm). Water temperature and dissolved oxygen were measured at the upstream and downstream ends of each zone; these values were averaged for data analysis purposes.

Electrofishing results for each zone were summarized using two community indices: the modified Index of Well-Being (MIwb) and Index of Biotic Integrity (IBI) [6]. These data were analyzed for location effects (i.e., upstream versus downstream) using the Wilcoxon-Mann-Whitney rank sum test [7]. Associations between community index values and water quality data were evaluated using correlation analysis.

Biological sampling was conducted during seven years (1991, 1993, 1994, 1995, 1999, 2000, 2001). Electrofishing was conducted during summer and fall months (June through October), although sampling during all five months was only conducted during one year (1999). In general, electrofishing was conducted only when fully mixed downstream temperatures (either measured or predicted) exceeded 32.2°C for at least three consecutive days. Thus, during some years sampling was not conducted due to fully mixed temperatures not approaching 32.2°C.

During two summers (1993 and 1999), critical low flow, high temperature conditions were observed in the Muskingum River due to drought conditions. Because these conditions provided the greatest probability for adverse effects to occur, a separate analysis of biological and water quality data for these two years was conducted. A similar analysis was then conducted for all sampling years combined.

Results—All Sampling Years Combined

An analysis of pooled multi-year biological sampling data was conducted to understand how community index scores and metrics vary (and what factors may affect this variation) during a long-term period. The period from 1991 to 2001 encompassed a wide range of hydrothermal conditions during summer and fall months: two years with critical low flow and high temperature conditions (1993 and 1999); one year with extreme low flows only (1991); and eight years of normal to elevated flow conditions. No electrofishing sampling was conducted during 1992, 1996, 1997, and 1998 because downstream river temperatures did not reach 32.2°C for at least three consecutive days.

Summary statistics for water quality and fish sampling data during the 11-year period are provided in Table 8-1. At Conesville Plant, community index scores and two metrics (species richness and relative number, or CPUE) were slightly lower at downstream zones relative to upstream. These values, however, were not significantly different between locations ($P<0.05$). The only variables showing a significant location effect were water temperature and dissolved oxygen concentrations. Water temperature at downstream zones was significantly higher, and dissolved oxygen was significantly higher at upstream zones. These results would be expected, as increased temperature lowers the solubility of oxygen of water.

Table 8-1
Mean Values for Water Quality and Electrofishing Variables for Sampling Conducted Upstream and Downstream of Conesville and Muskingum River Plants, 1991–2001. A total of 17 Electrofishing Replicate Passes (5 Zones Sampled per Pass) were Conducted at Conesville; 18 at Muskingum River Plant

Variable	Conesville Upstream	Conesville Downstream	Muskingum River Upstream	Muskingum River Downstream
Temperature (°C)	22.8	25.0[1]	23.7	27.4[1]
Dissolved Oxygen (mg/L)	8.3[1]	7.1	7.9[1]	7.3
Index of Biotic Integrity	39.6	37.0	41.9	40.4
Modified Index of Well-Being	8.40	8.22	8.75	8.80
No. Species	18.7	17.8	18.1	17.0
Relative Number (# fish/km)	536	549	815	771
No. Electrofishing Samples	34	48	36	49

1. Significantly higher compared to other location (Wilcoxon-Mann-Whitney rank sum test; $P<0.05$).

Correlation analysis for Conesville biological and water quality data indicated few significant correlations between variables. At downstream zones, IBI scores were inversely related to water temperature ($r = -0.36$; $P<0.05$) and positively correlated with dissolved oxygen concentrations ($r = 0.40$; $P<0.01$).

Long term monitoring results at Muskingum River Plant indicated similar results concerning water quality variables; water temperature was significantly higher at downstream zones while dissolved oxygen levels were significantly higher at upstream zones. None of the biological variables were significantly different between locations ($P>0.05$). There were, in fact, less apparent location differences in mean values for biological variables. Mean IBI scores were only slightly lower at downstream zones, whereas mean MIwb scores were slightly lower at upstream zones. No significant correlations between water quality and biological variables were observed for Muskingum River data.

In summary, analysis of long term fisheries monitoring results at both plants indicated few significant differences in variables between location (upstream versus downstream). Moreover, mean community index values at both locations were in attainment of the applicable ecoregion biological criteria for boat sampling sites (36–40 for IBI; 8.1–8.5 for MIwb) [8]. Water temperatures were significantly higher at downstream zones, which was expected due to proximity of the once-through cooling discharge. Long term mean temperatures at downstream zones, though statistically significant, did not approach levels that would be expected to cause adverse thermal effects. The applicable chronic (30-day average) temperature criterion for the Muskingum River during summer and fall months is 29.4°C [8]. The mean temperature at downstream zones was lower than this criterion, indicating that chronic, long term thermal stress at the facilities would not be expected. The fisheries monitoring data, likewise, indicate no significant shifts in community index scores among locations when several years of monitoring (encompassing a wide range of thermal exposures) are considered.

Results—Low Flow, High Temperature Years Only

A separate analysis of upstream/downstream monitoring data was conducted for two periods when critical low flow, high temperature conditions occurred on the Muskingum River. During summer 1993, low flow, high temperature conditions began late in the summer (mid-August). Electrofishing sampling began in late August and continued until mid-October; a total of five sampling events were conducted. Stressful hydrothermal conditions in summer 1999 were first observed in mid-June. River flows decreased steadily in May and June due to drought conditions. Electrofishing sampling began in late June and continued through October, with sampling events occurring approximately every two weeks; a total of six sampling events occurred.

Summary data for water temperature measurements and community index scores at Conesville are provided in Table 8-2. At Conesville Station, sampling results during 1993 indicated a mean increase in downstream zone temperatures of about 6°F. IBI values were similar between upstream and downstream zones, but the MIwb values were lowered downstream, reflecting less equitability of abundance between species. Of the four mean community metric values, the MIwb value at downstream zones was the only one that did not attain the applicable ecoregion biocriterion.

During 1999, mean IBI and MIwb scores at upstream zones (44 and 8.8, respectively) were higher than mean downstream values (37 and 8.5). Mean values for both community indices attained the applicable biological criteria. The highest downstream water temperature was 88.5°F (31.4°C). The lowest dissolved oxygen reading was 4.8 mg/L, measured at downstream zone 4 on June 29.

Table 8-2
Results of Water Temperature Measurements and Community Index Scores at Conesville Station during Two Low Flow, High Temperature Summer Events

1993 Low Flow Conditions		
Variable	Mean Values	
	Upstream	Downstream
Temperature (max)	67.5° (83.1°)	73.7° (86.7°)
IBI (range)	36* (30–46)	37* (30–46)
MIwb (range)	8.16* (7.1–9.3)	7.83 (6.1–9.0)
1999 Low Flow Conditions		
Temperature (max)	77.5° (84.4°)	80.6° (88.5°)
IBI (range)	44* (28–54)	37* (28–44)
MIwb (range)	8.8* (7.5–9.8)	8.5* (7.3–9.4)

* Attains ecoregion biocriterion.

An upstream dissolved oxygen (DO) stress was clearly evident during July and August. High organic loadings from an upstream paper mill caused nighttime depletion of oxygen, as indicated by measurements made by plant staff just upstream of the cooling water intake. During the period July 18–30, plant staff measured DO in the morning (0630–0800 hrs) and afternoon (1500–1600 hrs) each day. Morning DO values ranged between 3.2–4.9 mg/L, while afternoon readings were typically 2–5 mg/L higher than corresponding earlier measurements. Clearly, some background DO stress was evident. While no die-offs of fish or other aquatic life were observed, the discharge of heated water from the plant's once-through cooling discharge lowered DO concentrations (relative to ambient), somewhat worsening the period oxygen depletion condition.

Statistical analysis of IBI and MIwb scores indicated that IBI values at upstream zones were significantly greater than downstream zone values ($P<0.05$). I analyzed the influence of measured water temperature on the absolute difference in IBI scores between upstream and downstream zones, assuming that higher downstream temperatures (or a higher temperature differential) would cause a greater differential of IBI scores between locations. This relationship, however, was not evident; the magnitude of temperature increase was not associated with IBI score differential. Warmer temperatures at downstream zones may contributed to lower IBI scores at downstream zones, but the direct influence of this single variable appears to be subtle based on the limited sample sizes. A more likely explanation of lower IBI scores is the combined effect of increased temperature and reduced oxygen saturation.

At Muskingum River Plant, adverse effects of the once-through cooling discharge during low flow, high temperature critical conditions were not apparent (Table 8-3). Mean IBI and MIwb scores were very similar among locations. The mean water temperature at downstream zones (83.7°F) was about 4°F higher than the mean temperature at upstream zones. During the sampling event on July 31, measured water temperatures at downstream zones ranged from 92.3°F to 94.0°F (33.5°C to 34.4°C). IBI values at these sites were 36 (zone 3) and 38 (zones 4 and 5); interestingly, these values were somewhat higher than scores at upstream zones (32 and 36). Statistical analyses of IBI and MIwb scores obtained near Muskingum River Plant during summer 1999 indicated no significant location effects.

Conclusions

Biological monitoring, conducted at various years and varying thermal exposure conditions at Conesville Station and Muskingum River Plant, yielded results that provide insight to the relative sensitivity of aquatic life in different portions of the Muskingum River. During drought conditions, nutrient enrichment (caused by POTW and paper mill facilities) appears to cause a significant water quality stress in the upper portion. Though no fish kills or other obvious aquatic life injuries are observed during these conditions, nighttime DO sags and aesthetic problems (coloration changes to the river) occur near Conesville Station. This “background” stress interacts with a localized heated water effect. During drought conditions IBI scores downstream of Conesville Station are temporarily reduced compared to upstream areas. Despite reduced IBI scores, the applicable biocriterion is still maintained. Neither increased water temperature nor reduced dissolved oxygen levels, by themselves, are so extreme that lowered IBI scores were likely caused by one of these variables. Rather, a complex combined (antagonistic) effect of these water quality changes appears to be the cause of temporary avoidance by some species. Reduced IBI scores at downstream Conesville zones during drought conditions were temporary and reversible. Sampling results for 2000 indicated improvements in downstream zone scores relative to the previous summer. In short, the existing thermal loading limitations at Conesville Station appear to be sufficiently protective, and less stringent limitations (at least for high temperature, low flow conditions) would not be justified.

Biological monitoring results at Muskingum River Plant indicated no evidence of adverse effects caused by increased water temperature, or the combined effect of increased temperature and slightly lowered DO concentrations. Attainment of the IBI criterion was observed at water temperatures up to 94°F (34.4°C). There are some possible reasons for the somewhat disparate results at the two facilities. The fish community near Conesville Station is exposed to multiple water quality stresses during drought conditions. These stresses likely “kick in” only when river flows reach a certain level, preventing adequate dilution of point-source effluents.

Table 8-3
Results of Water Temperature Measurements and Community Index Scores at Muskingum River Plant During Two Low Flow, High Temperature Summer Events

Variable	Mean Values	
	Upstream	Downstream
1993 Low Flow Conditions		
Temperature (max)	70.0° (83.3°)	79.0° (91.4°)
IBI (range)	41* (32–48)	39* (32–44)
MIwb (range)	8.31* (7.2–9.3)	8.52* (7.4–9.6)
1999 Low Flow Conditions		
Temperature (max)	78.9° (86.4°)	83.7° (93.9°)
IBI (range)	40* (32–48)	40* (32–44)
MIwb (range)	8.9* (7.7–9.7)	8.8* (7.9–9.7)

* Attains applicable biocriterion.

At Muskingum River Plant, the availability of a greater volume of water, and the presence of nearby cool tributaries, likely creates a significant buffer against potential thermally-related adverse effects. If this were the sole reason for the healthy fish community at downstream zones during drought conditions, very few fish would be found at near-shore habitats. This, obviously, is not the case. Another factor that may explain the apparent greater tolerance of fish to elevated temperature is enhanced acclimation. Ambient water temperatures upstream of Muskingum River Plant can reach 30.5 C during drought conditions. Because acclimation temperature dictates thermal tolerance, fish downstream of Muskingum River Plant may be more thermally tolerant because of higher ambient temperatures during critical high temperature conditions. This hypothesis is consistent with many previous findings showing that intraspecific thermal tolerance varies with geographic range and/or ambient physicochemical conditions within a given water body [9, 10].

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9
FISHERIES THERMAL ASSESSMENT STUDIES AT TWO GENERATING STATIONS ON THE WABASH RIVER

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Introduction

The specific reaches of the Wabash River near Cinergy/PSI's Cayuga Generating Station and Wabash River Generating Station have been monitored nearly every year since 1980 [1, 2]. Fisheries studies were conducted to assess the potential thermal effects of the operation of the generating stations on the aquatic community of the Wabash River. The sampling near the station has included electrofishing, gill netting, trammel netting, seining, trawling and rotenone collections. These studies have been concerned primarily with the impact of station operation on the resident fish community. In addition to the Cinergy/PSI sampling, the fish community of the middle Wabash River has been studied annually since 1974 by researchers from DePauw University [3]. These samplings have found a very diverse and abundant fish community in the Wabash River.

Fish health indices (i.e., Index of Biotic Integrity and Index of Well-Being) were calculated from the electrofishing data collected near the stations and compared statistically to determine: 1) Significant differences in the fish community upstream and downstream from the stations; 2) Specific times or river flow conditions when the differences are more pronounced; 3) Differences in the fish community through the years; and 4) Assess the potential for combined impacts of the two stations.

The Study Area

The Cayuga Electrical Generating Station is composed of two 535 MW units, located on the west bank of the Wabash River (RM 250.3) near Cayuga, Indiana in Vermillion County (Figure 9-1). The Wabash River Generating Station is located on the west bank of the Wabash River (RM 215) in Vigo County about four miles north of Terre Haute, Indiana. The Wabash River Station (WRS) has three 90 MW units, one 103 MW unit, one 342 MW unit and repowered unit 1 has 262 MW. Water is removed from the river and used within the plant for cooling.

Table 9-1 illustrates the differences between annual and 7Q10 river flows, and station circulating water pump capacities. The heated water from the station is returned to the river via a discharge canal. Cayuga Station has helper cooling towers which provide once through cooling; however, WRS has no cooling towers.

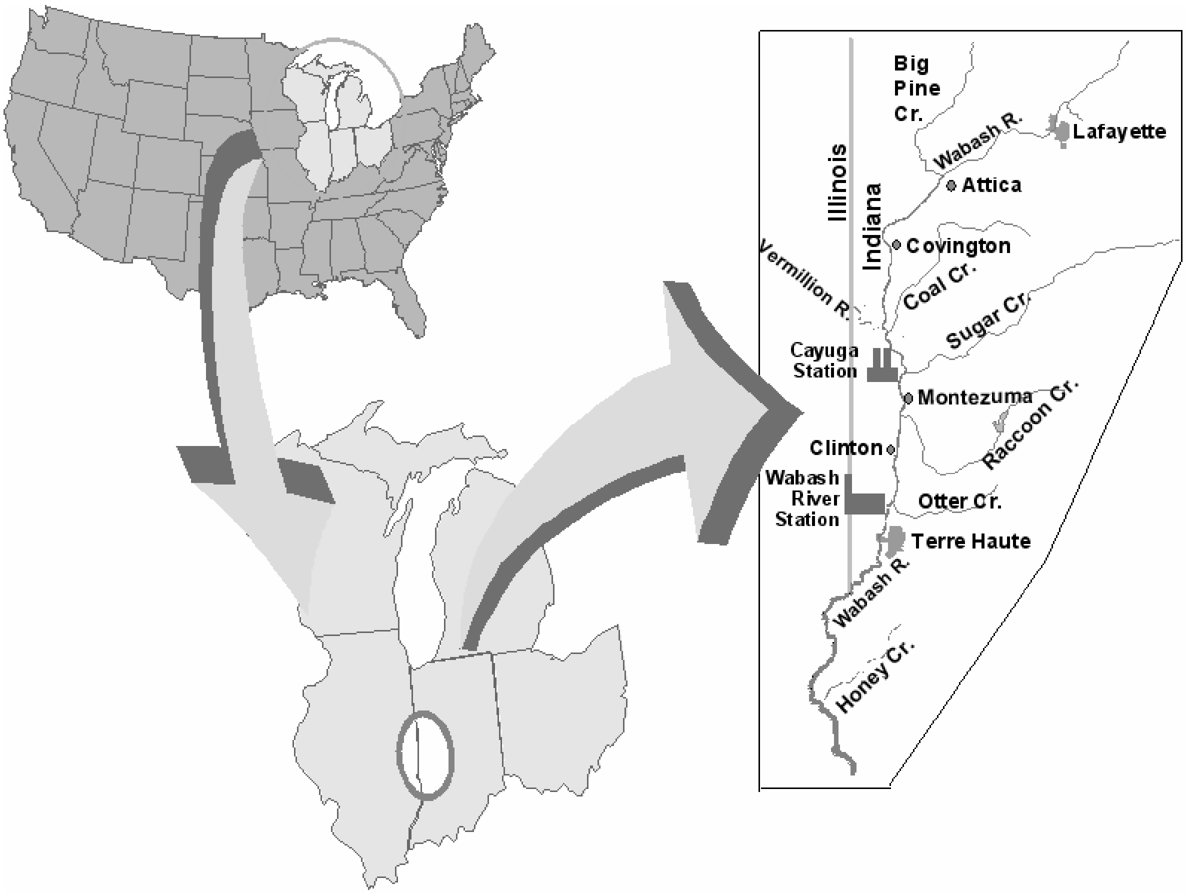

Figure 9-1
Location of Cayuga and Wabash River Stations on the Wabash River

Table 9-1
Comparison of river flows and station pump capacity at Cayuga and Wabash River Stations

	Cayuga Station	Wabash River Station
Annual Mean Flow	10,070 cfs	11,110 cfs
7Q10 Flow	1,210 cfs	1,390 cfs
Station Size	1,075 MW	930 MW
Max Pump Capacity	1,270 cfs	1,156 cfs
Avg. Pump Flow	782 cfs	667 cfs

Methods and Materials

Fish Collections

At each generating station, fish were sampled at six zones in the Wabash River (three upstream from the station and three downstream) during 1991–2002. Prior to 1991, there was some variability in the number and location of the upstream and downstream zones; however several zones have remained consistent throughout the study period.

Because of its effectiveness in capturing a large number of individuals and species, electrofishing is considered the main capture technique in the Wabash River. Statistical comparison were only made on the fish sampled by electrofishing from June through November.

Fish were shocked with a boom fitted, 18-ft jon boat with a Smith-Root Model 5.0-GPP electrofisher powered by a 5000-watt, 10-horsepower Briggs & Stratton generator. The electrodes were modified from those described by Novotny and Priegel [4] with two circular anode assemblies (four anodes per boom) suspended by booms with the boat acting as a cathode. Fish were shocked using pulsing direct current interrupted at 60 pulses per second. The voltage is variable but is adjusted to maintain an output amperage of approximately 10 amps. Each zone was shocked downstream to upstream, remaining as close to shore and submerged cover as possible, then shocked in a downstream direction farther from shore in order to collect the deep-water species.

Fishing effort was determined by zone length (measured in meters). Fish were collected by two netters on the bow of the boat with long-handled 3/16-inch mesh dipnets and immediately placed in an onboard livewell to await processing.

Fish Processing and Data Analysis

At the end of each collection, most fish were identified, weighed, measured and released immediately after collection. Fish less than 20 mm in length (young-of-the-year) were not included in the electrofishing catch because the collection of these small fish is not consistent among zones and they may bias the measurement of ecosystem health [5]. Total length (snout to tail) was determined to the nearest millimeter on a measuring board. Weights were taken to the nearest gram on an HOMS Model 1000 spring dial pan scale for fish weighing one kilogram or less. Fish weighing in excess of one kilogram were weighed to the nearest 50 g on a spring scale. If more than 15 fish of a single species were collected in a given zone, a representative sample consisting of 15 fish was measured and weighed. The additional fish were counted and mass weighed. The average length and weight of these fish were determined by taking the average length of the 15 processed fish and the average weight of the mass weighed fish. Fish too small to process and identify in the field were preserved in formalin for later analysis. Common and scientific names of fish followed the nomenclature listed by the American Fisheries Society [6].

Recorded data from the processed fish were entered into a computer using Microsoft Access which then calculated the following for each sampling method per zone: number per species; percent composition; mean, minimum and maximum weight per species; percent by weight per species; mean, minimum and maximum length per species; total number per kilometer; total weight per kilometer; Shannon diversity indices for number and weight; and composite Index of Well-Being developed by Gammon [7, 8]. The Index of Well-Being (IWB) combines: Shannon species diversity index based on numbers, Shannon index based on weights, number of fish captured per kilometer and the weight of fish caught per kilometer. This index reflects the general "health" or "well-being" of the fish community. IWB is given by the following equation:

$$\text{IWB} = 0.5 \ln N + 0.5 \ln W + \text{Shannon (no.)} + \text{Shannon (wt.)}$$

where,

N = the number of fish captured per kilometer

W = the weight of fish captured per kilometer

Shannon (no.) = Shannon Index based on numbers

Shannon (wt.) = Shannon Index based on weights

In addition, the Index of Biotic Integrity (IBI) as modified by Ohio EPA [5] was manually calculated for each electrofishing sample. The IBI is made of 12 metrics which are scored to assess the health of the fish community. They include:

Category	*Metric*
Species Composition	Total species
	% Round-bodied suckers
	Sunfish species
	Sucker species
	Intolerant species
	% Tolerant
Trophic composition	% Omnivores
	% Insectivores
	% Top carnivores
	% Simple lithophils
Fish condition	% DELT anomalies
Fish abundance	Fish numbers per km

In addition to the statistical comparisons of the IWB and IBI scores, comparisons were also made for the IBI metrics round-bodies suckers (using numbers instead of %) and intolerant species. Round-bodied suckers comprise a sensitive component of river fish fauna while the intolerant species metric is designed to distinguish stream of the highest quality [5].

Analysis of variance (ANOVA), Tukey's Studentized Range Test, and Fisher's LSD Test were used to test for spatial and temporal differences in round-bodied sucker numbers, number of intolerant species, IBI values, and IWB values. Round-bodied suckers and intolerant species are two of the more pollution sensitive metrics used to calculate the IBI. Before each data set was statistically evaluated it was analyzed to determine whether or not the data were normally distributed. If the data were not normally distributed, they were transformed using Log (Y+1).

Results and Discussion

Cayuga Station

The number of round-bodied suckers (e.g., redhorse, blue sucker) upstream and downstream from Cayuga Station was plotted from 1981 to 2002 (Figure 9-2). Blue sucker is a species of "Special Concern" in Indiana [9, 10] that is commonly collected from the Wabash River. Blue sucker of various sizes were collected upstream and downstream from the station, indicating a healthy, reproducing population. The only significant upstream/downstream difference during this period was the larger number of round-bodied suckers at the upstream zones during the drought of 1988. A general trend of increased numbers of round-bodied suckers was noted from 1981 to 2002.

Table 9-2
Cayuga Station Upstream vs. Downstream IBI Statistical Comparisons, 1981–2002

Comparison/Parameter	Upstream Mean	Downstream Mean	Significant Difference[a]	F Value	P Value
All Years and Passes Combined [b]	36.2	34.8	Yes	5.32	0.02
All Years Combined - July/August Passes [b]	35.7	34.1	No	2.43	0.12
1981–1986 Combined - July/August Passes [b]	29.0	26.4	No	2.64	0.11
1987–1990 Combined - July/August Passes [c]	32.7	30.9	No	0.50	0.48
1991–2002 Combined - July/August Passes [b]	38.1	38.4	No	0.17	0.68

(a) Results of one-factor parametric Analysis of Variance tests (alpha=0.05).

(b) Data ranks used for statistical analyses as raw data and log transformed data are not normally distributed.

(c) Log transformed data used for statistical analyses because they are normally distributed.

Another Index of Biotic Integrity (IBI) metric, number of intolerant species, was compared upstream and downstream from Cayuga Station during 1981–2002 (Figure 9-3). The upstream numbers were higher during 1981–1988 (significantly higher in 1988), but similar upstream and downstream in the years after downstream temperature limits and less restricted cooling tower operation were initiated. Like round-bodied suckers, a general trend of increased numbers of intolerant species was noted in the Wabash River from 1981 to 2002.

The mean of the upstream and downstream IBI scores near Cayuga Station was plotted for 1 981–2002 (Figure 9-4). Similar to the IBI metric above, the IBI scores were substantially higher upstream during 1981 to 1988, then similar between the upstream and downstream zones during the subsequent years. The IBI values exhibited an gradual increasing trend from 1981 to 2002 and the community health classification went from "Fair" to "Good". Except for the difference in 1988, there were no significant upstream/downstream differences for individuals years or blocked years (Table 9-5). For all years combined the difference was significant, but not for all years during the July/August samplings, which typically is worst case conditions (i.e., low river flow and high river temperatures).

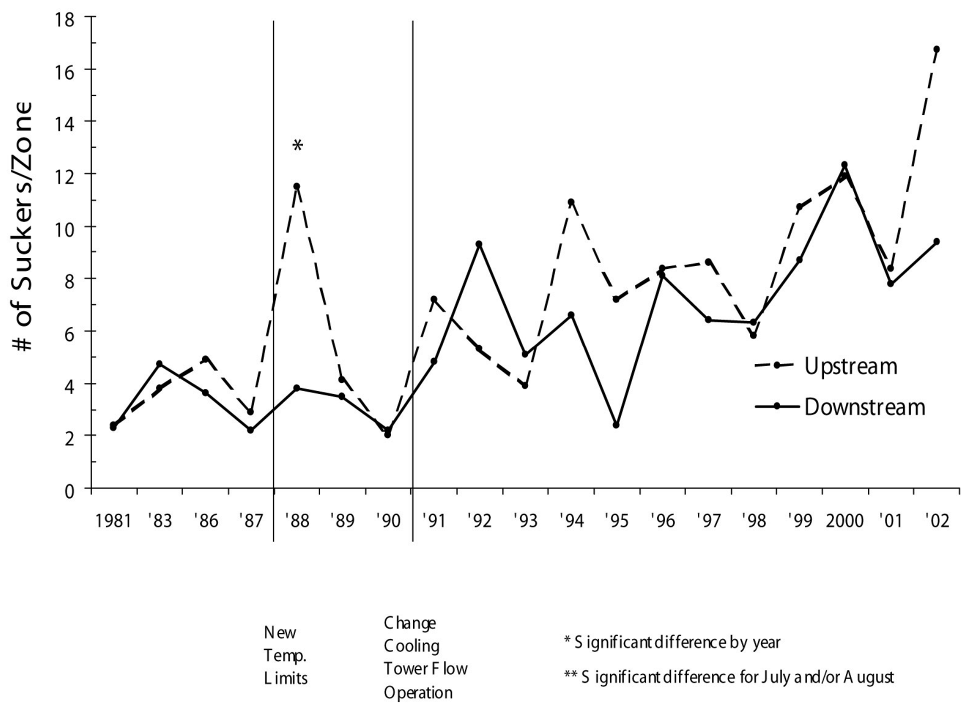

Figure 9-2
Number of Round-Bodied Suckers at Zones Upstream and Downstream from Cayuga Station, 1981–2002

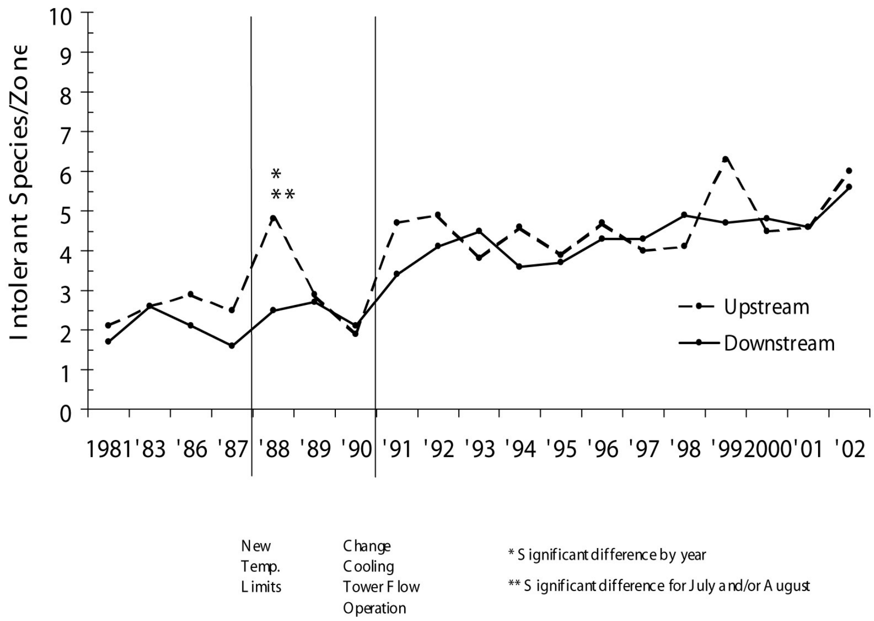

Figure 9-3
Number of Intolerant Species at Zones Upstream and Downstream from Cayuga Station, 1981–2002

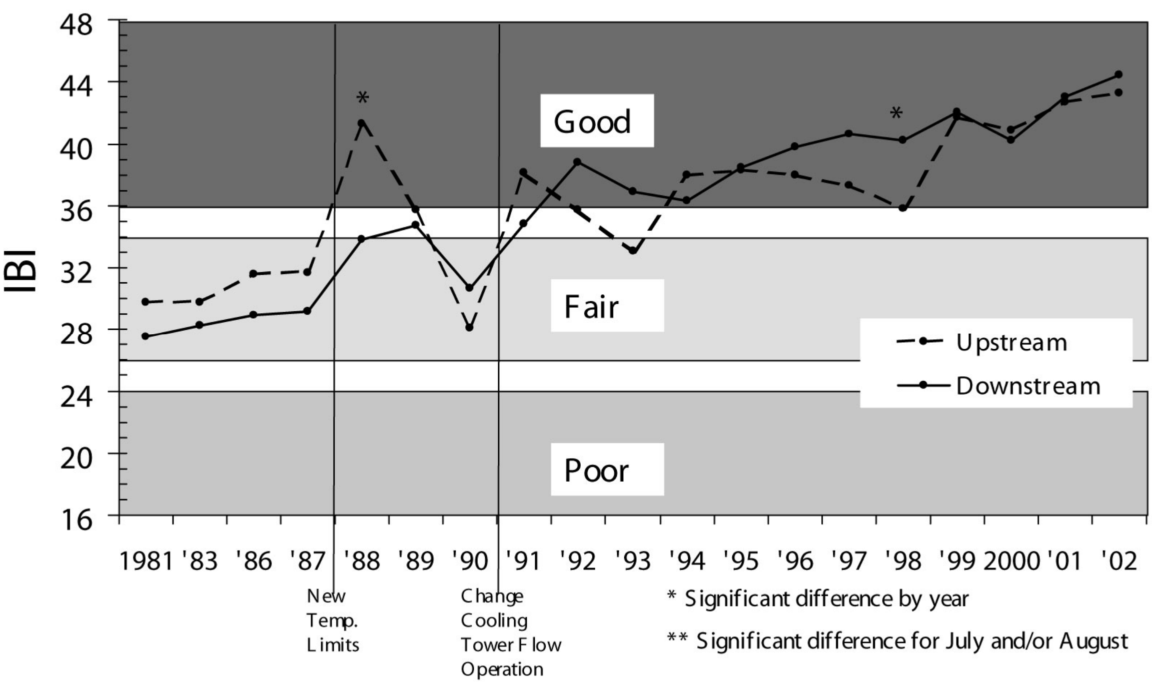

Figure 9-4
Index of Biotic Integrity Scores at Zones Upstream and Downstream from Cayuga Station, 1981–2002

The upstream/downstream Index of Well-Being (IWB) values exhibited trends similar to the IBI scores (Table 9-5). The IWB values were similar at the upstream/downstream zones during most of 1981–1988, but increased substantially at the downstream zones during the subsequent years and were significantly higher at the downstream zones during three of the years. From 1981 to 2002, the IWB scores also showed a gradual improvement, and the community health classification went from "Fair" to "Excellent" at upstream and downstream zones. When comparing the IWB values in blocks of years and during all years, there were no significant differences (Table 9-3).

Table 9-3
Cayuga Station Upstream vs. Downstream IWB Statistical Comparisons, 1981–2002

Comparison/Parameter	Upstream Mean	Downstream Mean	Significant Difference[a]	F Value	P Value
All Years and Passes Combined [b]	8.1	8.1	No	0.41	0.52
All Years Combined - July/August Passes [b]	8.1	8.0	No	0.00	0.95
1981–1986 Combined - July/August Passes	5.8	6.2	No	1.10	0.30
1987–1990 Combined - July/August Passes [b]	7.4	7.3	No	0.16	0.69
1991–2002 Combined - July/August Passes [b]	8.8	9.0	No	2.88	0.09

(a) Results of one-factor parametric Analysis of Variance tests (alpha=0.05).

(b) Data ranks used for statistical analyses as raw data and log transformed data are not normally distributed.

One of the reasons for the generally lower round-bodied sucker, intolerant species, and IBI numbers downstream from the station was the substantially lower values at discharge Zone 4. This zone was forced to be located in the immediate station discharge area because of another industrial discharge situated just a short distance downstream. During low-river flow conditions, near-field Zone 4 consists predominantly of heated cooling water prior to being mixed with the Wabash River. When comparing the IBI and IWB scores among the upstream, near-field downstream (i.e., Zone 4) and far-field downstream zones, the near-field zone had substantially lower scores than the other areas (Table 9-4). These differences were significant during some blocks of years and for all years combined. In contrast to the near-field downstream differences, the far-field downstream zones had significantly higher IBI and IWB scores than at the upstream zones for the 1991–2002 period.

When comparing IBI and IWB values near Cayuga Station over time, there were significant differences (increasing numbers) among the blocked years (Table 9-5). In addition to comparing all years and passes, the July/August samplings and the July/August samplings using only matched zones were compared and showed similar differences. These increased scores over time are probably due to a combination of improved water quality in the Wabash River and possibly to improved collecting techniques. Gammon [3] also reported water quality and fish community improvements in the middle Wabash River during this time.

Table 9-4
Results of Upstream vs. Near-Field vs. Far-Field Statistical Comparisons for Electrofishing Catch Indices Collected at the Same Sampling Zones Near Cayuga Station, July/August 1981–2002

Comparison/Parameter	Upstream Mean	Near-Field Mean	Far-Field Mean	Significant Difference (a)	F Value	P Value
All Years Combined						
IBI (b)	36.5	32.0	36.8	Yes	5.07	0.01
	A	B	A(c)			
IWB (b)	8.5	7.9	8.4	Yes	4.25	0.02
	AB	B	A			
1981–1986 Combined						
IBI	29.0	23.0	26.3	Yes	3.99	0.0497
	A	B	AB			
IWB	5.8	6.2	6.2	No	0.21	0.81
1987–1990 Combined						
IBI(c)	34.9	28.3	32.6	No	0.95	0.40
IWB	8.0	7.3	7.2	No	0.73	0.49
1991–2002 Combined						
IBI (b)	37.2	35.0	40.2	Yes	7.00	<0.01
	B	B	A			
IWB (b)	8.8	8.4	9.2	Yes	11.33	<0.01
	B	B	A			

(a) Results of one-factor parametric Analysis of Variance tests (alpha=0.05).
(b) Data ranks used for statistical analyses because raw data and log transformed data are not normally distributed.
(c) Results of Tukey's Studentized Range Test; values with the same letters are not significantly different (alpha=0.05).
(d) Results of Fisher's LSD Test; values with the same letters are not significantly different (alpha=0.05).
(e) Log transformed data used for statistical analyses because they are normally distributed.

Wabash River Station

The number of round-bodied suckers at Wabash River Station (WRS) was variable between upstream and downstream zones during 1982–2002, but was significantly greater upstream during the drought of 1988 prior to the station having downstream temperature limits (Figure 9-5). Similar to what was observed at Cayuga Station, the round-bodied suckers also showed a general increase at WRS from 1982 to 2002. However, the overall numbers were generally lower at WRS than at Cayuga Station during these years.

Table 9-5
Cayuga Station Period vs. Period Statistical Comparisons, 1981–2002

Comparison/Parameter	1981–1986 Mean	1987–1990 Mean	1991–2002 Mean	Significant Difference[(a)]	F Value	P Value
All Zones and Passes						
IBI[(b)]	29.1	32.9	38.8	Yes	98.21	<0.01
	C	B	A[(c)]			
IWB[(b)]	6.7	7.5	8.9	Yes	170.17	<0.01
	C	B	A			
All Zones and July/August Passes						
IBI[(b)]	27.4	31.6	38.3	Yes	50.01	<0.01
	C	B	A			
IWB[(b)]	6.0	7.4	8.9	Yes	86.00	<0.01
	C	B	A			
Matched Zones and July/August Passes						
IBI[(b)]	25.7	32.3	38.0	Yes	27.95	<0.01
	C	B	A			
IWB[(b)]	6.1	7.5	8.9	Yes	34.36	<0.01
	C	B	A			

(a) Results of one-factor parametric Analysis of Variance tests (alpha=0.05).

(b) Data ranks used for statistical analyses as raw data and log transformed data are not normally distributed.

(c) Results of Tukey's Studentized Range Test; values with the same letters are not significantly different (alpha=0.05).

The number of intolerant species tended to be higher at the upstream zones than the downstream zones near WRS during 1982–2002 and were significantly higher during the July/August 1997 (Figure 9-6). The reason for the lower downstream scores throughout this period was related to the poorer far-field sampling habitat, which will be discussed below. The intolerant species also demonstrated gradual increases in numbers from 1982 to 2002.

The IBI exhibited similar values upstream and downstream from WRS during 1982–2002 although two years had significantly higher upstream IBI scores (Figure 9-7). Neither the temperature limits initiated in 1988 nor the Energy Emergency of 1999 appeared to have any substantial impacts on the IBI scores. Like the IBI metrics described above, the IBI scores.showed an improvement from 1982 to 2002 with the community health going from "Poor" to "Good". When comparing IBI values during various time blocks and river flow regimes, the only significant differences were for all years combined, for all years only during July/August, and for high river flow years only during July/August (Table 9-6).

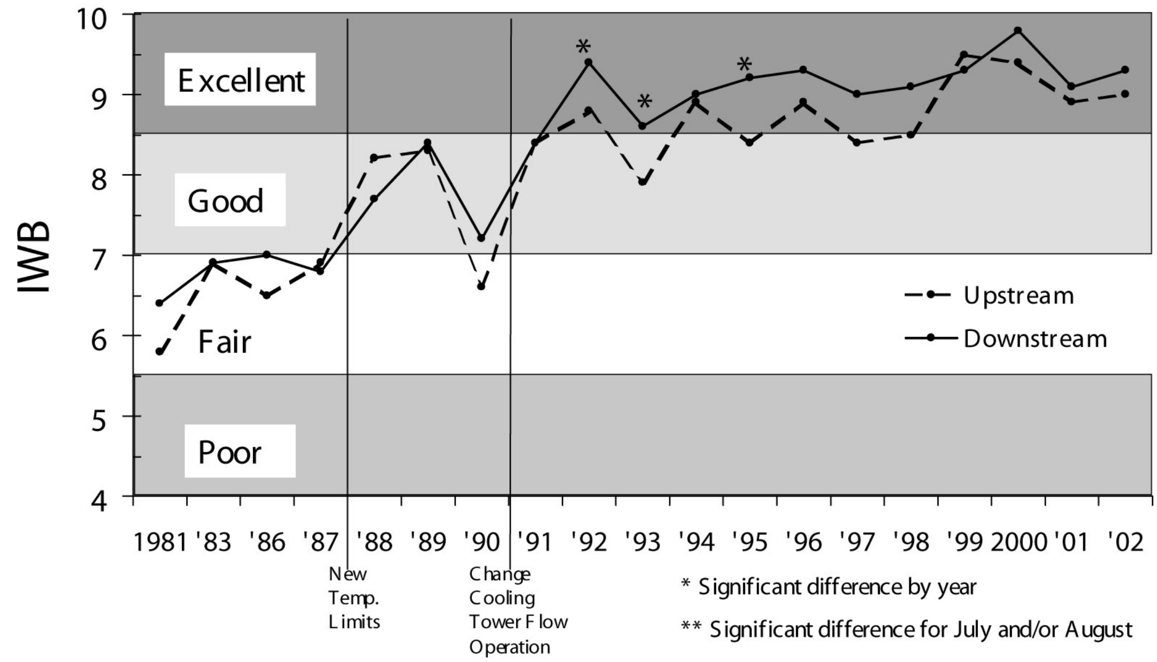

Figure 9-5
Index of Well-Being scores at zones upstream and downstream from Cayuga Station, 1981–2002

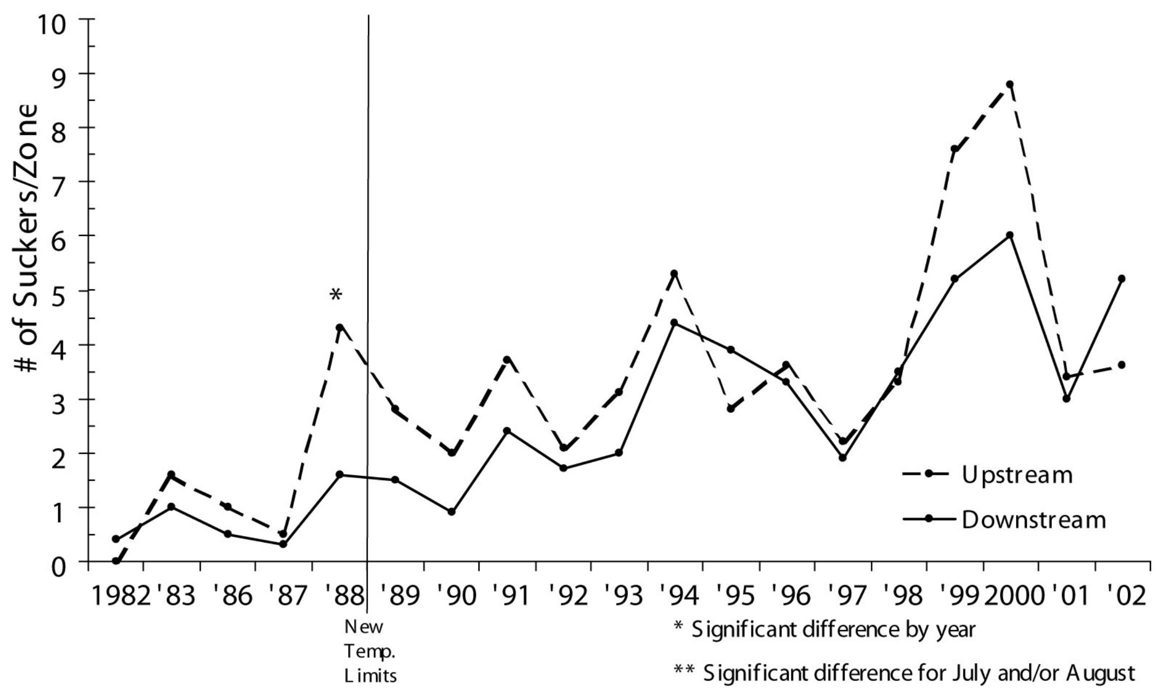

Figure 9-6
Number of Round-Bodied Suckers at Zones Upstream and Downstream from Wabash River Station, 1982–2002

The new temperature limits at WRS initiated in 1988 appeared to have improved the downstream IWB scores in the subsequent years (Figure 9-8). The upstream/downstream differences were very small from 1988 to 2002 with only 2001 have a significantly higher upstream mean IWB value. The IWB scores also improved from 1982 to 2002, with the community health classification going from “Fair” to “Excellent”. Like the IBI values and the individual IBI metrics, the IWB scores at WRS were consistently lower than at Cayuga Station during 1982–2002. When comparing IWB values during various time blocks and river flow regimes, the only significant difference was for all years combined (Table 9-7).

The main reason for the generally lower round-bodied sucker and intolerant species numbers downstream from the station was the substantially lower values at the far-field downstream zones. The near-field discharge zone has good habitat matching what is found at the upstream zone; however, the far-field zones located near Terre Haute have much poorer fish habitat and some industrial dischargers present. When comparing the round-bodied suckers, intolerants species, IBI and IWB values among the upstream, near-field downstream and far-field downstream zones for July/August samplings during 1991–2002, the far-field zones had substantially lower scores than the other areas (Table 9-8). These differences were significant during some blocks of years and for all years combined. It should also be noted that the near-field downstream zone had significantly higher tolerant species, IBI and IWB values than at the upstream zones indicating that the far-field fish community differences were related to the poorer fish habitat.

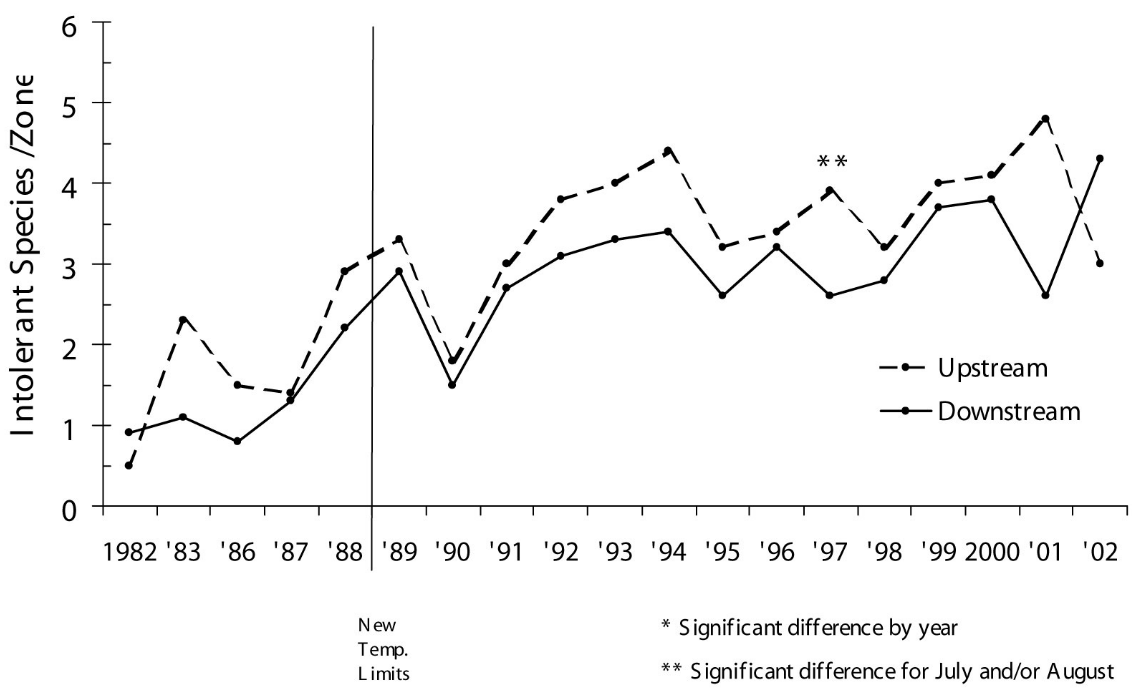

Figure 9-7
Number of Intolerant Species at Zones Upstream and Downstream from Wabash River Station, 1982–2002

Table 9-6
Wabash River Station IBI Upstream vs. Downstream Statistical Comparisons, 1982–2002

Comparison/Parameter	Upstream Mean	Downstream Mean	Significant Difference [a]	F Value	P Value
All Years and Passes Combined [b]	35.3	33.0	Yes	13.11	<0.01
All Years Combined - July/August Passes [b]	35.9	33.4	Yes	6.00	0.02
1982–1987 Combined - July/August Passes	26.8	26.5	No	0.01	0.92
1989–1993 Combined - July/August Passes	35.8	34.1	No	0.92	0.34
1994–1997 Combined - July/August Passes	37.0	34.7	No	1.81	0.19
1999–2002 Combined - July/August Passes	40.0	37.9	No	1.39	0.25
Low Flow Years Combined - July/August Passes	36.9	35.0	No	0.63	0.43
High Flow Years Combined - July/August Passes	34.7	31.7	Yes	4.85	0.03

(a) Results of one-factor parametric Analysis of Variance tests (alpha=0.05).

(b) Data ranks used for statistical analyses as raw data and log transformed data are not normally distributed.

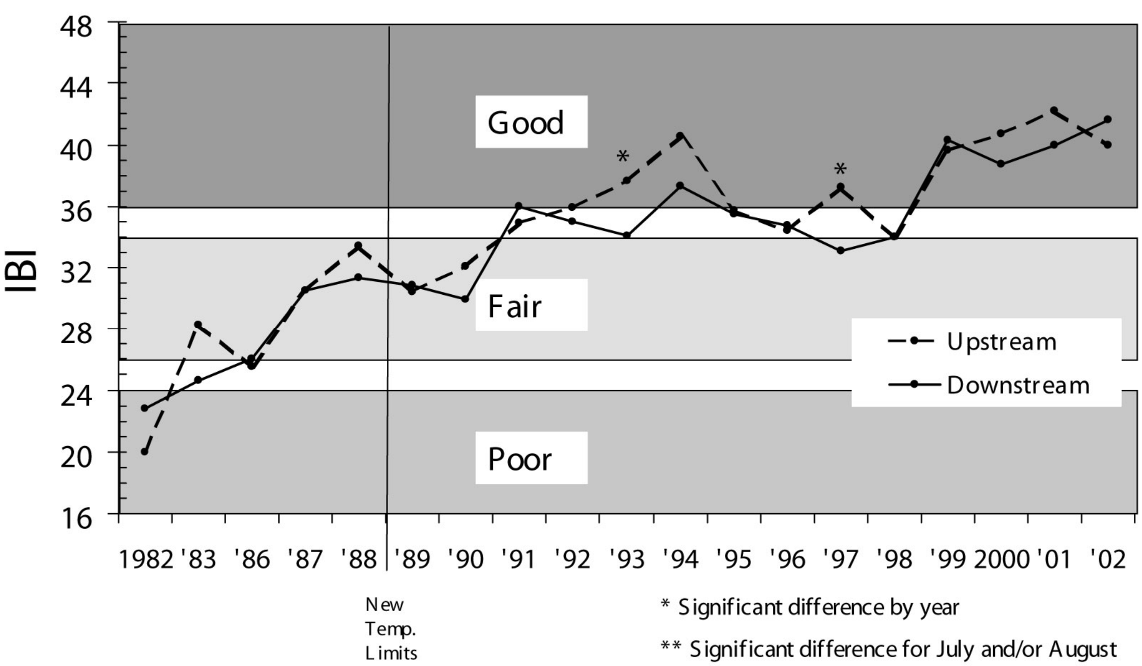

Figure 9-8
Index of Biotic Integrity Scores at Zones Upstream and Downstream from Wabash River Station, 1982–2002

Table 9-7
Wabash River Station IWB Upstream vs. Downstream Statistical Comparisons, 1982–2002

Comparison/Parameter	Upstream Mean	Downstream Mean	Significant Difference [a]	F Value	P Value
All Years and Passes Combined [b]	8.0	7.6	Yes	4.82	0.03
All Years Combined - July/August Passes [b]	8.1	7.7	No	1.27	0.26
1982–1987 Combined - July/August Passes	6.2	5.7	No	0.64	0.43
1989–1993 Combined - July/August Passes [b]	8.0	7.9	No	0.09	0.77
1994–1997 Combined - July/August Passes	8.5	8.5	No	0.00	0.97
1999–2002 Combined - July/August Passes	8.9	8.7	No	0.59	0.45
Low Flow Years Combined - July/August Passes [b]	8.2	8.0	No	0.10	0.76
High Flow Years Combined - July/August Passes [b]	8.0	7.4	No	1.45	0.23

(a) Results of one-factor parametric Analysis of Variance tests (alpha=0.05).
(b) Data ranks used for statistical analyses as raw data and log transformed data are not normally distributed.

When comparing IBI and IWB values near WRS over time, there were significant differences (increasing numbers) among the blocked years (Table 9-9). In addition to comparing all years and passes, the July/August samplings and the July/August samplings using only matched zones were compared and showed similar differences. As mentioned above, these increased scores over time are probably due to a combination of improved water quality in the Wabash River and possibly to improved collecting techniques. The water quality and fish community improvements in the middle Wabash River during this time have been noted by other researchers. Gammon attributed these improvements primarily to long-term 50 percent reductions in BOD loading by municipal and industrial treatment plants [3].

Combined Effects of the Two Stations

Because Cayuga and Wabash River stations are separated by approximately 35 river miles, it has been suggested that there are combined impacts of the two stations downstream from Wabash River Station. Comparisons of the zones upstream and downstream from both generating stations during all years and samplings revealed significant differences for round-bodied suckers, intolerant species, IBI and IWB numbers (Table 9-10). However, when comparing the upstream/downstream differences for all years during July/August, all years during July/August with matched zones, and only low river flow years during July/August samplings, the differences going from upstream to downstream were variable and were not significantly different. The absence of significant differences during the July/August periods is unusual as this typically would be worst-case conditions for thermal impacts on the fish community.

Table 9-8
Results of Upstream vs. Near-Field vs. Far-Field Statistical Comparisons for Electrofishing Catch Parameters Collected at the Same Sampling Zones Near Wabash River Station, July/August 1991–2002

Comparison/Parameter	Upstream Mean	Near-Field Mean	Far-Field Mean	Significant Difference[(a)]	F Value	P Value
All Years Combined						
IBI[(b)]	38.3	41.9	34.0	Yes	15.89	<0.01
	B	A	C©			
IWB[(b)]	8.6	9.0	8.3	Yes	6.95	<0.01
	B	A	B			
No. Round-Bodied Suckers[(b)]	3.8	4.7	2.4	Yes	6.65	<0.01
	A	A	B			
No. Intolerant Species[(b)]	3.9	4.7	2.4	Yes	19.83	<0.01
	B	A	C			
1991–1993 Combined						
IBI	38.3	42.4	31.1	No	2.28	0.12
IWB	8.3	8.7	8.4	No	1.10	0.35
No. Round-Bodied Suckers[(b])]	2.7	2.2	1.6	No	0.87	0.43
No. Intolerant Species[(b)]	3.8	4.8	3.0	No	2.92	0.07
1994–1997 Combined						
IBI	37.0	41.8	31.1	Yes	14.00	<0.01
	A	A	B			
IWB	8.5	9.3	8.1	Yes	9.89	<0.01
	B	A	B			
No. Round-Bodied Suckers[(b)]	3.3	5.1	2.1	Yes	4.21	0.02
	AB	A	B			
No. Intolerant Species[(b)]	3.9	4.8	1.8	Yes	17.16	<0.01
	A	A	B			
1999–2002 Combined						
IBI	40.0	41.7	36.0	Yes	3.37	0.05
	A	A	B[(d)]			
IWB[(b)]	8.9	8.9	8.6	No	0.72	0.49
No. Round-Bodied Suckers[(e)]	5.5	6.2	3.7	No	1.95	0.16
No. Intolerant Species	3.8	4.7	2.8	No	2.96	0.07

(a) Results of one-factor parametric Analysis of Variance tests (alpha=0.05).
(b) Data ranks used for stat. analyses as raw and log transformed data are not normally distributed.
(c) Results of Tukey's Studentized Range Test; values with the same letters are no significantly different (alpha=0.05).
(d) Results of Fisher's LSD Test; values with the same letters are not significantly different (alpha=0.05).
(e) Log transformed data used for statistical analyses because they are normally distributed.

Table 9-9
Results of Period vs. Period Statistical Comparisons for Electrofishing Catch Parameters Collected Upstream and Downstream of the Wabash River Station, 1982–2002

Comparison/Parameter	1982–1988 Mean	1989–1993 Mean	1994–1998 Mean	1999–2002 Mean	Significant Difference[a]	F Value	P Value
All Zones and Passes							
IBI[b]	28.4	33.4	36.0	40.3	Yes	64.42	<0.01
	D	C	B	A[c]			
IWB[b]	6.2	7.8	8.5	8.9	Yes	131.80	<0.01
	D	C	B	A			
All Zones and July/August Passes[d]							
IBI[b]	26.6	34.8	35.8	38.9	Yes	18.73	<0.01
	C	B	AB	A			
IWB[b]	5.8	7.9	8.5	8.8	Yes	38.77	<0.01
	C	B	A	A			
Matched Zones and July/August Passes[d]							
IBI[b]	27.9	36.1	39.0	41.0	Yes	10.47	<0.01
	B	A	A	A			
IWB[b]	6.4	8.0	8.8	8.9	Yes	26.87	<0.01
	C	B	A	A			

(a) Results of one-factor parametric Analysis of Variance tests (alpha=0.05).
(b) Data ranks used for statistical analyses as raw data and log transformed data are not normally distributed.
(c) Results of Tukey's Studentized Range Test; values with the same letters are not significantly different (alpha=0.05).
(d) 1988 and 1998 are omitted from July/August analyses because no data were collected in those months.

Gammon also noted a downstream decline of the fish community in the river reach extending from upstream from Cayuga downstream to Terre Haute [3]. Gammon suggested that this decline may be related to the active and derelict coal mines only found in this river reach and their negative impacts on water quality. Simon and Stahl also noted a slight decreasing trend in IBI values in the Wabash River with increasing drainage [11]. The ecoregions in Indiana also change in this area upstream from Cayuga Station from Central Corn Belt to Interior River Lowland [12] and likely could result in fish habitat changes in the river causing these fish community differences.

The lack of upstream/downstream fish community differences in this river reach during worst-case conditions (i.e., high summer temperatures and low river flow), the overall upstream/downstream declining fish community trend noted by other researchers on the Wabash, and the change in ecoregion classification immediately upstream from this river reach all indicate that the upstream downstream differences noted at the Cayuga and Wabash River stations' river reach is not station related.

Table 9-10
Results of Area vs. Area Statistical Comparisons for Electrofishing Catch Parameters Collected Upstream and Downstream of the Cayuga and Wabash River Stations, 1983–2002

Comparison/Parameter	Upstream Cayuga Mean	Downstream Cayuga Mean	Upstream WRS Mean	Downstream WRS Mean	Significant Difference (a)	F Value	P Value
All Years and Passes Combined (b)							
IBI (c)	36.4	35.2	35.6	33.3	Yes	8.54	<0.01
	A	A	A	B (d)			
IWB (c)	8.2	8.3	8.0	7.7	Yes	8.94	<0.01
	AB	A	BC	C			
No. Round-Bodied Suckers (c)	7.1	5.2	3.5	2.3	Yes	27.35	<0.01
	A	B	B	C			
No. Intolerant Species (c)	4.0	3.3	3.3	2.5	Yes	18.17	<0.01
	A	B	AB	C			
All Years - July/August Passes (b,c)							
IBI (c)	35.6	34.5	36.3	33.9	No	2.18	0.09
IWB (c)	8.1	8.1	8.2	7.8	No	0.92	0.43
All Years - Matched Zones July/August Passes (b,c)							
IBI (c)	36.3	35.5	36.1	38.5	No	1.40	0.25
IWB (c)	8.5	8.3	8.1	8.5	No	2.64	0.0505
Low Flow Years - July/August Passes (f)							
IBI (c)	36.0	34.1	36.9	35.0	No	0.66	0.58
IWB (c)	8.2	8.1	8.2	8.0	No	0.23	0.88
No. Round-Bodied Suckers (c)	6.4	5.2	4.2	2.9	No	0.76	0.52
No. Intolerant Species (c)	3.9	2.9	3.2	2.7	No	1.24	0.30

(a) Results of one-factor parametric Analysis of Variance tests (alpha=0.05).
(b) 1981 and 1982 omitted from analyses because both plants were not sampled.
(c) Data ranks used for statistical analyses because raw data and log transformed data are not normally distributed.
(d) Results of Tukey's Studentized Range Test; values with the same letters are not significantly different (alpha=0.05).
(e) 1988 and 1998 are omitted from analyses because no data were collected in July or August near WRS.
(f) 1988 is omitted from analyses because no data were collected in July or August near WRS.

Summary

Cayuga Station

- Biotic indices exhibited an increase from 1981 to 2002 both upstream and downstream from Cayuga Station. The increase was significant among blocked periods.
- New temperature limits in 1987 and changes in cooling tower operation appear to have improved downstream biotic index scores and the intolerant species metric.
- Downstream zones had significantly lower downstream IBI scores for all samplings, but not for just July/August or other blocked periods. IWB showed no difference.

Near-field biotic index scores were significantly lower than upstream or far-field downstream scores. This near-field difference was likely related to the impact of unmixed station discharge water at this zone.

Wabash River Station

- Biotic indices exhibited an increase from 1982 to 2002 both upstream and downstream from Wabash River Station. The increase was significant among blocked periods.
- Changes in the IWB, but not IBI, indicate that the new temperature limits in 1988 improved the downstream fish community.
- Downstream biotic indices were significantly lower for all sampling combined, but generally not for July/August passes or for individual periods.
- Fish habitat differences among zones were evident as downstream near-field indices were higher than the upstream zones, but substantially lower at the far-field downstream zones.
- All biotic indices were slightly lower at Wabash River Station than at upstream Cayuga Station during the 1982–2002 period.

Combined Effects of the Two Stations

- Biotic indices had significant upstream to downstream declines near the two station river reaches for combined data, but not for the July/August samplings or for low river flow years.
- A similar upstream to downstream decrease in biotic index scores and a change in river habitat have also been documented in the Middle Wabash River by other researchers.

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10

APPLICATION OF MULTIMETRIC BIOASSESSMENT TECHNIQUES IN A 316(A) DEMONSTRATION AT GEORGIA POWER COMPANY'S PLANT BRANCH, LAKE SINCLAIR, GEORGIA

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Introduction

This paper presents the application and findings of a reservoir multimetric bioassessment conducted by Georgia Power Company (Georgia Power) on Lake Sinclair for the purpose of developing a Clean Water Act (CWA) Section 316(a) demonstration to obtain a thermal variance for Plant Branch. Plant Branch is an electric generating facility that withdraws water from Lake Sinclair, a 15,330-acre [6,204 hectare (ha)] man-made reservoir, for cooling purposes and discharges heated effluent to the reservoir under the authority of an NPDES permit. Georgia Power began operating a new cooling tower system in 2002 designed to remove approximately one-half of the thermal output from the plant during summer months and is conducting a two-year scientific investigation, as set forth in an approved Study Plan, to determine if a balanced, indigenous aquatic community is protected and maintained in the thermally influenced portion of Lake Sinclair designated as the Primary Study Area.

In addition to the bioassessment, other filed studies included reservoir-wide water quality and temperature monitoring, habitat assessments, conventional macroinvertebrate and fish community assessments, and aquatic macrophyte surveys. The data gathered in these studies are being used for comparisons between the Primary Study Area and non-thermally influenced reference areas within Lake Sinclair. The second year of field sampling was completed in December 2003. This paper presents the results of the macroinvertebrate and fish community multimetric bioassessment for 2002.

Study Background

Georgia Power owns and operates Plant Branch, which is located adjacent to Lake Sinclair, an impoundment of the Oconee River, near Milledgeville, Georgia (Figure 10-1). Plant Branch is a coal-fired electric power generating facility consisting of four generating units with a total design rating of approximately 1,540 megawatts (MW). Units 1 through 4 utilize once-through cooling water withdrawn from the Little River arm of Lake Sinclair and discharged through NPDES-permitted Outfalls 01A and 02A into a constructed discharge basin and then to the Beaverdam Creek embayment of Lake Sinclair. The Beaverdam Creek embayment consists entirely of impounded, reservoir water and does not contain any free-flowing reach of stream.

On 30 November 1995, the Georgia Department of Natural Resources (GDNR), Environmental Protection Division (EPD), renewed the Plant Branch NPDES permit. Concurrently, EPD issued an Order which directed Georgia Power to evaluate operational measures for providing non-lethal conditions for fish in the Beaverdam Creek embayment of Lake Sinclair during the summer. Georgia Power subsequently developed plans for a cooling tower system to reduce the heat load discharged to Lake Sinclair and provide non-lethal conditions in Beaverdam Creek embayment during summer months, which tend to pose the most critical ambient conditions for aquatic biota from the standpoint of elevated water temperature. The goals for establishing non-lethal conditions for fish can be met through summertime operation of the cooling tower system, which was installed and initially operated during summer 2002. Georgia Power believes that the State's water temperature criteria are more stringent than necessary to protect and maintain a balanced, indigenous aquatic community in Lake Sinclair and is seeking a Section 316(a) thermal variance (i.e., alternate thermal limits) for Plant Branch.

During the NPDES permit renewal process, Sierra Club and the U.S. Environmental Protection Agency (USEPA), Region 4, raised concerns related to the facility's thermal discharge. To resolve these issues, Georgia Power entered into a Memorandum of Agreement (MOA) with EPD, USEPA, and the Sierra Club (the MOA participants). The MOA requires Georgia Power to complete specific tasks related to the Section 316(a) demonstration to justify alternate thermal limits. On 14 December 2000, EPD issued a new NPDES permit to Plant Branch that required installation of the cooling system and the conduct of this Section 316(a) demonstration study and other related studies.

Section 316(a) Demonstration Study Plan

As required by the MOA, Georgia Power developed a Section 316(a) Demonstration Study Plan for review and comment by the MOA participants, including the GDNR Wildlife Resources Division (WRD). The Study Plan set forth the study design and detailed methods for determining scientifically whether a balanced, indigenous aquatic community exists in Lake Sinclair, specifically in the area designated as the Primary Study Area [1]. As specified in the MOA, the Study Plan was developed "in substantial conformance" with the 1 May 1977 USEPA guidance document entitled "Draft Interagency 316(a) Technical Guidance Manual and Guide for Thermal Effects Sections of Nuclear Facilities Environmental Impact Statements" (Guidance Manual) [2]. Georgia Power addressed the review comments received by the MOA participants and EPD subsequently approved the Study Plan in a December 2001, prior to initiation of the first year of field studies.

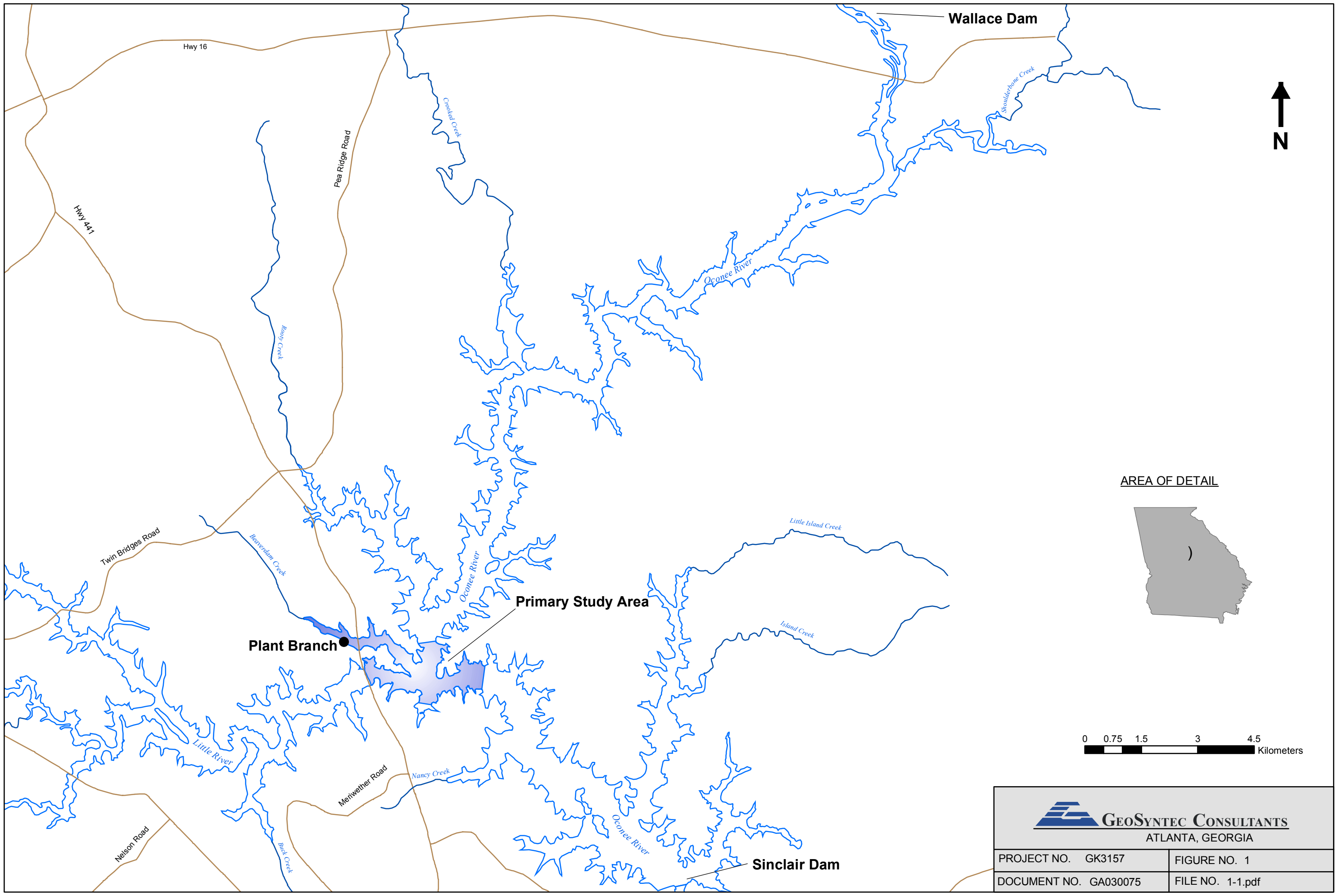

Figure 10-1
Lake Sinclair, Near Milledgeville, Georgia

Georgia Power's goal through the implementation of the Study Plan is to determine: 1) if the characteristics of a balanced, indigenous aquatic community exist within the Primary Study Area of Lake Sinclair; or 2) if there is evidence of previous harm, that the protection and propagation of a balanced, indigenous aquatic community will nevertheless be protected and assured under alternate thermal limits combined with summer operation of the Plant Branch cooling system.

Plant Branch

Plant Branch is a four-unit, coal-fired steam electric generating station with a nameplate generating capacity of 1,540 megawatts (MW). The plant occupies a 716-acre site on a peninsula between the Little River arm of Lake Sinclair and the Beaverdam Creek embayment (Figure 10-1). Units 1 through 4 utilize once-through cooling with water drawn in from the Little River arm of Lake Sinclair. After passing through the plant, the heated water discharges into the Beaverdam Creek embayment.

Commercial operation of Plant Branch began with the startup of Unit No. 1 in June 1965. The remaining three units began operation in 1967, 1968, and 1969. Historically annual net generation at Plant Branch has ranged from less than 5 million megawatt hours (MWH) in 1979, to more than 10 million MWH in 1985, 1989, and 1990 [3].

Cooling System Design and Operation

Georgia Power began operating a new mechanical draft cooling tower system during summer 2002 to provide auxiliary cooling of the Plant Branch cooling water discharge and establish non-lethal conditions for reservoir fish in the Beaverdam Creek embayment. Non-lethal conditions were defined by EPD in the Plant Branch NPDES permit as measuring a water temperature of not greater than 93°F (33.9°C) at a depth somewhere in the vertical water column that contains dissolved oxygen concentrations of not less than 3.0 milligrams per liter (mg/L).

The cooling tower system is a conventional mechanical draft, counter-flow type that, at times, utilizes electric motor-driven fans to provide airflow through the system. The cooling system operates only during the summer months and is designed to remove up to one-half of the BTU thermal heat load in the effluent before it is discharged to the Beaverdam Creek embayment. A beneficial side effect of the cooling system is that the discharges are typically saturated with dissolved oxygen. This addition of dissolved oxygen is beneficial to aquatic life in Lake Sinclair, particularly during summer months when saturated dissolved oxygen conditions in reservoirs are atypical.

The cooling system is located on the west side of the Plant Branch powerhouse, and discharges the cooled water to the Beaverdam Creek embayment upstream of the existing plant discharge tunnels. The cooling system discharge provides non-lethal conditions for fish by creating a thermally stratified condition in Beaverdam Creek embayment where the oxygenated cooler water is in the lower portion of the water column.

Georgia Power designed the discharge structure using a computer-based, three-dimensional hydrothermal model of Beaverdam Creek embayment to maximize the stratification effect. The discharge structure is located parallel to the south bank of the Beaverdam Creek embayment and discharges cooled water near the bottom.

To further enhance thermal stratification of the Beaverdam Creek embayment, a discharge basin with overflow weir openings was constructed to enclose the two existing plant discharge tunnels. The purpose of the weir is to force the plant discharge flow that does not enter the cooling tower to the surface of Beaverdam Creek embayment so that a cooler water wedge can be established at depth.

Lake Sinclair

Lake Sinclair is a 15,330-acre multi-purpose impoundment located on the Oconee River in central Georgia, north of the city of Milledgeville (Figure 10-1). The lake serves as a source for public drinking water supply, cooling water supply, hydroelectric power production, pumped-storage operation, and water-based recreation. Lake Sinclair was created in 1952 upon completion of Sinclair Dam, which Georgia Power owns and operates as a 45-MW hydroelectric peaking facility. At normal pool, the reservoir provides about 330,000 acre-feet (108 billion gallons) of storage capacity. Maximum depth is about 21.5 meters (m).

Lake Sinclair is a moderately dendritic reservoir with approximately 417 miles (671 km) of shoreline at normal pool elevation of 340 feet (ft) plant datum (103.6 m). The reservoir has two major tributary arms (Oconee and Little River) and several small tributary embayments, including Beaverdam Creek, Rooty Creek, Crooked Creek, Shoulderbone Creek, Little Island Creek, Island Creek, Buck Creek, and Cedar Creek (Figure 10-1).

The area around Lake Sinclair is located approximately 400-ft above sea level. Underlying crystalline rock formations are predominant, while soils are generally red with sandy clay and silty-mica clay textures. Lake Sinclair is located within the Southern Outer Piedmont (45b) eco-region of Georgia [4].

316(a) Demonstration Study Approach

The Section 316(a) demonstration study was designed to evaluate the status of the biological community within the MOA-designated and thermally influenced Primary Study Area of Lake Sinclair relative to the aquatic community found in other areas of the reservoir that are not thermally influenced by the Plant Branch discharge. This was accomplished by first establishing a definition of non-thermal areas (i.e., thermal reference conditions) followed by implementation of bioassessment techniques involving the comparison of biological community attributes of the Primary Study Area to those of the reference areas. An essential element of the Study Plan is the *a priori* establishment of bioassessment performance levels (i.e., biological integrity ratings) agreed to by the MOA participants that, if met or exceeded, confirmed the presence of a balanced, indigenous aquatic community in the Primary Study Area.

The following sections outline the principal concepts being applied in this demonstration study. Detailed methodology is described in the approved Study Plan [1].

Study Scope

The exclusive anthropogenic activity of interest for this study is the Plant Branch thermal discharge. Adverse impacts to the biological community within Lake Sinclair, including the Primary Study Area, resulting from other perturbations in the watershed, while important, were deemed outside the scope of the 316(a) demonstration study. Based on the premise that upstream watershed perturbations, such as those associated with land use changes (i.e., land disturbance and development activities), affect aquatic biological communities on a relatively broad scale compared to single point-source discharges, the relative impact of the point-source thermal discharge to the biological community can be determined. The condition of biological communities present outside of the thermal influence of the Plant Branch discharge, though potentially adversely affected by other watershed activities, nonetheless represents the "standard" or "reference" to which the biological community within the Primary Study Area was comparatively evaluated. If the biological community subjected to both the thermal and other watershed perturbations potentially present within the Primary Study Area compares favorably (i.e., meets or exceeds performance levels) to the reference area community subjected solely to watershed influences, as determined from the bioassessment and supporting empirical data, then the presence of a balanced, indigenous aquatic community expected for a reservoir is supported.

In the absence of an established reference condition for biological assemblages in reservoirs of the Piedmont eco-region of Georgia, Georgia Power developed a site-specific reference condition within Lake Sinclair. This reference condition was based on aquatic community data collected from multiple sites within Lake Sinclair determined to be unaffected by the thermal discharge from Plant Branch (see below). The reference condition formed the baseline to which biological community attributes measured in the Primary Study Area were compared and scored using multimetric bioassessment protocols [1].

Primary Study Area

The MOA defined the Primary Study Area based on the Section 316(a) Guidance Manual as: "the area of Lake Sinclair where the thermal discharge raises ambient temperature by at least 2° Centigrade one meter below the surface". Reservoirs usually exhibit three zones (riverine, transitional, and lacustrine), which correspond to flowing, river-like conditions; transition to lake conditions; and non-flowing, lake-like conditions near the dam, respectively [5]. The thermally influenced Primary Study Area encompasses approximately 732 acres (296 ha) of Lake Sinclair in the area of the confluence of the Oconee River and Little River, and lies within the transition zone of the 6,204-ha reservoir (Figure 10-1). The Primary Study Area includes the entire 206-acre (83-ha) Beaverdam Creek embayment; portions of the main body of Lake Sinclair adjacent to, and upstream and downstream of, the Beaverdam Creek embayment; and the lower portion of the Little River arm downstream of U.S. Highway 441. The Primary Study Area contains approximately 4.8 percent of the total surface area of Lake Sinclair.

Study Reference Areas

Bioassessment techniques use baseline or least-disturbed reference conditions as the basis for determining whether or not a waterbody meets expected ecological attributes at population and community levels. The USEPA defines a reference condition as "the chemical, physical, or biological quality or condition exhibited at either a single site or an aggregation of sites that represent the least impaired or reasonably attainable condition at the least impaired reference sites" [5]. A reference site is selected on "a waterbody, which represents the best attainable physical habitat, water chemistry, and biological parameters for specific environmental conditions."

Man-made reservoirs require a different approach than streams, rivers, or natural lakes for determining reference conditions. As artificial systems, there are no natural reference sites for determining biological community characteristics that would be expected in systems unaffected by human activity. The reference conditions applicable to this study were developed from biological community attributes measured at multiple sites within Lake Sinclair shown to be uninfluenced by the Plant Branch discharge during critical summer months. This was accomplished through the adaptation of USEPA's bioassessment methods used to determine reservoir reference conditions when evaluating the status of the biological community of an entire reservoir [5].

The suitability of using sampling sites within Lake Sinclair for reference purposes had been questioned by USEPA and other involved regulatory agencies in prior thermal variance requests for Plant Branch. For the current study, the absence of established reference conditions or biocriteria for reservoirs in Georgia lead to an analysis of the suitability of other southeastern reservoirs for providing comparable reference conditions, or demonstrating that areas existed in Lake Sinclair that were suitable for use in establishing reference conditions. Developing an eco-regional reference condition for reservoirs in the Piedmont physiographic province of Central Georgia was beyond the scope of the Plant Branch 316(a) demonstration.

Lake Sinclair and Lake Oconee are the only impoundments greater than 1,000 acres on the Oconee River. Lake Oconee is located immediately upstream of Lake Sinclair. Lake Oconee is 26 years younger than Lake Sinclair and has 37 percent more surface area and 41 percent more storage. If another reservoir were to be selected as comparable to Lake Sinclair for the purpose of establishing a reference condition for comparison to the Primary Study Area of Lake Sinclair, Lake Oconee would intuitively be the best candidate. And, importantly, Lake Oconee has no point source thermal discharges and is unaffected by the Plant Branch thermal discharge.

If a biological community reference condition existed for reservoirs located in the Piedmont physiographic province of Central Georgia, determining the status of the biological community in the Primary Study Area of Lake Sinclair would seem to be a straightforward process. A favorable comparison would support the presence of a balanced, indigenous community, and thus satisfy the 316(a demonstration. Alternatively, however, an unfavorable comparison would not necessarily indicate thermal stress as the source of impacts to the biological community in the Primary Study Area if areas outside the thermal influence of the discharge likewise compared unfavorably to the reference condition. Therefore, evaluation of the biological community attributes within the Primary Study Area compared to other areas in Lake Sinclair would still need to be undertaken to determine the relative thermal impacts to the biological community. It is solely the thermal component of anthropogenic perturbations that was targeted for evaluation during the 316(a) demonstration.

It is intuitive that the Plant Branch plume does not influence all of Lake Sinclair's 15,330 acres based on the location and nature of the discharge, the morphology of the reservoir and operation of the system. Ultimately it was determined to conduct continuous temperature monitoring outside of the Primary Study Area to demonstrate that areas existed in Lake Sinclair that are unaffected by the Plant Branch thermal discharge and as such, could serve as suitable reference areas for the 316(a) demonstration.

Criterion for "No Thermal Influence"

The Section 316(a) Guidance Manual indicates that the "reference ambient temperature" shall be recorded at a location agreed upon by the appropriate regulatory agency. The "reference ambient temperature" for this study was recognized as the water temperature monitored at the intake monitoring location "INT" at a depth of one meter as specified in the Plant Branch NPDES permit. The INT monitoring point was located outside of the Primary Study Area about two miles upstream from Beaverdam Creek embayment in the Oconee River arm of Lake Sinclair (see Section 4.1.2). Water temperatures measured in areas outside the Primary Study Area were compared to the reference ambient temperature to make a determination of "no thermal influence". Reference areas in Lake Sinclair are those areas considered as having no thermal influence attributable to Plant Branch when representative water temperatures measured during summer months at a depth of one meter outside of the Primary Study Area are not greater than 1.5° Celsius (C) [2.7 Fahrenheit (F)] above those temperatures similarly measured at the reference ambient temperature location (i.e., the NPDES permit-specified "INT" location). Georgia's water quality criteria for temperature applicable to Lake Sinclair[1] allow for an increase above ambient temperature of up to 2.8°C (5°F). Thus, Georgia Power's thermally based selection criterion for reference areas of 1.5°C (2.7°F) or less differential from the reference ambient temperature represents a conservative approach and one that assures the selection of appropriate reference areas in Lake Sinclair for the purposes of the demonstration.

The 95th percentile of the Daily Average Temperature (DAT) was used as the representative metric for the determination of potential references areas in Lake Sinclair. The DAT represents the mathematical mean of multiple, equally spaced temperatures measured over a 24-hour period. The 95th percentile DAT provides a reasonably conservative approach to allow for expected short-term diel variations in temperature to which fish are well adapted [6].

Bioassessment Performance Criteria

Georgia Power's demonstration study integrated major study elements of the Section 316(a) Guidance Manual with a contemporary bioassessment approach used by USEPA and numerous other resource agencies throughout the country, including Georgia EPD and WRD [4, 7].
The bioassessment approach provides a structurally definitive decision-making process for determining biological community impairment and has been used for developing numerically based biocriteria in several states [5]. Application of the bioassessment approach provides a rational, scientifically based method of determining the presence or absence of a balanced, indigenous aquatic community in the Primary Study Area of Lake Sinclair.

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[1] "Temperature: Not to exceed 90°F. At no time is the temperature of the receiving waters to be increased more than 5°F above intake temperature…" Georgia's Rules and Regulations for Water Quality Control Chapter 391-3-6(6)(b)(iv).

During the 2002 bioassessment of Lake Sinclair, field sampling data were collected and used to rate multiple attributes (biological metrics) describing the condition of the fish and benthic macroinvertebrate communities in the Primary Study Area. The individual metrics correlate either positively or negatively with increasing environmental degradation and were compared to the expected conditions of macroinvertebrate and fish communities in the non-thermally influenced areas of the reservoir, as represented by sampling data collected in the reference areas. The individual metric scores were then tallied to yield total index scores for the fish and benthic macroinvertebrate communities in the Primary Study Area. The total scores correspond to "biological integrity" classes, or ratings, describing the overall health and condition of the aquatic community. Although this bioassessment approach was applied seasonally, emphasis was placed on the late summer (i.e., August) benthic macroinvertebrate and fish community data sets that were collected during thermally critical summer conditions to calculate the biotic indices as measures of biological integrity.

Community ratings derived from the biotic indices fell into one of five biological integrity classes: very poor; poor; fair; good; or very good/excellent. EPD and WRD consider stream sites receiving integrity ratings of "fair" or better to be supporting their designated uses and "non-impaired" from a regulatory perspective. Georgia Power's bioassessment of Lake Sinclair adopted the same support/impairment criterion used by EPD and WRD in developing Performance Criteria for the Plant Branch 316(a) Demonstration Study. Hence, biotic communities of the Primary Study Area were considered "impaired" if resultant index scores (i.e. Performance Criteria") yielded biological integrity ratings of less than "fair." Conversely, if both the macroinvertebrate and fish communities of the Primary Study Area yield integrity ratings of "fair" or better, these results demonstrate that:

1. Biological integrity occurs in the Primary Study Area.
2. The Primary Study Area is not impaired.
3. The Primary Study Area is meeting designated uses.
4. A balanced, indigenous aquatic community exists in the Primary Study Area.
5. There is an absence of prior appreciable harm.
6. Alternate thermal discharge limits for the Plant Branch discharge are supported.

Study Implementation

Environmental Setting

Study conditions in 2002 were characterized by Plant Branch's highest net annual generation since 1985, above normal air temperatures through the critical summer months, below average rainfall through August as part of sustained drought conditions, and below average stream flow into Lake Sinclair and discharging from Sinclair Dam, all reflective of four-year drought conditions. Because power production was high, the withdrawal, use, and discharge of cooling water were also high.

Georgia Power operated the cooling tower system for the first time in 2002, effectively cooling the plant discharge an average of 6.3°C during the critical summer months July through September.

Despite reduced reservoir inflow due to ongoing drought conditions, Georgia Power managed Lake Sinclair water levels within the normal operating band of 338 to 340 ft plant datum.

The reservoir fish and benthic macroinvertebrate communities in Lake Sinclair experienced an environment typical of drought conditions with low river inflows and outflows, but lake levels were maintained within a narrow range. Interpretation of the biological community data in the context of this 316(a) demonstration study is not biased by atypical environmental conditions, but reflects high heat load under summer drought conditions.

Reference Area Verification

Water temperatures were monitored continuously from January 2002 through December 2002. Hourly water temperatures were recorded using temperature data loggers (StowAway® TidbiT[2]) deployed at eight locations in the Primary Study Area and 12 locations in reference areas, including one at the NPDES-specified ambient monitoring location designated "INT" (Figure 10-2). The data loggers were installed below the water surface in a manner that maintained an approximate recording depth of 1 m while accounting for normal fluctuations of water levels. Continuous temperature measurements were used to characterize seasonal and spatial trends in reservoir water temperatures, confirm the presence of reference conditions in areas of the reservoir outside of the Primary Study Area, and to assist with interpretation of biological data. Approximately 157,000 hourly measurements of water temperature were recorded throughout Lake Sinclair during 2002.

Primary Study Area mean daily average water temperatures (DATs) were consistently higher than those for other areas of Lake Sinclair when compared either individually or collectively (Table 10-1; Figure 10-3). The greatest differences were noted for the Little River arm followed by the upstream Oconee and downstream Oconee River arms, respectively. When the Primary Study Area mean monthly DAT was compared temporally to the other areas, temperatures were on average 2.9°C warmer with the greatest difference (5°C) noted during December and the least (1°C) occurring during July.

Evaluation of the water temperature dataset exclusively encompassing the June through September, 2002 critical conditions period indicated that no representative 95th percentile DAT-delta values exceeded the 1.5°C above-ambient reference condition criterion at the 12 stations located outside of the Primary Study Area (Table 10-2; Figure 10-4). Most notably, all discrete station-specific maximum temperature delta values during this period also were less than the reference condition criterion further supporting the presence of reference conditions in Lake Sinclair outside of the Primary Study Area during the critical conditions period of 2002.

Aquatic Community Data Collection

Multiple gear types were used to collect macroinvertebrate and fish samples from representative aquatic habitats shared by the Primary Study Area and reference area in Lake Sinclair. The results of detailed reservoir habitat mapping, assessment and scoring formed the basis for the selection of discrete sampling locations. Sampling effort was equal among study areas during standardized seasonal sampling for macroinvertebrate and fish assessments. During the August critical conditions Index Period sampling effort for fish was increased for reference areas.

[2] Onset Computer Corporation, Bourne, Massachusetts

Fish community sampling was conducted in five different habitat types identified in the Primary Study Area and reference areas of Lake Sinclair. Sampling gear types included electrofishing, gill nets, seines and hydroacoustic techniques. All fish community sampling was conducted at night.

Hydroacoustic surveys were conducted to assess abundance, distribution (vertical and longitudinal distributions), and movement of limnetic fishes reservoir-wide and in response to the thermal discharge. Reservoir-wide surveys were performed in February (winter) and August (summer) 2002 to obtain bathymetry information for Beaverdam Creek embayment and to map the horizontal and vertical distributions of fishes and provide estimates of fish density for use in the bioassessment.

The Lake Sinclair macroinvertebrate community was sampled on a seasonal basis using Hester-Dendy artificial substrate samplers, dip nets, and petite Ponar® grab samplers deployed exclusively in natural habitats in the Primary Study Area and reference area. Hester-Dendy samplers were deployed for a period of eight weeks prior to retrieval. Particle size analysis was conducted to identify comparable substrates to minimize Ponar® sampling bias.

Table 10-3 shows how the level of effort directed at sampling macroinvertebrate and fish communities differed with each gear and habitat type between standardized seasonal sampling and Index Period sampling.

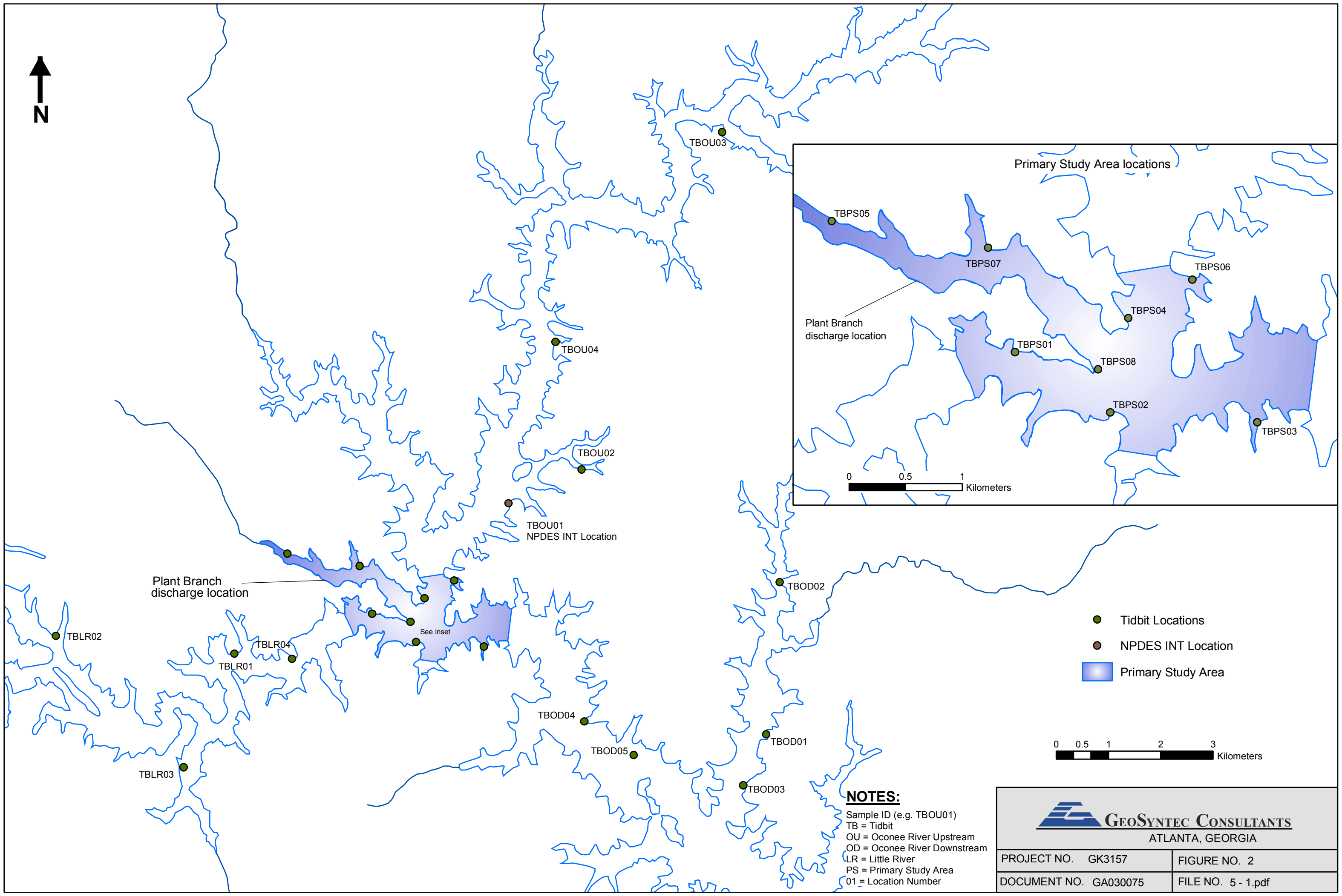


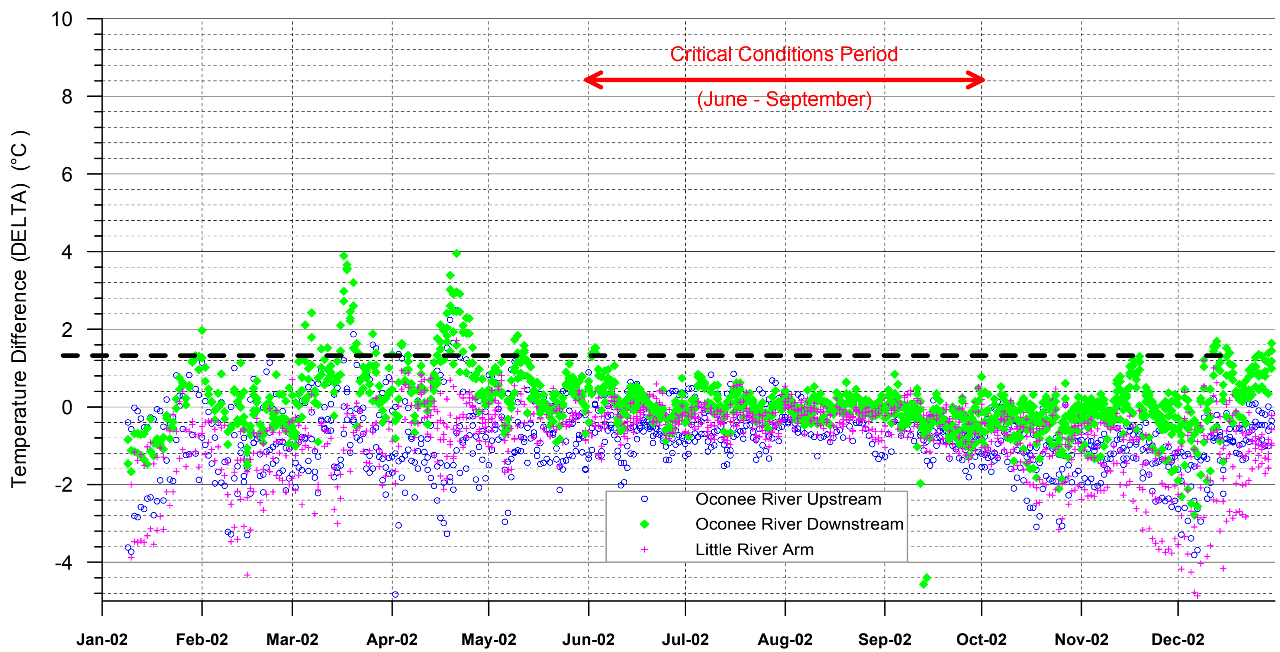
Figure 10-2
Deployment Sites for the Continuous Water Temperature Monitors ("Tidbits") in Lake Sinclair, 2002

Table 10-1
Summary Statistics for DAT (Daily Average Water Temperature) and DAT-Delta in Lake Sinclair, January – December 2002

Statistic	Reference Area											
	Upstream Oconee River Arm			Downstream Oconee River Arm					Little River Arm			
	TBOU02	TBOU03	TBOU04	TBOD01	TBOD02	TBOD03	TBOD04	TBOD05	TBLR01	TBLR02	TBLR03	TBLR04
Min Temp	9.6	8.4	9.3	10.6	9.9	10.9	10.6	10.6	8.6	7.9	8.6	8.9
Avg Temp	21.2	20.8	21.3	23.2	21.2	24.1	23.2	24.2	21.3	21.5	23.1	24.5
Max Temp	32.0	31.4	31.0	31.8	31.6	31.7	31.9	31.9	31.4	31.4	31.7	31.5
DAT-delta												
Min delta	-1.5	-4.8	-2.6	-1.4	-4.6	-1.5	-1.1	-0.8	-2.5	-4.9	-3.1	-1.7
Avg delta	-0.1	-1.4	-1.0	0.1	-0.1	0.1	0.3	0.1	-0.6	-1.3	-0.8	-0.2
Median delta	-0.1	-1.3	-0.9	-0.1	-0.2	0.0	0.2	0.1	-0.5	-0.9	-0.7	-0.2
95th percentile delta	1.1	-0.3	-0.4	1.4	**1.9**	1.2	1.5	1.1	0.2	0.1	0.2	0.5
Max delta	**2.5**	0.4	0.0	**3.5**	**3.9**	**2.6**	**3.7**	**1.6**	0.8	**1.7**	0.9	1.2
No. of days with delta > 1.5°C	5	0	0	14	21	5	14	1	0	1	0	0
Percent of time delta not > 1.5°C	98%	100%	100%	96%	93%	98%	95%	99%	100%	100%	100%	100%
No. of days with data	319	355	286	326	319	271	292	199	325	319	302	258

Statistic	Temp at TBOU01 (INT)	Primary Study Area							
		TBPS01	TBPS02	TBPS03	TBPS04	TBPS05	TBPS06	TBPS07	TBPS08
Min Temp	9.7	10.6	11.7	11.3	10.1	14.5	10.4	15.4	13.2
Avg Temp	22.2	23.5	24.7	24.3	23.4	27.1	24.8	24.6	26.4
Max Temp	31.7	32.7	33.7	33.1	32.4	33.7	33.3	33.5	34.0
DAT-delta									
Min delta	–	-0.9	0.9	0.3	-1.6	-0.5	0.2	**2.2**	1.3
Avg delta	–	1.3	**2.5**	1.5	0.6	**4.9**	**1.6**	**5.0**	**3.2**
Median delta	–	1.2	**2.4**	1.5	0.6	**5.5**	**1.6**	**4.9**	**3.1**
95th percentile delta	–	**2.7**	**4.1**	**2.6**	**1.6**	**8.1**	**2.6**	**7.3**	**4.9**
Max delta	–	**3.5**	**5.3**	**3.4**	**2.4**	**9.9**	**3.3**	**8.7**	**6.7**
No. of days with delta > 1.5°C	–	113	315	142	20	305	168	200	317
Percent of time delta not > 1.5°C	–	68%	11%	55%	94%	14%	48%	0%	2%
No. of days with data	–	355	355	319	333	355	322	200	322

Notes:
All temperatures reported in degrees Celsius (°C).
DAT-deltas greater than 1.5°C reported in bold text.



95th percentile daily average temperature deltas

Reference areas	Upstream Oconee River Arm			Downstream Oconee River Arm					Little River Arm			
	TBOU02	TBOU03	TBOU04	TBOD01	TBOD02	TBOD03	TBOD04	TBOD05	TBLR01	TBLR02	TBLR03	TBLR04
95th percentile delta	1.1	-0.3	-0.4	1.4	1.9	1.2	1.5	1.1	0.2	0.1	0.2	0.5

GeoSyntec Consultants
ATLANTA, GEORGIA

FIGURE NO.: 2
PROJECT NO.: GK3157
DOCUMENT NO.: GA030075
FILE: DAT deltas.xls

NOTES:
Symbols plotted above the dashed line represent DAT-deltas greater than 1.5°C

Figure 10-3
Daily Average Temperature "Deltas": Comparison of Reference Area Water Temperatures to Ambient Water Temperature ("INT" or TBOU01) in Lake Sinclair, January – December 2002

Table 10-2
Summary Statistics for DAT and DAT-Delta and in Lake Sinclair, June – September 2002 (Critical Conditions Period)

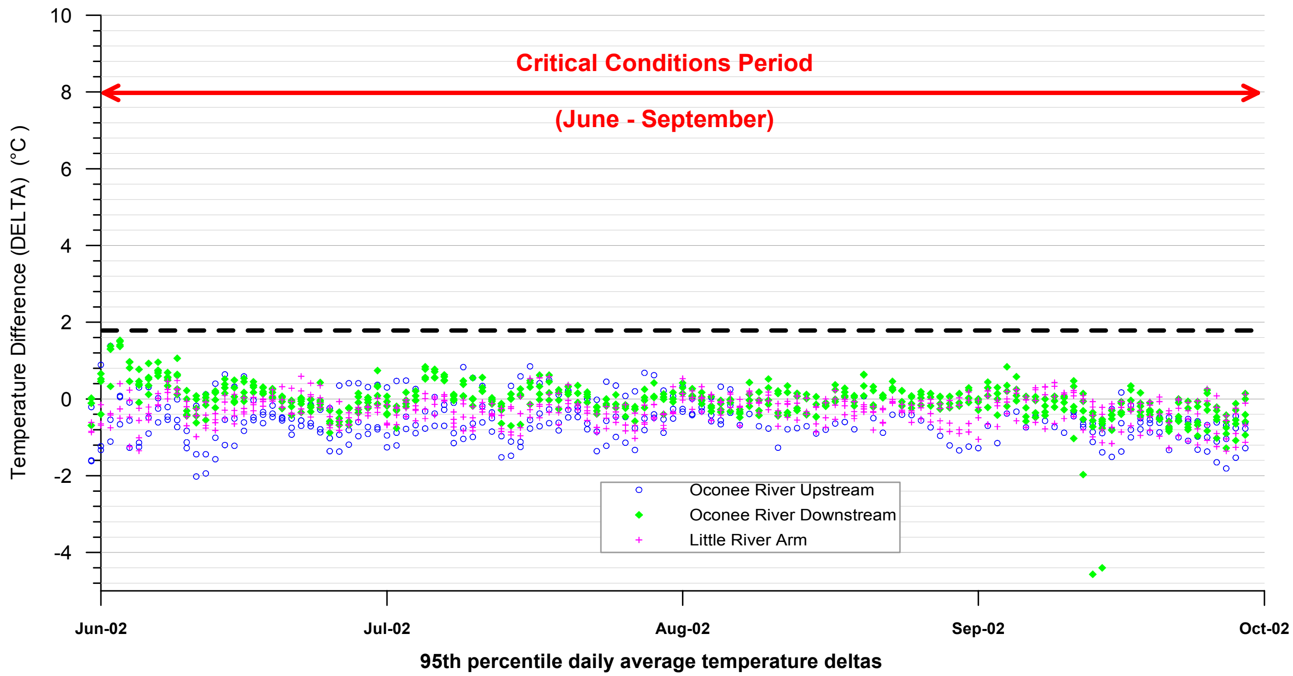
Statistic	Reference Area											
	Upstream Oconee River Arm			Downstream Oconee River Arm					Little River Arm			
	TBOU02	TBOU03	TBOU04	TBOD01	TBOD02	TBOD03	TBOD04	TBOD05	TBLR01	TBLR02	TBLR03	TBLR04
Min Temp	23.1	21.6	22.7	23.5	23.5	24.3	23.6	24.4	22.8	22.9	22.5	23.0
Avg Temp	28.5	28.0	27.7	28.8	28.4	29.2	29.5	28.6	29.2	28.4	28.5	28.8
Max Temp	32.0	31.4	31.0	31.8	31.6	31.7	31.9	31.9	31.4	31.4	31.7	31.5
DAT-delta												
Min delta	-1.0	-2.0	-1.6	-0.9	-4.6	-0.8	-0.8	-0.8	-1.0	-1.4	-1.3	-0.6
Avg delta	0.0	-0.8	-0.7	-0.1	-0.2	0.0	0.0	0.0	-0.3	-0.5	-0.3	-0.1
Median delta	0.0	-0.8	-0.7	-0.2	-0.1	0.0	0.1	0.0	-0.3	-0.5	-0.2	-0.1
95th percentile delta	0.7	0.0	-0.4	0.5	0.9	0.7	0.7	0.5	0.2	0.0	0.4	0.3
Max delta	1.4	0.3	-0.2	1.4	1.5	1.5	1.1	1.4	0.3	0.4	0.6	0.6
No. of days with delta > 1.5°C	0	0	0	0	0	0	0	0	0	0	0	0
Percent of time delta not > 1.5°C	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
No. of days with data	86	122	86	122	86	122	116	61	117	122	122	122

Statistic	Temp at TBOU01 (INT)	Primary Study Area							
		TBPS01	TBPS02	TBPS03	TBPS04	TBPS05	TBPS06	TBPS07	TBPS08
Min Temp	23.2	23.8	25.0	24.4	23.6	28.1	24.8	27.7	26.2
Avg Temp	28.9	29.9	30.9	29.9	29.4	31.6	30.1	31.6	31.6
Max Temp	31.7	32.7	33.7	33.1	32.4	33.7	33.3	33.5	34.0
DAT-delta									
Min delta	–	-0.1	0.9	0.3	-0.3	-0.5	0.3	**2.2**	1.4
Avg delta	–	1.0	**1.9**	1.0	0.5	**2.0**	1.2	**3.7**	**2.6**
Median delta	–	0.9	**1.9**	0.9	0.5	**1.9**	1.2	**3.8**	**2.5**
95th percentile delta	–	**1.7**	**2.7**	**1.7**	1.1	**4.2**	**1.8**	**5.0**	**3.5**
Max delta	–	**2.3**	**3.6**	**1.9**	**1.6**	**4.9**	**2.5**	**5.1**	**4.1**
No. of days with delta > 1.5°C	–	**15**	**94**	**12**	1	**72**	**22**	**25**	**118**
Percent of time delta not > 1.5°C	–	88%	23%	90%	99%	41%	82%	0%	3%
No. of days with data	–	122	122	122	122	122	122	25	122

Notes:

All temperatures reported in degrees Celsius (°C).

DAT-deltas greater than 1.5°C reported in bold text.



Reference areas	Upstream Oconee River Arm			Downstream Oconee River Arm					Little River Arm			
	TBOU02	TBOU03	TBOU04	TBOD01	TBOD02	TBOD03	TBOD04	TBOD05	TBLR01	TBLR02	TBLR03	TBLR04
95th percentile delta	0.7	0.0	-0.4	0.5	0.9	0.7	0.7	0.5	0.2	0.0	0.4	0.3

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FIGURE NO.: 4
PROJECT NO.: GK3157
DOCUMENT NO.: GA030075
FILE: DAT deltas.xls

NOTES:
Symbols plotted above the dashed line represent DAT-deltas greater than 1.5°C

Figure 10-4
Daily Average Temperature "Deltas": Comparison of Reference Area Water Temperatures to Ambient Water Temperature ("INT" or TBOU01) in Lake Sinclair, June – September 2002

Table 10-3
Fish and Macroinvertebrate Community Sampling Effort in Lake Sinclair, 2002

Survey Task	Study Area	Effort	January (1)	February	March	April	May (1)	June	July	August (2)	September	October	November (1)	December
Fish Community & Principal species - electrofishing	Primary Study Area	#mins/ #runs	10/12				10/12	5/12	5/12	10/12	5/12		10/12	
	Reference Area	#mins/ #runs	10/12				10/12	5/12	5/12	10/24	5/12		10/12	
Fish Community & Principal species - gill netting	Primary Study Area	net locations	6				6			6			6	
	Reference Area	net locations	6				6			12			6	
Juvenile fish seining	Primary Study Area	net hauls	6		6	6	6	6	6	6	6	6		
	Reference Area	net hauls	6		6	6	6	6	6	12	6	6		
Hydroacoustics	Lake Sinclair	Lakewide vs Primary Study Area		Lake wide x 1				Primary Study Area x 2	Primary Study Area x 2	Primary Study Area x 2 plus Lakewide x 1	Lake wide x 1			Primary Study Area x 1
Macroinvertebrate Community	Primary Study Area	Hester-Dendy, dip net, and petite ponar		8 locations x 3 gear types			8 locations x 3 gear types			8 locations x 3 gear types +15 min hand pick			8 locations x 3 gear types	
	Reference Area	Hester-Dendy, dip net, and petite ponar		16 locations x 3 gear types			16 locations x 3 gear types			16 locations x 3 gear types+15 min hand pick			16 locations x 3 gear types	

Notes:

(1) Seasonal sampling event.

(2) Index period and seasonal sampling event.

Shaded boxes indicate survey task not performed.

Application of The Multimetric Bioassessment

The multimetric bioassessment was applied to the macroinvertebrate and fish community data collected from the Primary Study Area and the reference area of Lake Sinclair during 2002. Seasonal biological data sets were used concurrently with the habitat assessment data to evaluate the comparability between study areas and to determine if a balanced, indigenous community was present in the Primary Study Area. Emphasis was placed on the summer Index Period (August) as being most reflective of thermally critical summer conditions in Lake Sinclair. However, bioassessment techniques were also applied during other seasons to provide a holistic seasonal representation of the status of macroinvertebrate and fish communities in the Primary Study Area.

The condition of the Primary Study Area biological community was scored against the reference condition in Lake Sinclair as a measure (index) of the expected, best attainable condition in the absence of thermal influences related to Plant Branch. Adaptation of traditional reference condition characterization and metric scoring was required because multimetric bioassessment protocols have not been developed for Georgia reservoirs. Development of the bioassessment scoring matrix for this study relied on a combination of USEPA's Reservoir Bioassessment Guidance, Georgia's macroinvertebrate and fish bioassessment protocols for wadeable streams, and as provided by those protocols, professional judgment [5, 4, 7]. The process of developing site-specific bioassessment indices for Lake Sinclair consisted of several study elements including:

- conducting a detailed habitat characterization of the study area and associated quantitative habitat assessment,
- verification of the non-thermal reference condition within Lake Sinclair outside of the Primary Study Area,
- evaluation and selection of individual biological metrics for macroinvertebrate and fish communities, and
- the scoring of each metric (measure) and summation to yield biotic index scores for macroinvertebrate and fish bioassessment indices.

Biological metrics were calculated from the representative, multi-gear composite samples of the macroinvertebrate and fish communities collected from the Primary Study Area and reference area during the Index Period and other seasons. The status of the macroinvertebrate and fish communities residing in the Primary Study Area relative to the reference area was then evaluated using the multimetric indices. The basis of the index scoring framework stemmed from characterization of the reference (non-thermal) condition which represents the best attainable biological condition for macroinvertebrates and fish in Lake Sinclair in the absence of thermal influences related to Plant Branch. Resultant scores were calculated for each metric corresponding to its deviation from the expected reference value. Individual metric scores were then summed to yield bioassessment "index" scores for the macroinvertebrate and fish communities.

Establishment of a "decision" reference mark within each macroinvertebrate and fish biotic index relied on the "bisection" index scoring methodology described in USEPA's reservoir guidance manual [5]. The USEPA guidance indicates that the bisection index scoring method with its lower percentile (i.e., 25^{th} percentile) cut-off is preferred when reference sites are representative of relatively unimpaired conditions. In the context of this study, the term "unimpaired" equates to non-thermal influence by the Plant Branch discharge.

The following sections present the methodology used for developing and applying bioassessment indices for Lake Sinclair as a tool to determine in accordance with CWA Section 316(a) whether a balanced, indigenous aquatic community was present in the Primary Study Area during the 2002 Index Period and other seasons.

Defining Reference Conditions

Characterization of the reference biological condition within Lake Sinclair was necessary in the application of a multimetric bioassessment to establish the range of expected sample distributions (score ranges) for test site (Primary Study Area) comparison. The reference condition for a site (or region) can be defined as the highest "biological integrity" attainable for a site or the level of biological integrity a regulatory agency expects as highest attainable under prevailing conditions for a site or region. Biological integrity is defined as the ability of an ecological system to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of *natural habitat* of the region [8]. Based on Karr's definition, a condition of biological integrity does not exist in reservoirs because they are man-made, multi-use systems, not natural systems. However, reservoirs can support aquatic communities that are adapted to the habitat conditions and range of management actions afforded them. This view of adaptive reservoir communities is consistent with WRD's recognition that "reference" conditions do not represent pristine conditions, as it would be "unrealistic and inappropriate" [7]. Rather, WRD establishes reference sites that represent baseline conditions on least-altered systems that exist given the land-use patterns within an eco-region.

The "site-based" method was used for establishing the reference condition in Lake Sinclair rather than using the "eco-region reference (non-impacted) condition" method [5]. It is important to note that the objective of this study is to determine how the biological communities in the Primary Study Area compare to those in representative non-thermal areas of Lake Sinclair, and *not* how they may compare to an eco-regional reference. Both the WRD and EPD bioassessment protocols recognize the site-based or "control" area approach for assessment of macroinvertebrate and fish communities [4, 7]. Reference areas in Lake Sinclair for this study represented non-thermal "control" areas outside the Primary Study Area as confirmed by the continuous water temperature measurements. The structure and diversity exhibited by biological communities in the reference area established the "best-attainable" condition for comparison to the Primary Study Area. That comparison formed the basis of the multimetric index-based assessment of thermal impacts attributable to the operation of Plant Branch.

Upon establishing the spatial extent of the reference area in Lake Sinclair, site-specific reference conditions were developed for application of the bioassessment based on intensive collection of macroinvertebrate and fish community data from habitats representative of non-thermal areas of the reservoir yet comparable to habitats in the Primary Study Area. The underlying design of this site-specific approach within Lake Sinclair is analogous to development of regional reference conditions in contemporary, State-sponsored, bioassessment programs as provided in USEPA guidance [5]. Sampling in comparable habitats is important as it reduces habitat-related biases that can affect index scoring and interpretation of the bioassessment results. Physical habitat is known to affect the structure and function of aquatic communities [9].

Index Scoring

Application of bioassessment indices relies on the comparison of metric results from a test site(s) to a reference (expected) condition and making a judgment whether that value falls within the expected range (Integrity Classification or Ecological Condition category). In developing the multimetric macroinvertebrate and fish indices for Lake Sinclair, biological data obtained from the reference areas were summarized, graphed, and empirically analyzed to illustrate the range of distribution for community-level characteristics. In traditional application of many multimetric indices, and in this demonstration, metric score categories were expressed with ordinal values of 5, 3, or 1 points where:

- the test site (i.e., the Primary Study Area) ecological condition approximates the reference condition (score of 5);
- the ecological condition of the Primary Study Area deviates somewhat from the reference condition (score of 3); or
- the ecological condition of the Primary Study Area deviates strongly from the reference condition (score of 1).

For certain numerical metrics, modification was introduced into this scoring convention as a means to accommodate narrow ranges of reference condition values. More detail is provided on this approach later in the text.

Individual macroinvertebrate and fish community metrics for the Primary Study Area datasets were scored against the reference condition then summed to yield a total index score for each biotic index (i.e., macroinvertebrates and fish). The maximum score possible in either the fish or macroinvertebrate indices is dependent on the number of metrics used and how the score categories are valuated. For example, Karr's original IBI [10, 8] consisted of 12 metrics each capable of receiving a maximum score of five producing a potential maximum index score of 60. In this example, a test site receiving a score of 60 would be indicative of a site equivalent to the best reference condition.

Bioassessment indices developed for Lake Sinclair were patterned after approaches provided in Georgia's current bioassessment protocols [4, 7]. The maximum attainable score for the Lake Sinclair fish community index is 60 points (i.e., 12 individual metrics), whereas the maximum possible macroinvertebrate community index score is 35 point (i.e., 7 individual metrics). Progressively increasing scores for the Primary Study Area denote higher degrees of comparability with the reference condition.

As previously presented, index scoring methodology used in this bioassessment followed the bisection methodology recommended by USEPA when reference site(s) represent unimpaired conditions, which under the current study are represented by non-thermal conditions [5]. The lower quartile (25th percentile) of the range of reference values is most frequently taken as the cutoff in determining degrees of impairment at a test site [5]. Metric values from a given test site that exceed the 25th percentile value of the reference condition are considered to be comparable (unimpaired) and subsequently received a score of 5 points (Figure 10-5). Those metric values occurring between 0 and the 25th percentile are bisected with values in the top half receiving a score of 3 points and values in the lower half receiving a score of 1 point.

In certain instances, reverse scoring metrics are applied in bioassessment indices. Three metrics each used for assessing the macroinvertebrate and fish communities in this demonstration were reverse scoring metrics. A reverse scoring metric is necessary when a negative attribute such as the occurrence and abundance of undesirable species in a community needs to be characterized.

The reverse scoring metric was developed by establishing the 75th percentile of the reference range then bisecting the area above that value (Figure 10-6). The range of metric values occurring below the 75th percentile (the more desirable condition) corresponded to a score of 5 points. The upper half of the bisected range above the 75th percentile, representing the least desirable condition, corresponded to a metric score of 1 point and the lower half of the bisected range (intermediate range) corresponded to a score of 3 points.

Following Georgia's bioassessment worksheets (for fish community assessment), the overall index scoring range (i.e., 0 to 60 points), derived by summation of individual metric scores, was divided and classified into several score ranges labeled as "Integrity Classes" [7]. EPD's bioassessment protocol for macroinvertebrate communities in the Piedmont eco-region is scaled differently in that percentage breakpoints, based on comparison to the maximum possible score, are used to define "Ecological Condition" categories on a scale ranging up to 35 points. For example, an Ecological Condition designation of "Very Good" in the EPD protocol equates to a test site with an index score equal to or greater than 87 percent of the reference condition [4].

Application of Score Results

From a regulatory perspective, resultant index scores correspond to conditions of "impaired" or "unimpaired" waters. The EPD and WRD bioassessment protocols for small wadeable streams support five integrity classes ranging from "Excellent/Very Good" to "Very Poor" that were likely based in part on best visual fit of the data (including data from impaired and un-impaired sites) using professional judgment [4, 7]. A determination of "impairment" for a test site corresponds with a particular break point within the integrity class ranks.

The bioassessment of Lake Sinclair adapted, as applicable, the integrity class categorization scheme and use-support/impairment criteria developed by WRD and EPD. Test site(s) receiving a score in categories below (less than) the "Fair" integrity class are considered to be "*not supporting*" designated use, which is interpreted as "*impaired*" to some degree from a regulatory perspective. The MOA-approved Study Plan provides that biotic communities of the Primary Study Area are considered "supporting" (i.e., a balanced, indigenous aquatic community is supported) if resultant index scores reflect biological integrity of at least the "Fair" category. If both macroinvertebrate and fish samples of the Primary Study Area receive an index score(s) of "Fair" or better following completion of the bioassessment, then Performance Criteria have been achieved (Section 3.3).

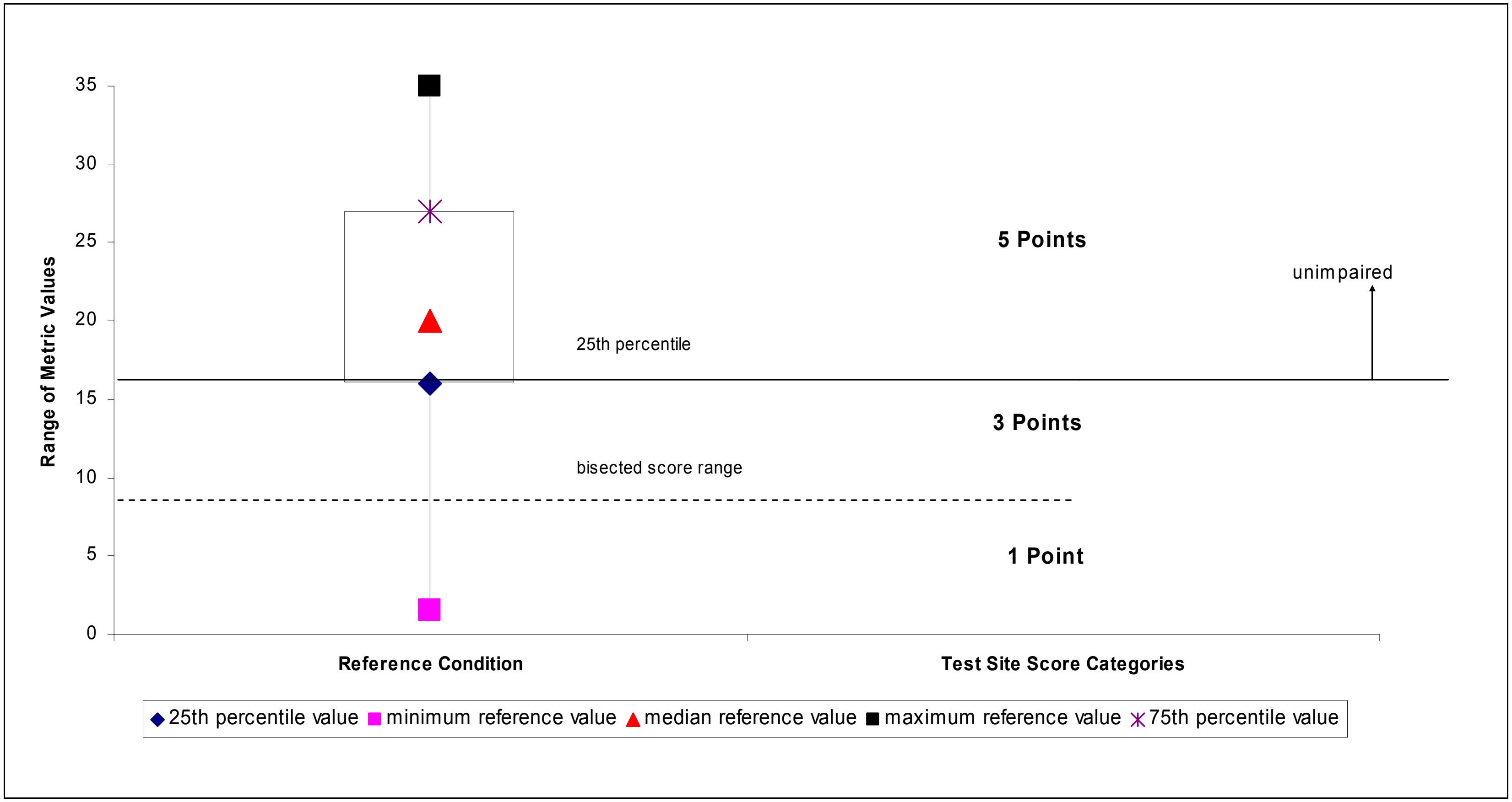


Figure 10-5
Schematic of 25th Percentile Metric Scoring Method

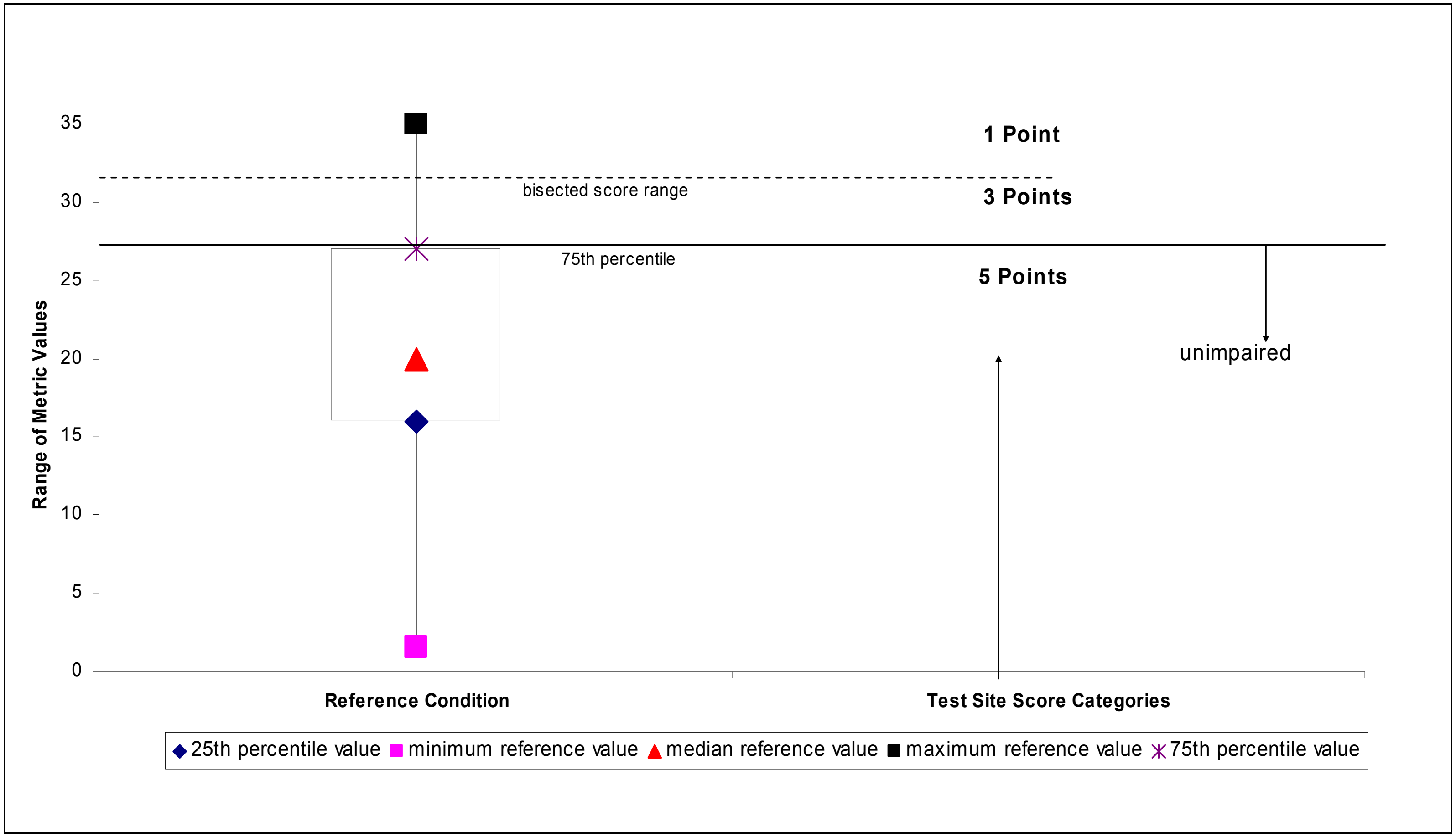


Figure 10-6
Schematic of 75th Percentile "Reverse" Metric Scoring Method

Biotic Community Metrics

Comprehensive multimetric indices embrace several attributes of the sampled community assemblage [11]. Useful metrics must be able to discriminate between impaired ecological conditions and conditions that meet designated uses. Macroinvertebrate and fish indices consist of measures representing several broad functional biological categories [12, 5]. The Lake Sinclair bioassessment directly utilized existing metrics or adapted metrics and scoring criteria recommended or currently used by USEPA in their Reservoir Bioassessment Guidance, and EPD and WRD in their stream bioassessment protocols for macroinvertebrate and fish community assessments [5, 4, 7].

Candidate metrics and metrics for consideration as alternates for each biotic category were presented in the MOA-approved Study Plan. Consistent with agency guidance, metrics used in this study for either macroinvertebrate or fish community indices were selected to represent a cross-section of the following functional ecological categories, when data supported their use.

- Category I–Species richness and composition
- Category II–Feeding measures and trophic dynamics
- Category III–Tolerance/Intolerance measures
- Category IV–Reproduction and Abundance (fish community only)
- Category V–Health (fish community only)

Descriptions of individual metrics used in this demonstration are provided later in the text.

Macroinvertebrate Bioassessment

Metrics evaluating stream benthic macroinvertebrate communities have been extensively developed at the federal and state levels [5]. The Georgia EPD has developed a macroinvertebrate index in their standard operating protocols for biological assessments of wadeable streams utilizing freshwater macroinvertebrates [4]. However, little work has been conducted on assessment of reservoir health in the southeast using macroinvertebrate indices, and metrics have not been developed for Georgia reservoirs. However, applicable bioassessment approaches for lakes and reservoirs are being developed regionally (e.g., TVA reservoirs and Florida lakes). Minor modifications to metrics tested and used by EPD and other natural resource agencies for stream applications provided the basis for developing appropriate metrics for this study.

Macroinvertebrate Community Metrics

The following macroinvertebrate metrics were selected for the bioassessment to determine if a balanced, indigenous macroinvertebrate community exists in the Primary Study Area of Lake Sinclair.

Category I–Species Richness and Composition

Metric 1–Number of taxa—this metric is widely applicable for multimetric bioassessments because it responds to increased environmental stress in the form of reduced species richness (i.e., fewer taxa). It provides a measure of overall variety of the macroinvertebrate assemblage. Increased taxa diversity correlates with increased health of the assemblage and suggests that habitat and food sources are adequate to support the survival and propagation of a diverse macroinvertebrate assemblage [5]. The average number of taxa reported from the Primary Study Area samples were compared to the 25th percentile value of the number of taxa in samples from the reference area samples.

Metric 2–Number of ETO taxa—This metric is a modification of the EPT taxa metric that is commonly applied to stream assessments [4]. The modification involved substituting Odonata (damsel and dragonflies) taxa for Plecoptera (stoneflies) taxa which are primarily found in streams. The *number of ETO taxa* represents the number of species present in the Orders of Ephemeroptera (mayflies), Trichoptera (caddisflies), and Odonata. Twenty-eight ETO taxa were reported in the 2002 samples reservoir-wide including nine mayfly taxa, 10 caddisfly taxa, and nine Odonate taxa. ETO taxa are representative of long-lived aquatic insects typically found in higher quality habitats. The ETO taxa metric has been demonstrated to respond to environmental stresses in Florida lakes with decreased number of taxa [13].

Metric 3–Number of Dipteran taxa—This metric indicates the number of taxa in the order Diptera (aquatic true flies), which represents the most diverse group of macroinvertebrates collected from Lake Sinclair in terms of taxa richness and ecological characteristics. Reservoir-wide, dipterans accounted for 69 of the 168 taxa reported in 2002 and represented six functional feeding groups. The *number of dipteran taxa* metric is analogous to the *number of Chironomidae taxa* metric used by the EPD [4]. In this case the two metrics are essentially the same as all but six dipteran taxa reported in 2002 were from the family Chironomidae. The number of dipteran taxa would be expected to decrease if conditions in the Primary Study Area are perturbed compared to the reference area.

Category II–Feeding Measures and Trophic Dynamics

Metric 4–Percent Chironomidae—this metric performs as a reverse scoring metric. It accounts for the relative abundance of Chironomidae (midge) taxa that are generally expected to increase in abundance as intolerant organisms decline in response to decreased levels of dissolved oxygen. This metric potentially targets elevated water temperature effects because of the inverse relationship between water temperature and dissolved oxygen concentrations.

Metric 5–Percent Filterers—this metric accounts for the relative abundance of taxa that belong to the filtering collector functional feeding group that is expected to decrease in stressed environments [5].

Category III–Tolerance/Intolerance Measures

Metric 6–North Carolina Biotic Index (NCBI)—this metric also performs as a reverse scoring metric. It is a modification of the Hilsenhoff Biotic Index that was developed as a means for detecting organic pollution in benthic macroinvertebrate communities. New Jersey has tested and accepted this metric for use in the assessment of lakes and reservoirs [5]. The index is intended for examination of the general level of pollution regardless of the source in various community types. Tolerance values for individuals range from 0 (very intolerant) to 10 (very tolerant). The North Carolina Biotic Index is included in Georgia's stream assessment protocol [4].

Metric 7–Percent dominant taxa—The percent contribution of the numerically dominant taxon to the total number of organisms is an inverse metric. This is an indicator of community balance at the lowest taxonomic level (usually genus or species level). Metric 7 is considered a measure of tolerance/intolerance on the basis that communities dominated by a few taxa are reflective of degraded conditions. Healthy communities are expected to exhibit relatively balanced proportions and trophic structure.

Macroinvertebrate Bioassessment Results

The seven selected metrics were used to describe and compare community structure and trophic function in the Primary Study Area relative to the reference condition for the critical conditions Index Period of August 2002 and the other three seasonal sampling events. Individual metrics were scored against the reference condition then summed to yield a total bioassessment index score. Index score categories describe whether the macroinvertebrate community in the Primary Study Area exhibits a balanced, indigenous aquatic community.

Index Period Score

The macroinvertebrate community biotic index score for the Primary Study Area during the August 2002 Index Period was 31 points. Based on the EPD integrity class ranges, an index score of 31 points places the biological integrity of the Primary Study Area in the "Very Good" integrity classification (31 to 35 points; Table 10-4). Five of the seven community metrics received maximum five-point scores. Metric 1–*total number of taxa* received a score of 3 because the average number of taxa (30) in the Primary Study Area samples was slightly lower than the 25^{th} percentile of the reference conditions. Metric 7–*percent dominant taxon*, a reverse scoring metric received a score of 3 because of the relatively high percent contribution of the dominant taxa in the Primary Study Area compared to the 25^{th} percentile for the reference condition. Each metric result was represented in schematic form presenting key information used to establish the reference condition scoring ranges. An example schematic is provided in Figure 10-7.

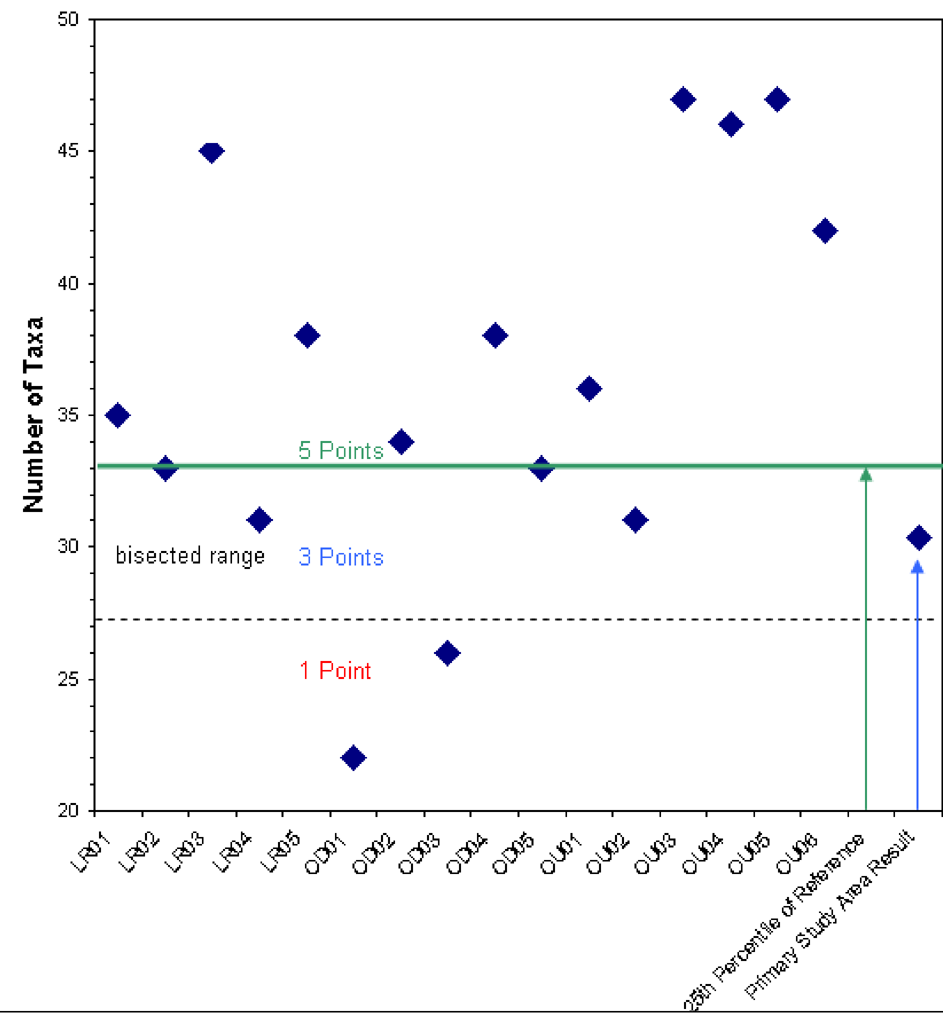
Table 10-4
Macroinvertebrate Community Bioassessment Index Score Sheet for the Primary Study Area for the August 2002 Index Period

Macroinvertebrate Index Period Worksheet				**Score Category**			
	Metric description	**Unit of measure**	**PSA result**	**5**	**3**	**1**	**Metric score**
1	Total number of taxa	no. of taxa	30	> 33	27-33	< 27	3
2	Number of ETO taxa	no. of taxa	6	> 5	4-5	< 4	5
3	Number of dipteran taxa	no. of taxa	16	> 15	13-15	< 13	5
4	Percent chironomidae	percent	67	< 76	76-79	> 79	5
5	Percent filterers	percent	39	> 32	29-32	< 29	5
6	North Carolina Biotic Index	NCBI value	7.72	< 8.20	8.30-8.20	> 8.30	5
7	Percent single most dominant taxon	percent	30	< 27	27-37	> 37	3
Macroinvertebrate Bioassessment Index Score (August 2002 Index Period)							**31**
EPD Integrity Class						**Very Good**	
EPD Integrity Class descriptions [5]							
Very Good - comparable to best situation expected.							35-31
Good - balanced community with sensitive species present.							30-25
Fair - expected species absent or in low abundance; few present species present.							24-19
Poor - low species richness, with tolerant species predominant, sensitive species absent.							18-14
Very Poor - expected species absent, having only tolerant organisms present.							<14

Note:
ETO denotes Ephemeroptera, Trichoptera, and Odonata taxa.
PSA denotes Primary Study Area.

Index Period Result

Result	No. taxa
Reference Area Samples	
LR01	35
LR02	33
LR03	45
LR04	31
LR05	38
OD01	22
OD02	34
OD03	26
OD04	38
OD05	33
OU01	36
OU02	31
OU03	47
OU04	46
OU05	47
OU06	42
25th Percentile of Reference	33
Primary Study Area Result	30



Score Criteria* based on Bisect Scoring Method	
5 Points for values >	33
3 Points for values =	27-33
1 Point for values <	27

Primary Study Area Score, Metric 1	3 Points

Figure 10-7
Schematic for Macroinvertebrate Metric 1—Total Number of Taxa

Seasonal Index Scores

Bioassessment methods also provided a mechanism for evaluating biological community integrity within seasons other than the Index Period. Conducting bioassessments in other seasons provided the opportunity to:

- lend "weight of evidence" to the overall interpretation of Plant Branch effects on the macroinvertebrate community,
- look for potential information redundancies or gaps in the selected metrics,
- examine temporal variation in community comparisons as a means to validate and bracket the August result, and
- examine sensitivity of individual metrics and the index overall to reflect trends observed in the empirical analyses.

Bioassessment index scores for the Primary Study Area were calculated for winter, spring, and fall seasons based on reference conditions developed for each season using the same method as was applied for the Index Period. As indicated previously, sampling effort (i.e., number of sampling locations) in the reference areas was the same throughout the study so that the data necessary were available to determine seasonal biotic integrity in the Primary Study Area.

The Primary Study Area macroinvertebrate community index scored 31, 31, and 33 in the winter, spring and fall seasons, respectively (Table 10-5). Based on the EPD integrity classifications, these scores correlate to ratings of "Very Good" (Figure 10-8).

Index Relationship to Empirical Data

The macroinvertebrate community metric scores in the spring, summer, and fall samples reflected higher scores for Metric 1–*number of total taxa* and Metric 7–*percent dominant taxa* (Table 10-5). Four of the seven metrics scored a 5 throughout the study. Scores less than 5 occurred for Metric 1 (winter and summer), Metric 5–*percent filterers* (spring and fall), and Metric 7–*percent dominant taxon* (spring and summer). These results are consistent with the empirical data that indicated consistent similarities among study areas with spatial and seasonal variability typically related to one or two sampling locations.

Fish Community Bioassessment

Data requirements for the bioassessment component of the study were satisfied through analysis of seasonal datasets. The fish community biotic integrity was scored for each seasonal sampling event including the critical conditions Index Period of August 2002. Thirteen fish community metrics were used to describe and compare community structure and trophic function in the Primary Study Area relative to the reference condition. Individual metrics were scored against the reference condition then summed to yield a total bioassessment index score. Index score categories describe whether the fish community in the Primary Study Area exhibits a balanced, indigenous aquatic community condition by how well it compared to the reference condition.

Table 10-5
Summary of Seasonal Macroinvertebrate Community Bioassessment Index Scores for the Primary Study Area in Lake Sinclair, 2002

Macroinvertebrate Community Metrics		Winter	Spring	Summer*	Fall
1	Number of taxa	3	5	3	5
2	Number of ETO taxa	5	5	5	5
3	Number of dipteran taxa	5	5	5	5
4	Percent chironomidae	5	5	5	5
5	Percent filterers	5	1	5	3
6	North Carolina Biotic Index	5	5	5	5
7	Percent single most dominant taxon	3	5	3	5
Seasonal Macroinvertebrate Index Scores		**31**	**31**	**31**	**33**
EPD Integrity Class		**Very Good**	**Very Good**	**Very Good**	**Very Good**

EPD Integrity Classes	Scoring Range	Percent comparability
Very Good	35-31	100-87%
Good	30-25	86-74%
Fair	24-19	73-49%
Poor	18-14	48-25
Very Poor	<14	25%

Notes:

* August Index Period.

ETO denotes Ephemeroptera, Trichoptera, and Odonata taxa.

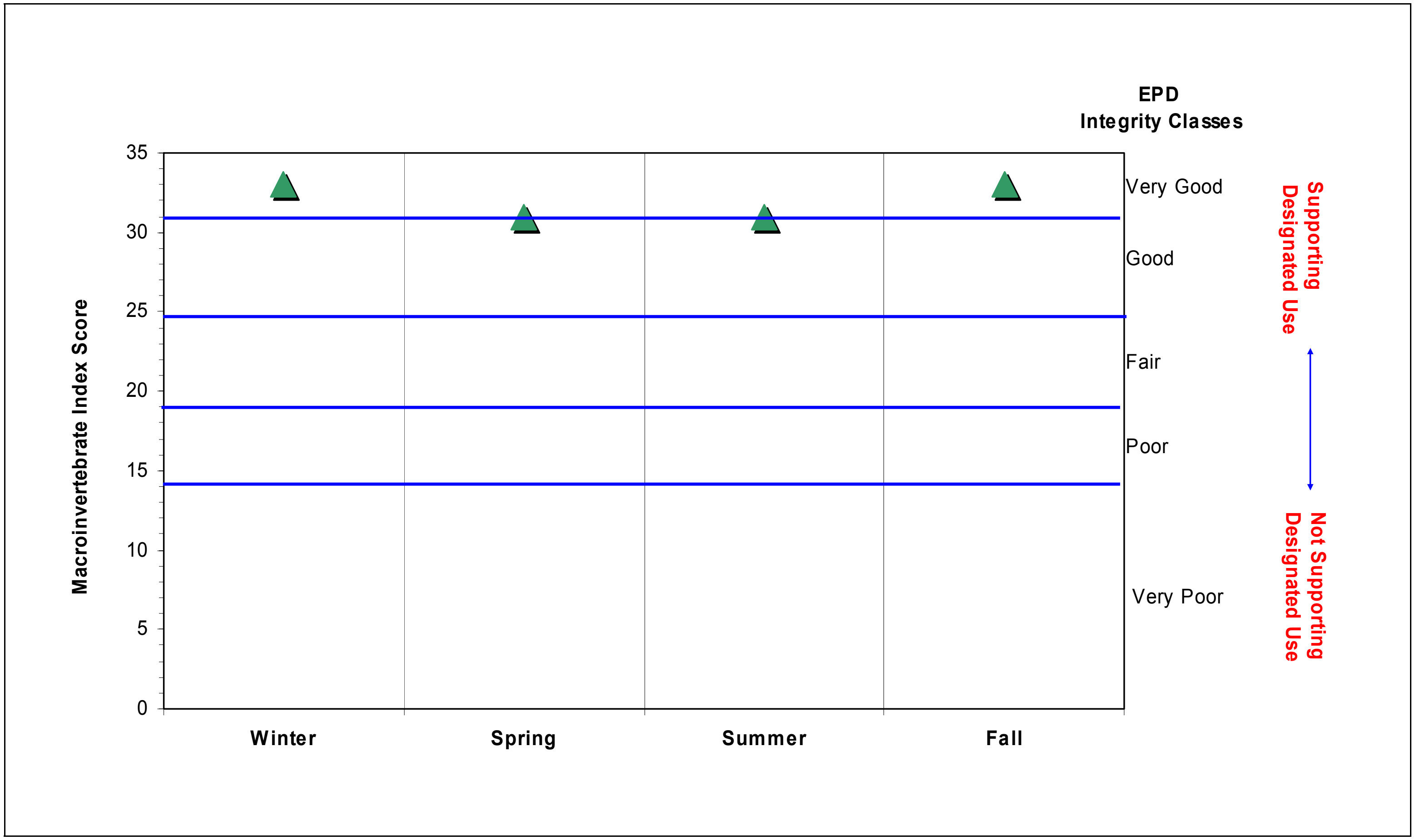


Figure 10-8
Seasonal Macroinvertebrate Community Bioassessment Index Scores for the Primary Study Area in Lake Sinclair, 2002

Fish Community Assessment–Metric Selection

Historical and management aspects of Lake Sinclair's fishery were considered in development of metrics for this assessment. The Study Plan presented 11 candidate metrics plus several other proposed alternative metrics that were evaluated before selecting 13 fish community metrics for use in this demonstration. Three of the 11 metrics first proposed in the Study Plan were changed or modified based on review comments. In order to mirror the scoring capacity of Georgia's biotic index, two more metrics were considered and ultimately selected for use following evaluation of applicability in this demonstration. The maximum score possible for Georgia's current stream fish community bioassessment is 60 points (12 metrics; 5 points maximum each plus a 13th metric applied as a negative offset [-4 points] depending on percent health anomalies observed).

Ultimately, metric selection was based on data evaluation, metric response or redundancy, and professional judgment in accordance with the established protocols. The following text presents a description of ecological function categories and community metrics selected for the Lake Sinclair bioassessment.

Category I–Species Richness and Composition

Metric 1–Total Number of Species (>25 mm total length)—this metric replaced the originally proposed candidate metric: *number of native species*. Screening the fish community datasets to exclude fish <25 mm total length effectively removes the population component attributed to young-of-year (YOY) fishes recently recruited into the fishery. If included in certain proportional metrics, fish measuring <25 mm could introduce bias and mask the performance of certain metrics due to normally high abundance of YOY during the Index Period. The selected metric is a measure of the number of all fish species represented in the fishery including hybrids (e.g., hybrid striped bass, *M. saxatilis. x M. chrysops;* hybrid sunfish, *Lepomis sp. x Lepomis sp.*), non-native species (e.g., green sunfish, *L. cyanellus;* yellow perch, *Perca flavescens*) and exotic species (e.g., common carp, *Cyprinus carpio*). This metric represents the diversity of the fish assemblage and is responsive to environmental stress, including thermal stress, in that the number of species is expected to decline with increased environmental degradation.

The *number of native species* metric is used in WRD's stream bioassessment protocol. Non-indigenous species have been found to increase in systems that are disturbed, particularly in small streams. However, Lake Sinclair like other large southeastern reservoirs supports a number of non-native, exotic, and potentially nuisance species, in some cases supported by routine stocking programs of species such as hybrid striped bass. Lake Sinclair's ability to support a variety of fishes other than indigenous species would be expected where, prior to impoundment, many of the indigenous species were adapted to a free flowing system that is no longer supported. As such, support of non-indigenous species in the reservoir is not necessarily a reflection of anthropogenic degradation from surrounding environments. It is generally known that trends in fish community trophic diversity gravitate towards a more omnivore/generalist fish community as reservoirs age. So, in evaluating the number of species metric, comparability between study areas becomes more an investigation of balance in species diversity in the fishery than a measurement of indigenous species consistent with watershed expectations. Therefore, the total number of species >25 mm is better suited to characterizing reservoir species richness expectations in Lake Sinclair.

Metric 2–Percent Sportfish Species—following comments received from WRD during Study Plan review, this metric replaced candidate metric *number of all sunfish species*. The selected metric is a measure of the percent contribution of sport fish species to the fish community. These species are regarded as important to anglers and/or are specifically managed by the WRD. Lake Sinclair's sport fishes include largemouth bass, *Micropterus salmoides;* bluegill, *L. macrochirus;* white bass, *M. chrysops;* hybrid striped bass*;* black crappie, *Pomoxis nigromaculatus;* redbreast sunfish, *L. auritus;* warmouth, *L. gulosus;* redear sunfish, *L. microlophus* and yellow perch. The proportion, rather than number, of sport-fish species was selected due to the potential overlap and informational redundancy with Metric 4–*number of piscivorous species*, which largely represents the same list of species.

Metric 3–Number of Temperature Sensitive Species—this metric replaced a candidate Metric 3–*number of sucker species*. Comments received during Study Plan review indicated a preference for modifying the candidate metric to include all temperature sensitive species. This metric represents the number of species residing in Lake Sinclair that are believed to be sensitive to higher water temperatures. Temperature sensitive species as designated by WRD included black crappie, white bass, spotted sucker, *Minytrema melanops;* silver redhorse, *Moxostoma anisurum;* and hybrid striped bass.

Metric 13–Shannon-Wiener (H') Index of Diversity—This metric provides a comparison of fish species diversity between study areas and was used as a substitute for the Evenness Index normally applied in WRD's fish bioassessment protocol. Shannon-Wiener was calculated using the natural log conversion cited in WRD's fish bioassessment protocol where:

$$H' = -\Sigma (n_i/N) \ln(n_i/N)$$

n_i = numbers of individual species

N = total number of individuals

ln = natural logarithm.

Expectations in WRD's Evenness Index are calibrated to reflect the range of conditions known for small wadeable streams of the Georgia Piedmont. As such, the current Evenness Index used by WRD's bioassessment is not applicable to Lake Sinclair. However, H′ is an underlying component of the WRD Evenness Index. As a valid measure of community diversity, H′ was adopted for this application. Expectations for responsiveness using H′ have not been calibrated to reservoirs by WRD. So, the diversity index (i.e., H′) was treated in the same manner as the other numerical metrics in the bisection scoring process used in this demonstration.

Category II–Feeding Measures and Trophic Dynamics

Metric 4–Number of piscivorous species—this metric discriminates between systems with high and moderate integrity. Piscivores are top carnivore species that feed principally on fish. Fish that are "occasional" piscivores, such as catfish (Ictalurids*),* are not included in this metric. Largemouth bass, which is the most popular sport fish in Lake Sinclair, were included along with hybrid striped bass, white bass, black crappie, warmouth, and yellow perch. Inferences can be drawn from knowledge of the capacity of the system to support the survival and propagation of the top carnivore. Without stable food dynamics, populations of top carnivores reflect stressed conditions. Biological conditions are considered good if the production rate at a site is high based on numerical abundance or biomass, and if high production is associated with the habitat type under study. The WRD uses a variation of this metric, *proportion of individuals as top carnivores*, in its protocol. As conveyed during Study Plan review, gar (*Lepisosteus* spp.) were excluded from this metric.

Metric 5–Percent invertivores—this metric increases as environmental quality increases due to the special dietary requirements of this trophic group and the limitations of their food source in degraded environments. Past studies have shown that the majority of Lake Sinclair's fish populations are invertivores. Most of Lake Sinclair's invertivores prefer littoral habitats, which include the area within the Primary Study Area most likely to be directly influenced by the buoyant thermal plume. As such, this metric should provide a responsive measure of trophic dynamics related to thermal effects.

Metric 6–Percent omnivores—this metric performs as a reverse scoring metric. Omnivores, including common carp and gizzard shad, *Dorosoma cepedianum,* are quite tolerant of environmental stresses due to their ability to vary their diets with food availability. If the food web becomes disrupted, omnivorous species generally increase in relative abundance as specialist species, such as invertivores, decline in number [14]. The WRD uses the same metric, *proportion of individuals as omnivores,* in scoring its wadeable stream assessment index.

Category III–Tolerance/Intolerance Metrics

Metric 7–Percent of individuals as tolerant species—this metric also performs as a reverse scoring metric as it's value increases as environmental conditions degrade, resulting in a community shift. Agency reviews of previous Lake Sinclair fish studies cited a concern about a potential shift in increased abundance of longnose gar, *Lepsosteus osseusi* and carp in the thermal discharge area. Based on WRD's input during study plan review, this metric includes common carp, longnose gar, golden shiner, *Notemigonus crysoleucas;* brown bullhead, *Ameiurus nebulosus;* and green sunfish as tolerant species.

Metric 8–Percent dominance (numerical percentage of most common species)—this metric measures the dominance of the single most abundant species. Healthy communities exhibit relatively balanced proportions of species and trophic groups represented. Percent dominance increases with declining species evenness.

Category IV–Reproductive Composition and Abundance

Metric 9–Number of Principal Species Exhibiting Young-of-Year Life Stages—During review of the Study Plan, WRD commented that a metric designated as *the number of intolerant species* might serve as a substitute *for the number of species exhibiting multiple life stages*. The number of intolerant species can provide a direct evaluation of habitat quality and the success of reproduction. Our rationale for not using intolerant species as a metric was based on the paucity of the number of intolerant species in Lake Sinclair and other southeast reservoirs. The intent of this functional category is to relate information about reproductive composition and abundance; therefore it was decided to use the number of principally important sport, commercial and prey species exhibiting YOY life stages. Principal species also represent Lake Sinclair's most abundant species and most important game fish. As such, this metric robustly integrates important information about productivity potential for the most important segment of the fish community.

Metric 10–Total number of individual fish, excluding shad—this metric evaluates community abundance and is based on the number of fish collected by all gear types combined and standardized to per-location abundance (Primary Study Area vs. reference area). Sites with lower integrity generally support fewer individuals. As recommended in the Study Plan, gizzard shad and threadfin shad, which account for a significant component of the Lake Sinclair forage base, were omitted from this metric. Species that are both tolerant and abundant are not expected to provide reliable response variables for an index designed to indicate degradation. Shad are often sporadically collected in large numbers due to their schooling behavior [15]. Their numbers can bias biotic integrity assessments because they do not exhibit random distributions. High numbers of shad in a sample have the effect of elevating index scores while obscuring patterns of the less dominant, yet important species. Following the recommendation of other investigators, shad were not included the analysis to avoid masking the assessment value of this metric [16]. Other species such as small-bodied cyprinids were considered in light of this concern as well. Cyprinids were not omitted in the metrics analysis because they occurred fairly uniformly throughout Lake Sinclair.

Metric 11–Fish Density (number/hectare)—this metric provides a measure of the abundance of limnetic fishes which accounts mostly for the component of shad that was excluded from Metric 10. Hydroacoustic data provided the basis for this metric. Shad comprise the principal component of the limnetic fishery in Lake Sinclair based on the hydroacoustic survey results and represent abundant forage for important piscivorous sport-fish predators. The hydroacoustic surveys provided a tool to assess the shad component of Lake Sinclair in terms of fish density (numbers of fish per hectare). This metric directly compares and scores the average density of limnetic fishes between the Primary Study Area and the reference area.

Category V–Fish Health

Metric 12–Percent Incidence of DELT (disease, fin erosion, skin lesions, or tumors)—This metric depicts the health and condition of individual fish. These conditions occur infrequently or are absent from minimally impacted sites. The WRD uses a variation of this metric, *proportion of individuals as diseased fish*, in application of their small stream fish assessment protocol. Where fish health anomalies exceed 1.2 percent of the sample population, a negative score (- 4 points) is applied to the fish IBI score for that location. If the percent DELT is equal to or less than 1.2 percent, this metric is scored as a zero (0). Based upon a review of the literature, most applications of biotic indices include a fish health metric similar to that described here for Metric 12, which has been shown to be a widely applicable, less subjective, and responsive metric.

Metric Scoring Methods

In most cases, the study expectation was that the reference area sample sites would yield a range of values for a given metric suitable for routine application of the typical 25th percentile scoring method. Region-wide applications of biotic indices sometimes discard metrics that exhibit little or no range of values. In those applications, measuring the biotic integrity of test sites against the *regional* expectation is paramount and directly related to the tested ability of a metric to correlate with a range of known anthropogenic stressors. This demonstration is non-traditional in the sense that the reference condition in Lake Sinclair was developed from areas within the lake that met the non-thermal definition established specifically for the study. This multimetric index becomes a standardized tool to facilitate comparison between the test site (Primary Study Area) and the reference area, rather than to compare the test site attributes against a region-wide expectation.

In certain instances involving positive scoring numerical metrics, it was determined that a narrow range of metric values existed between reference area sampling locations. In those cases, establishing a 25th percentile scoring criterion was either not mathematically possible or a fractional number resulted that was unusable for scoring a whole-integer metric. In these few instances, best professional judgment was used in conjunction with scoring guidance recommendations. Where the reference range was no larger than one integer (e.g., separation by only one species), scoring was applied using professional judgment as in the following hypothetical example.

Assuming the reference area exhibited a range of 5 to 6 temperature sensitive species (among reference sampling locations), a calculated 25th percentile of the range yields an actual value somewhere between 5 and 6 species. Because a fractional value for this metric is not logical, either "5" or "6" species could serve as the "rounded", practical scoring decision benchmark to follow based on professional judgment.

Bisection scoring guidance indicates that test site scores equal to or less than the 25th percentile value are impaired to some degree [5]. To receive a 5-point score in the previous example the test site metric value should be greater than the 25th percentile value. Assuming five species represented the 25th percentile, if the test site exhibited six or more temperature sensitive species, the protocol would specify that the metric receive a score of 5 points. If the test site exhibits five species it would receive a score of 3 points. If the test site exhibits 1 to 4 species, it would receive a metric score of 1 point. If no temperature sensitive species are collected at the test site, the metric score would be 0 points. Strict application of the scoring protocol in this manner would seemingly result in an overly conservative metric score for a test site (e.g., the Primary Study Area) if five of the six species occurred (maximum expected). In such instances, during the current study, professional judgment was exercised and the metric would receive a compromise score of 4 points.

In instances when all reference area values were the same for a particular numerical metric, calculation of a 25th percentile scoring criterion was not possible due to the absence of a reference range. Therefore based on professional judgment, scoring decision criteria were fitted into the index scoring matrix by assigning a score of 5 points for the test site metric if its value was equal to or greater than the reference area value. For example, if the reference expectation is four intolerant species then the test site would receive a score of 5 points if it exhibits four or more intolerant species. If the test site were to yield three of the four intolerant species observed in the reference condition, it would receive a metric score of 3 points and a score of 1 point if one or two intolerant species are collected. If no intolerant species were found in the test site catch, the metric would receive a score of zero.

It is anticipated that the ranges of reference values may change (expand or decrease) as new data are collected and examined during the second year of study in 2003.

Index Period Score

The fish bioassessment index score for the Primary Study Area during the August 2002 Index Period was 43 points. Based on the WRD integrity class ranges, an index score of 43 points places the biological condition of the Primary Study Area between the "Fair" Integrity Classification (34 to 42 points) and the "Good" classification (44 to 50 points; Table 10-6). Six of the thirteen fish community metrics (Metrics 1, 3, 7, 8, and 11) received maximum five-point scores. Metric 3–*number of temperature sensitive species*, received a compromise score of 4, having just one less temperature sensitive species, a managed species (hybrid striped bass) represented compared to the reference condition. Metrics 2 and 9 each received metric scores of 3 points, indicating relatively lower percent sportfish and numbers of principal species representing YOY life stages in the Primary Study Area compared to the reference area. Metrics 5, 6, and 10 each received a score of one point. These metrics represent the trophic dynamics and abundance functional ecological categories and indicated that the proportions of invertivores and omnivores and fish abundance (excluding shad) in the Primary Study Area were different from the reference area.

Metric 13, the DELT fish health measure, received a score of zero, reflecting a low incidence of anomalies observed in the Primary Study Area during the Index Period.

Each metric was represented in schematic form, an example of which is provided in Figure 10-9. Schematics provided key information used to establish the reference condition scoring ranges.

Seasonal Index Scores

Bioassessment index scores for the Primary Study Area were calculated for winter, spring, and fall seasons based on reference conditions developed for each season. As indicated previously, sampling effort in the reference areas during these three seasons was reduced from that conducted during the August 2002 Index Period. However, effort was sufficient to provide the data necessary to determine seasonal biotic integrity of the fish community in the Primary Study Area.

Seasonal results were scored using the same method applied during the Index Period. The Primary Study Area fish community index scored 42 (Fair), 56 (Excellent), and 38 (Fair) in winter, spring, and fall, respectively based on the WRD Integrity Classifications (Table 10-7 and Figure 10-10).

Table 10-6
Fish Community Bioassessment Index Score Sheet for the Primary Study Area in Lake Sinclair for August 2002 Index Period

Index Period Worksheet				Score Category						
Metric Description		Unit of measure	Primary Study Area result	5	4	3	1	0	Fish Health (–4 pts)	Metric Score
1	Total number of species > 25 mm	no. of species	24	> 20	–	19 - 20	< 19	–	–	5
2	Percent of sportfish species	percent	41.5	> 45.5	–	> 33.0 < 45.6	< 33.1	–	–	3
3	Number of temperature sensitive species	no. of species	3	> 2	–	2	1	0	–	5
4	Number of piscivorous species	no. of species	5	> 6	5	4	> 1< 4	0	–	4
5	Percent invertivores	percent	40.5	> 47.8	–	> 44.3 < 47.9	< 44.4	–	–	1
6	Percent omnivores	percent	40.0	< 26.8	–	> 26.7 < 27.8	> 27.7	–	–	1
7	Percent of tolerant species	percent	1.8	< 2.6	–	> 2.5 < 3.4	> 3.3	–	–	5
8	Percent single most dominant species	percent	28.7	< 37.9	–	> 37.8 < 38.8	> 38.7	–	–	5
9	Number of principal species exhibiting young-of-year life stage	no. of species	8	> 8	–	8	1 - 8	0	–	3
10	Number of individual fish excluding shad	no. of fish	765	> 930	–	> 856 < 931	< 857	–	–	1
11	Fish Density	no. of fish/ha	6916	> 4378	–	> 3488 < 4379	< 3489	–	–	5
12	Fish health (Percent DELT)	percent	0.03	–	–	–	–	< 1.2	> 1.2	0
13	Shannon-Wiener diversity index (H')	H' values	2.19	> 2.10	–	> 2.04 < 2.10	< 2.04	–	–	5
Fish Bioassessment Index Score (August 2002 Index Period)										**43**
WRD Integrity Class									**Fair/Good**	

WRD Fish Community Integrity Classes and descriptions developed for wadeable streams	**Points**
Excellent - comparable to best situation expected.	60 - 52
Good - species richness below expectation, especially due to loss of most intolerant species.	50 - 44
Fair - species richness declines as some expected species are absent; trophic structure skewed toward generalist species.	42 - 34
Poor - sample dominated by omnivores, tolerant, and pioneer species, sensitive species absent; condition factors commonly depressed.	32 - 26
Very Poor - few fish present, mostly tolerant and pioneer species, fish with disease, eroded fins, and tumors common.	24 - 8
No Fish.	No Fish

Index Period Result

Result	Metric Value
Reference Area Samples	
LR1	24
LR2	23
OU1	24
OU2	20
OD1	17
OD2	21
25th Percentile of Reference	20
Primary Study Area Result	24

Score Criteria* based on Bisect Scoring Method	
5 Points for values >	20
3 Points for values >	19 ≤ 20
1 Point for values <	19

PSA score for this Metric	5 Points

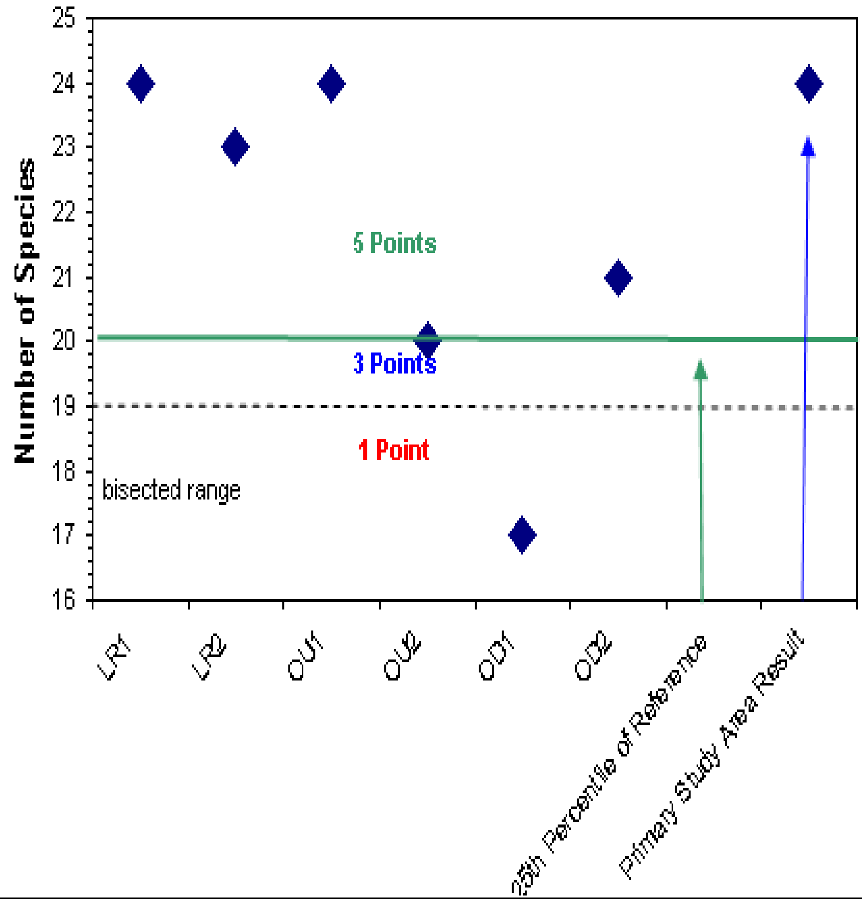


Figure 10-9
Schematic of Fish Community Metric 1–Number of All Species > 25 mm

Table 10-7
Summary of Seasonal Fish Community Bioassessment Index Scores for the Primary Study Area of Lake Sinclair, 2002

Fish Community Metrics		January	May	August*	November
1	Total number of species > 25 mm	5	5	5	5
2	Percent of sportfish species	1	5	3	1
3	Number of temperature sensitive species	5	5	5	5
4	Number of piscivorous species	5	5	4	5
5	Percent invertivores	1	5	1	1
6	Percent omnivores	3	1	1	1
7	Percent of tolerant species	1	5	5	3
8	Percent single most dominant species	5	5	5	5
9	Number of principal species exhibiting young-of-year life stage	5	5	3	5
10	Number of individual fish excluding shad	1	5	1	1
11	Fish Density	5	5	5	1
12	Fish health (Percent DELT)	0	0	0	0
13	Shannon-Wiener diversity index (H')	5	5	5	5
Seasonal Fish Community Index Scores		**42**	**56**	**43**	**38**
WRD Integrity Class		**Fair**	**Excellent**	**Fair/Good**	**Fair**

WRD Fish Community Integrity Classes	Scoring Range
Excellent	60 - 52
Good	50 - 44
Fair	42 - 34
Poor	32 - 26
Very Poor	24 - 8
No Fish	No fish

Notes:

* – August Index Period

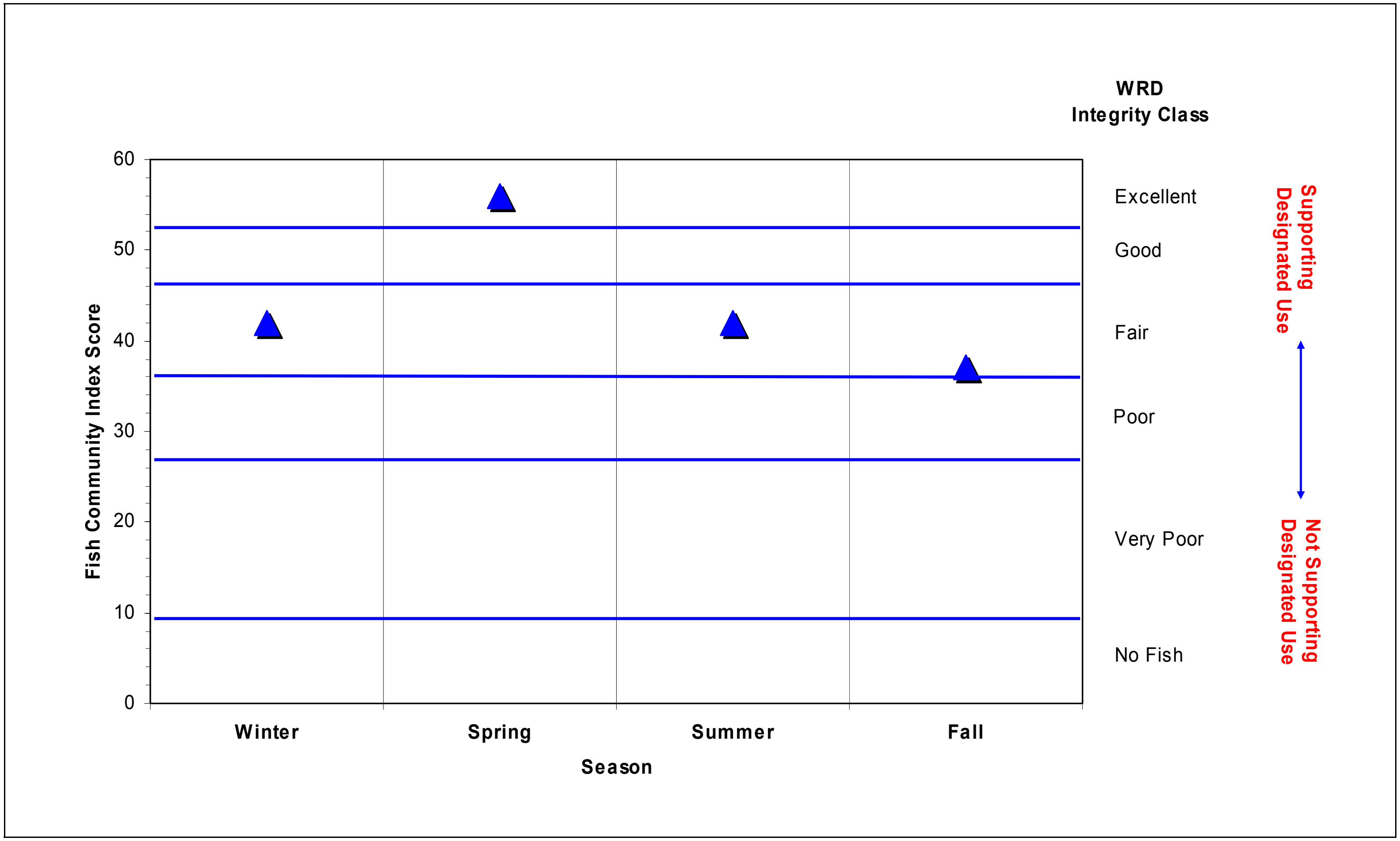


Figure 10-10
Seasonal Fish Community Bioassessment Index Scores for the Primary Study Area in Lake Sinclair, 2002

Index Relationship to Empirical Data

Metric score results as supported by empirical data lend perspective and weight of evidence to the overall biotic integrity scores. At times, individual metric scores may overstate a condition, which based upon the empirical data, seems largely benign. Factors likely affecting individual metric scores included seasonal attraction to the Primary Study Area, shifts in seasonal abundance due to springtime activities, potential gear selectivity in spring, and indications of trophic fluctuations in two classes including omnivores and invertivores. The biotic integrity index scores for 2002, particularly for the critical Index Period, were indicative, at an integrated level of detail, of a fish community present in the Primary Study Area that was comparable to the reference condition. These results are supported by the empirical data that indicated generally consistent similarities in spatial and seasonal variability of the fish community among study areas.

Biological Integrity Integration

A balanced, indigenous aquatic community includes the macroinvertebrate and fish communities in a collective context, not just independently. As such, the biological integrity scores for each community were combined (by averaging the two results) to arrive at an Integrated Biotic Community Index (IBCI) to fully address the overall concept of a balanced, indigenous aquatic community in the Primary Study Area of Lake Sinclair during the August 2002 Index Period and other seasons. This was accomplished by normalizing the EPD protocol-based index score for macroinvertebrates based on a maximum 35 point scale to the WRD scale of 60 total points, then averaging the two index scores (i.e., for macroinvertebrate and fish communities) to arrive at the IBCI (Table 10-8). The resulting IBCI score for the Primary Study Area during the Index Period was 48 points, and for other seasons 48 (winter), 55 (spring), and 48 (fall; Figure 10-11). The figure demonstrates how the IBCI scores compare individually to the EPD (60-point normalized) and WRD Integrity Classifications, both which were developed for Georgia streams. The Primary Study Area IBCI rated a "Good/Good" in the August Index Period, "Good/Good" during winter and fall seasons and an "Excellent/Very Good" rating in the spring under the WRD/EPD classification schemes, respectively.

Site-Specific Integrity Classifications

Considering that the EPD and WRD Integrity Classifications were developed for aquatic communities in streams rather than reservoirs, site-specific classifications were developed using established protocols specifically for Lake Sinclair based on data collected exclusively in the reference areas during the Index Period. This was accomplished by simply "scoring" the 25^{th} percentile condition for the reservoir based on the distribution of individual metric values (fish and macroinvertebrate independently) at each of the reference locations (Table 10-9 and Table 10-10). In this manner, the 25^{th} percentile condition, which represents the minimum expectation for the reference condition, is translated to a biotic index score. The index score for that area of Lake Sinclair, included in the current study, *located outside of the Primary Study* is 36 points for the fish community and 21 for the macroinvertebrate community, based on WRD and EPD scoring values, respectively.

Table 10-8
Summary of Seasonal Macroinvertebrate and Fish Bioassessment Scores and Integrated Biotic Community Index (IBCI) Scores for the Primary Study Area in Lake Sinclair, 2002

Index Score	Winter	Spring	Summer	Fall
Total Macroinvertebrate Score (standard EPD scoring)	31	31	31	33
"Normalized" Macroinvertebrate Score(1)	53	53	53	57
Fish Community Index Score (standard WRD scoring)	42	56	43	38
IBCI Score(2)	**48**	**55**	**48**	**48**
IBCI Integrity Class	**Good**	**Very Good**	**Good**	**Good**

IBCI Integrity Classes	**EPD scoring range[4]**	**EPD scoring range normalized to 60 points for IBCI**
Very Good/Excellent	35-31	60-52
Good	30-25	51-44
Fair	24-19	43-29
Poor	18-14	28-15
Very Poor	< 14	< 15

Note:
(1)The scoring range of 35 points has been normalized to a range of 60 points.
(2)The IBCI score is the average of the normalized macroinvertebrate score and the fish score.

IBI scores exceeding the 25th percentile of the reference condition mark the beginning of the "Good" Integrity Class [5, 17, 18]. In Lake Sinclair, fish community index scores of 36 points demarcate the beginning of the "Good" Integrity Class. Guidance on establishment of the remaining integrity classes (e.g., "Excellent/Very Good", "Very Poor", "Poor", and "Fair" is less defined, and more often than not, demarcated by resource agencies based on best professional judgment. Beginning with the protocol-based starting point for a "Good" classification, the remaining site-specific integrity classes were determined in the current study for both fish and macroinvertebrate communities by first scoring each individual reference location against the reference condition in the same manner the Primary Study Area was scored as an individual location (Table 10-11 and Table 10-12). This exercise provided the range of index scores for individual locations in the reference area of the reservoir.

For the fish community, a sampling station in the Little River arm of Lake Sinclair had the highest index score with 54 total points; therefore, 54 was selected based on best professional judgment as the value demarcating the "Excellent" classification (Table 10-11). Therefore, scores between 36 and 53 define the "Good" classification. Because all reference locations and Primary Study Area index scores exceeded minimum expectations (i.e., a score of 36), there was no firm basis for demarcating classes that were less than "Good". Consequently, professional judgment was applied and the area below "Good" was divided equally into the three remaining integrity classes.

For the macroinvertebrate community, nine of the 16 locations sampled had index scores of 31 to 35 total points representing the highest reference scoring locations in Lake Sinclair. Therefore, 31 was selected based on best professional judgment as the breakpoint value demarcating the beginning of the "Very Good" classification (Table 10-12). Fifteen of 16 reference locations and Primary Study Area index scores exceeded the minimum reference expectations (i.e., a score of 21 demarcating the "Good" classification); however, one location scored less than the 25th percentile reference condition (19 points). With so few data points below the "Good" classification, professional judgment was applied and the area below "Good" was divided equally into the three remaining integrity classes.

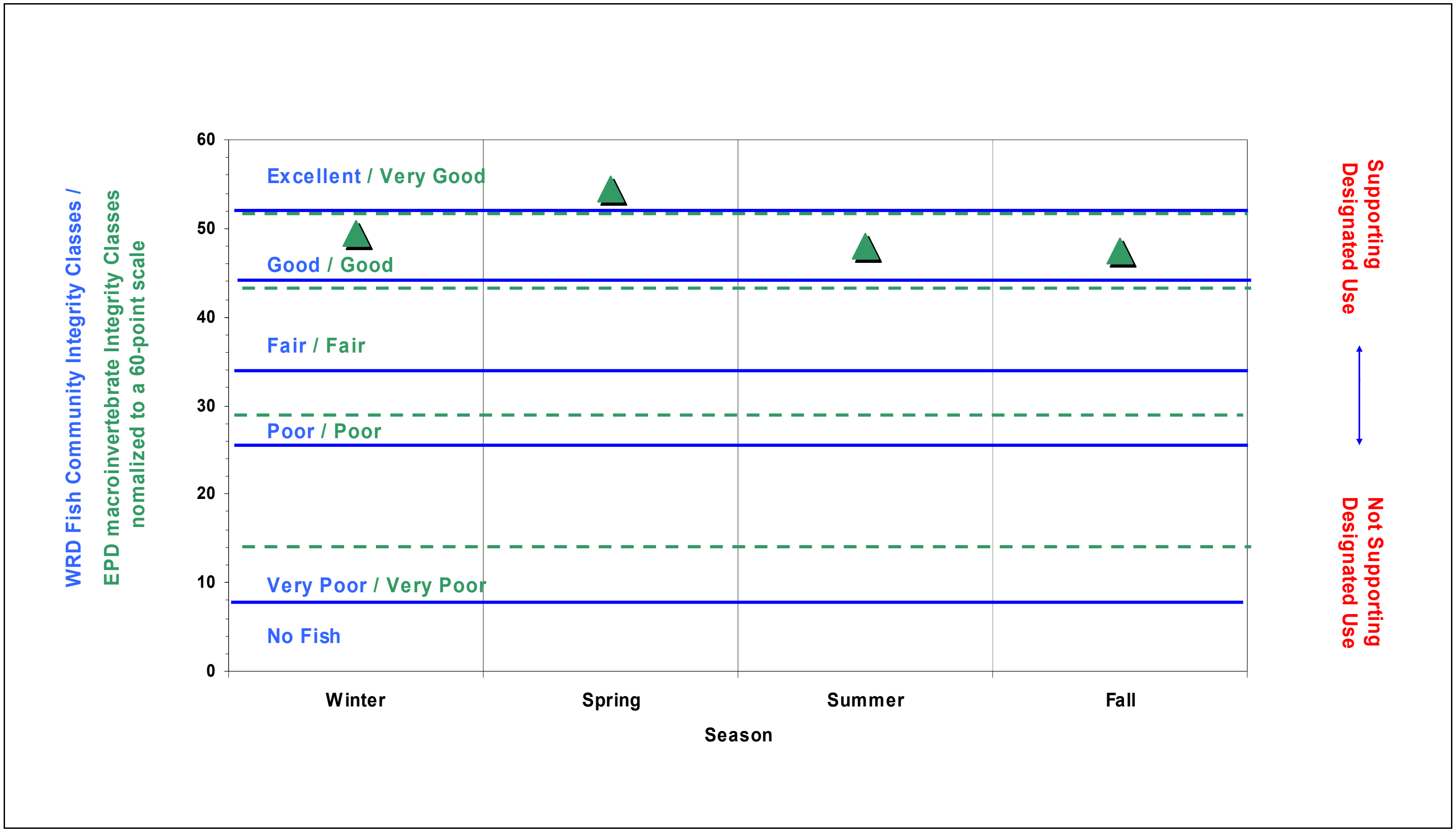


Figure 10-11
Seasonal Integrated Biotic Community Index (IBCI) Scores for the Primary Study Area of Lake Sinclair, 2002

Table 10-9
Fish Community Bioassessment Index Score Sheet Used to Establish the Minimum Reference Condition Expectation for the Fish Community in Lake Sinclair for the August 2002 Index Period

Index Period Reference Condition Worksheet				Score Category						
	Metric Description	**Unit of measure**	**25th/75th percentile benchmark result**	**5**	**4**	**3**	**1**	**0**	**Fish Health** (–4 pts)	**Metric Scores**
1	Total number of species > 25 mm	no. of species	20	> 20	–	19 - 20	< 19	–	–	3
2	Percent of sportfish species	percent	45.5	> 45	–	> 33.0 < 46	< 33.1	–	–	3
3	Number of temperature sensitive species	no. of species	2	> 2	–	2	1	0	–	3
4	Number of piscivorous species	no. of species	5	> 6	5	4	> 1< 4	0	–	3
5	Percent invertivores	percent	47.8	> 47.8	–	> 44.3 < 47.9	< 44.4	–	–	3
6	Percent omnivores	percent	26.8	< 26.8	–	> 26.7 < 27.8	> 27.7	–	–	3
7	Percent of tolerant species	percent	2.6	< 2.6	–	> 2.5 < 3.4	> 3.3	–	–	3
8	Percent single most dominant species	percent	37.9	< 37.9	–	> 37.8 < 38.8	> 38.7	–	–	3
9	Number of principal species exhibiting young-of-year life stage	no. of species	8.0	> 8	–	8	1 - 8	0	–	3
10	Number of individual fish excluding shad	no. fish	930	> 930	–	> 856 < 931	< 857	–	–	3
11	Fish Density	no. of fish/ha	4378	> 4378	–	> 3488 < 4379	< 3489	–	–	3
12	Fish health (Percent DELT)	percent	0.09	–	–	–	–	< 1.2	> 1.2	0
13	Shannon-Wiener diversity index (H')	H' values	2.10	> 2.10	–	> 2.04 < 2.10	< 2.04	–	–	3
Minimum Reference Index Score Expectation for Fish Community										**36**

Site-specific Integrity Classes based on Lake Sinclair's reference condition	**Points**
Excellent - comparable to best reference situation expected based on index scores for locations within the reference area.	60 - 54
Good - comparable to reference conditions (> 25th percentile of Reference Condition but < Excellent category.	53 - 37
Fair - somewhat impaired; loss of some sensitive and/or principal species anticipated.	36 - 25
Poor - impaired; decreased species richness and diversity; dominated by tolerant species.	24 - 13
Very Poor - severely degraded conditions.	12 - 0

Table 10-10
Macroinvertebrate Community Index Score Sheet Used to Establish the Minimum Reference Condition Expectation for the Macroinvertebrate Community in Lake Sinclair for the August 2002 Index Period

Macroinvertebrate Reference Condition Worksheet				Score Category			
	Metric Description	**Unit of Measure**	**25th/75th percentile benchmark result**	**5**	**3**	**1**	**Metric Score**
1	Total number of taxa	no. of taxa	33	> 33	27-33	< 27	3
2	Number of ETO taxa	no. of taxa	5	> 5	4-5	< 4	3
3	Number of dipteran taxa	no. of taxa	15	> 15	13-15	< 13	3
4	Percent chironomidae	percent	76%	< 76	76-79	> 79	3
5	Percent filterers	percent	32%	> 32	29-32	< 29	5
6	North Carolina Biotic Index	NCBI value	8.2	< 8.20	8.30-8.20	> 8.30	3
7	Percent single most dominant taxon	percent	32%	< 27	27-37	> 37	3
Minimum Reference Index Score Expectation for Macroinvertebrate Community							**21**
Site-specific Macroinvertebrate Integrity Classes based on Lake Sinclair's reference condition							**Points**
Very Good - comparable to best reference conditions in Lake Sinclair.							35-31
Good - balanced community, but with slightly lower taxa richness.							30-22
Fair - expected taxa present in low abundance and dominance of community by a few tolerant taxa.							21-15
Poor - low tax richness, expected species absent, and community dominated by tolerant taxa.							14-8
Very Poor - low taxa richness and low abundance.							<8

Table 10-11
Individual Reference Location Fish Community Index Scores in Lake Sinclair for the August 2002 Index Period

Fish Community Metrics		Individual Reference Location Metric Scores					
		OU2	**OU1**	**OD1**	**LR2**	**OD2**	**LR1**
1	Total number of species	3	5	1	5	5	5
2	Percent of sportfish species	3	1	5	5	5	5
3	Number of temperature sensitive species	3	5	3	5	5	5
4	Number of piscivorous species	3	5	3	5	5	3
5	Percent invertivores	1	1	5	5	5	5
6	Percent omnivores	5	1	5	3	5	5
7	Percent of tolerant species	1	5	5	3	5	5
8	Percent single most dominant species	5	5	1	1	5	5
9	Number of principal species exhibiting young-of-year life stage	5	5	5	3	3	5
10	Number of individual fish excluding shad	5	5	5	5	3	1
11	Fish Density	1	1	5	5	5	5
12	Fish health (Percent DELT)	0	0	0	0	0	0
13	Shannon-Wiener diversity index (H')	5	5	1	5	1	5
Reference Location Index Scores		**40**	**44**	**44**	**50**	**52**	**54**

Site-specific Integrity Classes based on Lake Sinclair's reference condition	**Points**
Excellent - comparable to best reference situation expected based on index scores within the reference area.	60 - 54
Good - comparable to reference conditions (> 25th percentile of reference condition and < Excellent category).	53 - 37
Fair - slightly impaired; loss of some sensitive and/or principal species.	36 - 25
Poor - impaired; decreased species richness and diversity; dominated by tolerant species anticipated.	24 - 13
Very Poor - severely degraded conditions.	12 - 0

Notes:

OD – downstream Oconee arm reference fish community sampling locations.

OU – upstream Oconee arm reference fish community sampling locations.

LR – Little River arm reference fish community sampling locations.

Table 10-12
Individual Reference Location Macroinvertebrate Community Index Scores in Lake Sinclair for the August 2002 Index Period

Macroinvertebrate Community Metrics		Individual Reference Location Metric Scores															
		BMLR01	BMLR02	BMLR03	BMLR04	BMLR05	BMOD01	BMOD02	BMOD03	BMOD04	BMOD05	BMOU01	BMOU02	BMOU03	BMOU04	BMOU05	BMOU06
1	Number of taxa	5	5	5	3	5	1	5	1	5	5	5	3	5	5	5	5
2	Number of ETO taxa	5	3	5	3	5	1	3	3	3	5	5	5	5	5	5	5
3	Number of dipteran taxa	5	5	5	3	3	1	5	1	5	3	5	5	5	5	5	5
5	Percent chironomidae	5	5	5	3	1	5	3	5	5	5	5	3	5	5	5	5
4	Percent filterers	5	1	5	5	5	5	5	5	5	1	3	1	5	5	5	5
6	North Carolina Biotic Index	5	5	5	1	5	5	5	5	5	5	3	3	5	5	3	5
7	Percent single most dominant taxon	5	5	5	5	1	1	5	5	5	5	5	5	5	1	5	3
Reference Location index scores		**35**	**29**	**35**	**23**	**25**	**19**	**31**	**25**	**33**	**29**	**31**	**25**	**35**	**31**	**33**	**33**

Site-Specific Macroinvertebrate Integrity Classes Based on Lake Sinclair's Reference Condition	Points
Very Good - comparable to best reference conditions in Lake Sinclair.	35-31
Good - balanced community, but with slightly lower taxa richness.	30-22
Fair - expected taxa present in low abundance and dominance of community by a few tolerant taxa.	21-15
Poor - low tax richness, expected species absent, and community dominated by tolerant taxa.	14-8
Very Poor - low taxa richness and low abundance.	<8

Notes:

OD – downstream Oconee arm reference fish community sampling locations.

OU – upstream Oconee arm reference fish community sampling locations.

LR – Little River arm reference fish community sampling locations.

ETO denotes Ephemeroptera, Trichoptera, and Odonata taxa.

The fact that all the individual fish community and macroinvertebrate reference locations scored in the "Good " or better categories in this site-specific exercise indicates that the reference condition, to which the Primary Study Area was compared, represents the "best attainable condition" for Lake Sinclair outside of the Primary Study Area. This may not have been the case, had other areas known to be degraded (e.g., Rooty Creek and Crooked Creek embayments) been included in the determination of reference conditions.

Figure 10-12 illustrates the site-specific integrity classes for the Primary Study Area during the August 2002 Index Period. Independent biotic index scores are provided for macroinvertebrate (scale normalized) and fish communities. Based on the Lake Sinclair site-specific Integrity Classifications developed using established protocols, the Primary Study Area index scores for the macroinvertebrate (53; 31 normalized to the 60-point scale) and fish (43) communities during the Index Period rated a "Very Good" and "Good" integrity classifications, respectively indicating conditions comparable to those found in the reference areas of the reservoir.

Bioassessment Summary

A multimetric bioassessment was applied to the macroinvertebrate and fish community data collected from the Primary Study Area and the reference area of Lake Sinclair during 2002. Seasonal biological data sets were used concurrently with the habitat assessment data to evaluate comparability between study areas and to determine if a balanced, indigenous community was present in the Primary Study Area. Biotic index scores were developed independently for macroinvertebrates and fish relying on a combination of USEPA's Reservoir Bioassessment Guidance, Georgia's macroinvertebrate and fish bioassessment protocols for wadeable streams adapted to a reservoir setting, and as provided by those protocols, use of professional judgment [5, 4, 7].

Seasonal macroinvertebrate index scores for the Primary Study Area were 33 in the winter, 31 in the spring, 33 in the fall, and 31 during the August 2002 Index Period. All macroinvertebrate Index Scores were rated "Good" or "Very Good" based on EPD's stream-based Integrity Classification system. Index Scores for the fish community were 42 in the winter, 56 in the spring, 43 during the Index Period, and 38 in the fall. Based on WRD's Integrity Classification system similarly developed for streams, the Primary Study Area rated "Fair" in winter and fall, "Excellent ("Very Good") in spring, and "Good to Fair" during the Index Period. All integrity class rankings for the macroinvertebrate and fish communities indicated that designated uses were supported in the Primary Study Area.

Because a balanced, indigenous aquatic community includes the macroinvertebrate and fish communities collectively, biotic community indices were integrated to provide a holistic assessment of the condition of the biological community in the Primary Study Area. Overall, the integrated biotic community integrity of the Primary Study Area was rated "Very Good" for the August 2002 critical condition Index Period, as based on the EPD (normalized) and WRD Integrity Classifications.

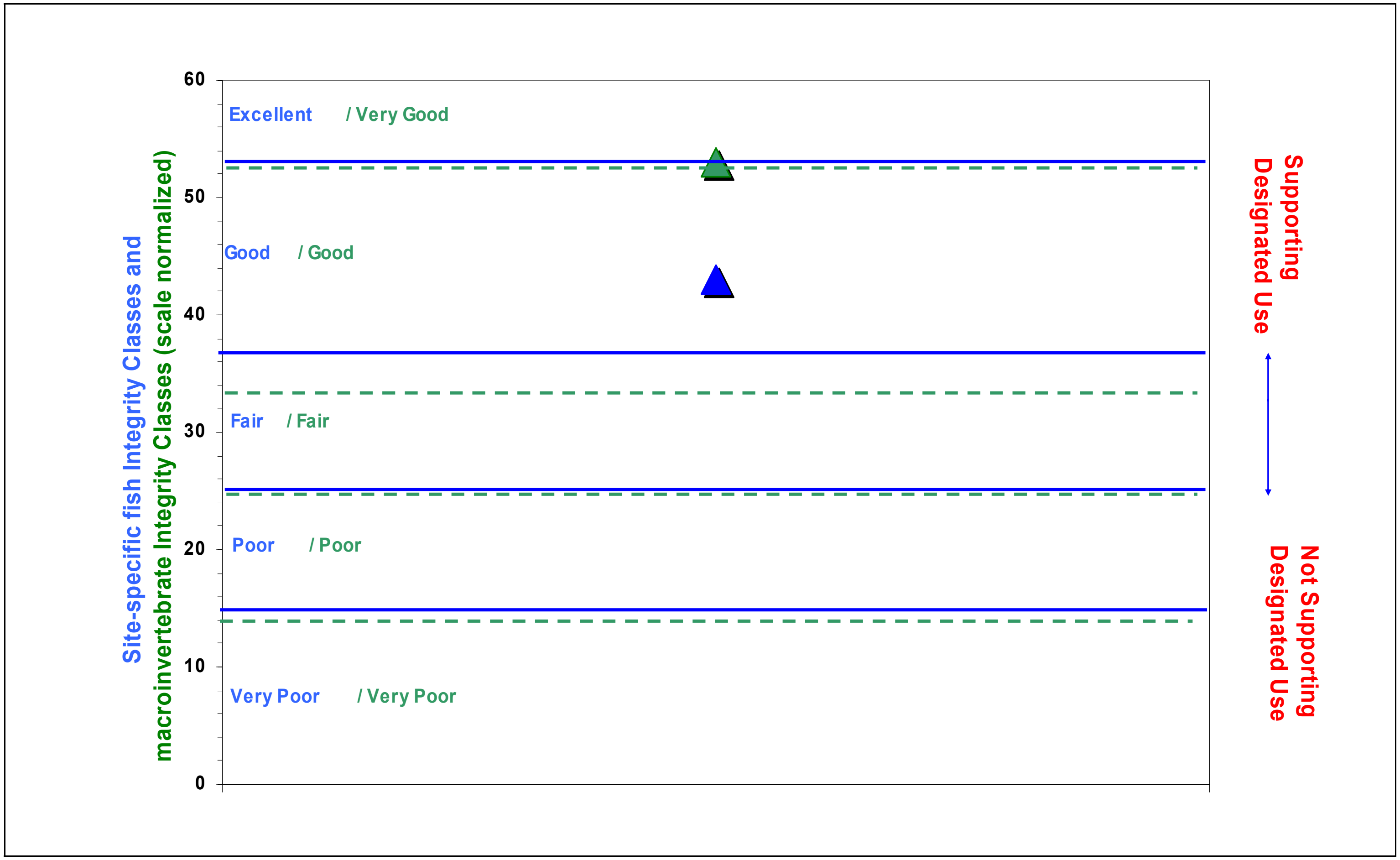


Figure 10-12
Fish and Macroinvertebrate Community Bioassessment Index Scores for the August 2002 Index Period Compared to Site- and Community-Specific Integrity Classes

When compared to Lake Sinclair site-specific Integrity Classifications developed using established protocols, the Primary Study Area index scores for the macroinvertebrate and fish communities during the Index Period rated a "Very Good" and "Good" integrity classifications indicating conditions comparable to those found in the reference areas of the reservoir.

Closing

This paper presents the results of a multimetric bioassessment conducted in 2002 by Georgia Power Company on Lake Sinclair for the purpose of developing a CWA Section 316(a) demonstration to obtain a thermal variance for Plant Branch. Studies were conducted based on an EPD approved Study Plan having been subject to review by the MOA participants.

Georgia Power's goal through the implementation of this study is to determine: 1) if the characteristics of a balanced, indigenous aquatic community exist within the Primary Study Area of Lake Sinclair; or 2) if there is evidence of previous harm, that the protection and propagation of a balanced, indigenous aquatic community will nevertheless be assured under alternate thermal limits combined with summer operation of the Plant Branch cooling system.

Based on the results of the 2002 bioassessment, supported by the empirical biological community data, the study demonstrated that designated uses are supported and the protection and propagation of a balanced, indigenous aquatic community is being maintained in the Primary Study Area of Lake Sinclair. Notably, such conditions were supported during a time of extended drought, above normal air temperatures, and high electrical demand. The second year of the demonstration study is expected to provide a more robust dataset with which to further evaluate the affects of the Plant Branch discharge on the aquatic biota of Lake Sinclair.

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11
MANAGING COOLING LAKE FISHERIES IN PARTNERSHIP WITH THE ILLINOIS DEPARTMENT OF NATURAL RESOURCES (IDNR)

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Introduction

Power plant cooling lakes provide high quality fisheries and recreational opportunities for the citizens of Illinois and other neighboring States. On December 16, 2002 Exelon Generation and the Illinois Department of Natural Resources (IDNR) signed new lease arrangements allowing public recreation access to the Company's Braidwood, Clinton and LaSalle County Station cooling lakes. The new lease agreements expire in September, 2026 for Clinton Lake, in October, 2026 for Braidwood Lake and in April, 2025 for LaSalle Lake. With the agreements, the citizens of Illinois and visitors to the State can enjoy boating, fishing and other recreational opportunities.

Text

The Illinois Department of Natural Resources (IDNR) began managing land at the 9,267 acre Clinton Lake State Recreation Area at Clinton Lake in 1979. Clinton Lake is located nearby Clinton, Illinois in DeWitt County. In addition to boating, swimming, fishing and waterfowl hunting on the 4,900 lake, the site features upland game hunting, hiking, camping, picnic facilities and horseback riding.

Braidwood Lake is another cooling lake located near Braidwood, Illinois in Will County. The Braidwood Cooling lake is situated on 4,454 acres of flat agricultural farmland that has been scarred from coal strip mining. IDNR began managing recreational land at the Braidwood State Fish and Wildlife Area complex at Braidwood Lake in 1981. The 3,005 acres available for public use feature fishing, waterfowl hunting, fossil hunting by permit and serve as a waterfowl refuge.

LaSalle County Lake is located near Seneca, Illinois in LaSalle County. Fishing is the key attraction at the 2,058 acre LaSalle County Lake State and Wildlife Area at LaSalle Lake, that was opened by IDNR for public fishing in 1986.

On December 16, 2002 Exelon Generation and the Illinois Department of Natural Resources (IDNR) signed new lease arrangements allowing public recreation access to the Company's Braidwood, Clinton and LaSalle County Station cooling lakes.

The challenge facing Exelon Generation IDNR was twofold: (1) find a way to maintain cooling lake fisheries for the people of Illinois in a changing world of industry deregulation and security changes and (2) figure out a way to provide quality fisheries following higher plant capacity factors and power up-rates that added stress to fish populations. The new lease agreements expire in September, 2026 for Clinton Lake, in October, 2026 for Braidwood Lake and in April, 2025 for LaSalle Lake. With the agreements, the citizens of Illinois and visitors to the State can enjoy boating, fishing and other recreational opportunities.

The cooling lake leases specify responsibilities for both IDNR and Exelon Generation. IDNR is to manage the lakes in accordance with the Cooling Lake Management Plan and the Cooling Lake Evacuation Plan, both of which are reviewed and renewed annually. The leases provide a written plan on how to manage, protect, sustain and promote Illinois' natural resources. They also identify and address land management opportunities and priorities for these three cooling lakes.

The Cooling Lake Management Plan (CLMP) addresses present and future fish and wildlife management strategies for the cooling lakes that ensure consistency with the goal of maintaining public recreation on waters with anticipated increased heat loads. The CLMP is an addendum to the lease with IDNR for each cooling lake and it describes management and monitoring activities undertaken by Exelon Generation and IDNR to meet their respective and combined objectives. The CLMP establishes responsibilities for each party. The CLMP calls for an annual meeting where parties review accomplishments for the past year and plan for what is expected during following year. These plans include a review of fish stocking plans and the number and selected species for stocking in the cooling lakes. A detailed work plan is then developed for each activity that requires resources, equipment or funding.

The Cooling Lake Evacuation Plan is also an addendum to the lease for each cooling lake. It describes the responsibilities and notifications to be implemented in the event of a required evacuation of fisherman and boats, including shoreline fishermen from the cooling lake while at the same time safeguarding lives and ensuring protection of plant property.

Exelon Generation dedicates resources who work hard to maintain a close working relationship with IDNR and who regularly communicate lake temperature conditions and predictions of lake performance via weekly telephone calls and e-mails during the summer months when lake temperatures can be most stressful to fish.

12
PSEG 316(A) STUDY EXPERIENCE AT SALEM AND HUDSON GENERATING STATIONS

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Introduction

Concern regarding thermal discharges resulted in the passage of Section 316(a) of the Clean Water Act described by Bulleit in this volume [1]. Section 316(a) allows a variance from effluent limitations that would otherwise be imposed to meet water quality standards provided the proposed alternative effluent limitations are protective of the balanced indigenous population or community (BIP). The applicant must demonstrate that the thermal discharge creates a thermal plume which neither blocks the natural migration of fish, causes significant lethality, excludes species from large areas of the receiving waterbody, adversely affects protected species, depletes oxygen levels to unacceptable levels, alters the structure of the BIP, nor increases the toxicity of other contaminants.

This paper describes biothermal assessments that Public Service Enterprise Group (PSEG) completed for its Hudson and Salem Generating Stations. Hudson is located on the Hackensack River estuary, in Hudson County, New Jersey. Salem is located on the Delaware River estuary, in Salem County, NJ (Figure 12-1). Both stations use a once-through cooling system.

The New Jersey Department of Environmental Protection (NJDEP) issued a New Jersey Pollutant Discharge Elimination Program (NJPDES) Permit (NJ 0000647) for the Hudson Generating Station (Hudson) requiring a Section 316(a) Demonstration. As part of the application to renew its 1990 NJPDES permit and continuation of a variance according to Section 316(a), PSEG elected to update its biothermal assessment to reflect significant improvements in water quality and the BIP of the Hackensack River since the original assessment. In 1994, NJDEP issued a NJPDES Permit (NJ 00005622) for the Salem Generating Station (Salem), requiring a new Section 316(a) demonstration, having a level of detail that would be required for a new facility.

Figure 12-1
Location of Hudson and Salem Generating Stations

PSEG's 316(a) studies include seasonal instream thermal monitoring surveys, numerical modeling to characterize the thermal plume for representative seasonal and hydrodynamic conditions, biological monitoring, and a biothermal assessment based on the resultant thermal plume characterizations. These four components are designed and integrated by a team of biologists, thermal-hydrodynamic modeling experts, and station personnel to ensure that the final assessment fully addresses the potential impacts of the proposed thermal effluent limits on the BIP. The primary objectives of instream thermal monitoring is to obtain data for calibrating and verifying a time-varying three-dimensional hydrothermal model that is used to characterize the extent and distribution of temperatures within thermal plume for a range of representative hydrodynamic and seasonal conditions. Hydrothermal modeling is needed because an adequate number of monitoring devices cannot be deployed to accurately characterize the full spatial extent of temperatures within the thermal plume. The objective of the biothermal monitoring is collect data on the seasonal variation in the populations and life stages of a subset of indigenous species that are representative of the BIP (i.e., the representative indigenous species). The calibrated and verified hydrothermal model is used to compute acute and chronic thermal exposures that are needed to assess lethality, growth inhibition, migration, exclusion, blockage, and other confounding effects (i.e. D.O. depletion and contaminant toxicity) on a seasonal basis. The biothermal assessment integrates the results from the biomonitoring and thermal characterization to determine if the proposed alternative thermal limits are protective of the BIP.

316(a) Methodologies: Thermal Monitoring and Modeling

A comprehensive understanding of the thermal exposures at the relevant temporal and spatial scales can be obtained from thermal monitoring surveys and hydrothermal modeling studies. The primary objective of the thermal monitoring survey is to obtain the necessary hydrodynamic and water quality data that are required for calibrating and verifying one or more hydrothermal models. The calibrated and verified hydrothermal models are then used to calculate within plume temperatures or incremental temperature increases due to the thermal discharge (i.e. the "excess temperature" or "ΔT"). The following sections summarize the approaches that PSEG used to conduct the surveys and studies for the Hudson and Salem 316(a) studies.

Thermal Monitoring

A thermal monitoring program should provide all the necessary information on the hydrodynamics of the receiving waterbody, the spatial and temporal distribution of temperatures and other water quality parameters (i.e. conductivity and dissolved oxygen) within the thermal plume, and meteorological conditions during the monitoring program. For estuarine systems, the hydrodynamic data include tidal elevations, current speeds and directions, and mapping of local bathymetry. Water temperature, conductivity and dissolved oxygen are measured throughout the region of the estuary occupied by the thermal plume, at points where the excess temperature is negligible (i.e. $\Delta T \approx 0°F$), and at points bounding the region of the estuary that will be included in the hydrothermal model. On-site or local representative meteorological data are collected to compute surface heat exchange and wind induced stresses on the water.

The design of thermal monitoring program is based on site-specific factors including the outfall design, the bathymetry of the waterbody, and the magnitudes of the discharge and ambient flow and temperatures. For instance, Hudson discharges waste heat to the Hackensack River through a discharge canal that terminates on a sharp bend in the river. In contrast, Salem discharges heat through large submerged pipes in a region of the estuary where current velocities are non-uniform and where tributaries that drain extensive marshes influence water temperatures.
As shown below, the thermal plumes created by the two facilities have generally similar characteristics but different spatial scales. Temperatures decrease rapidly in the immediate vicinity of the outfall where the momentum of the discharge induces mixing, and then decrease with distance at much reduced rates where passive mixing is controlled by ambient conditions.

The Hudson and Salem thermal monitoring programs included three common components. One was a bathymetric survey of the estuary extending several miles from the outfalls. The other two components were complimentary and were designed to provide data on temperatures within the thermal plume. One of these was a network of in-situ moorings equipped to measure various water quality parameters (namely, temperature, dissolved oxygen, and conductivity), water levels, and currents at frequent intervals (5 to 10 minutes) over a period of months. The data provided by the moorings were used primarily to characterize the long-term (i.e. seasonal) and short-term (intra-tidal) variability of temperatures with the plume, and secondarily to provide some information on spatial variability. The spatial resolution of measurement points was greatest in the immediate vicinity of the outfall where discernible changes in temperature with distance were expected. The other component was a series of ship-board surveys that measured vertical profiles of the water quality parameters at a dense set of discrete points in the plume over short periods (1 to 1.5 hours) at four phases of the tide (low water slack, maximum flood, high water slack, and maximum ebb). Data provided by the shipboard surveys were used primarily to characterize the spatial variability of temperatures with the plume, and to a lesser extent temporal variability. The monitoring at Hudson was conducted by Ocean Surveys, Inc. and at Salem by a team consisting of Eric Adams, Ph.D. (Massachusetts Institute of Technology), The Woods Hole Group and Lawler Matusky and Skelly Engineers.

Each monitoring program survey also included separate surveys related to site-specific conditions or issues. In the case of Hudson, a program to collect water quality and sediment samples for determining biological and sediment oxygen demands was performed to obtain data needed for a more accurate assessment of the potential affects of temperature on dissolved oxygen. In addition, dye studies to determine flow rates from the circulating water pumps were performed to compute the actual amount of heat being discharged to the river. Details of the Hudson thermal monitoring program are provided in PSEG's 316(a) Demonstration for Hudson Generating Station [2].

The Salem thermal monitoring program included a near-field dye-tracer and aerial infrared survey, an ambient temperature survey, marsh surveys, and an "initial condition" survey. Details of these special studies are provided in the following paragraphs and in PSEG's 316(a) Demonstration for Salem Generating Station [3].

The momentum from Salem's large submerged high-velocity discharges creates a highly turbulent region that extends several hundred feet from the outfall. Deploying and anchoring moorings in this area is impractical. Yet, because of its size and the existence of elevated ΔTs, data for this area of the thermal plume are needed to adequately assess the potential for adverse acute thermal exposures. Dye-tracer and infrared aerial surveys were used to obtain the necessary data. The dye-tracer study used shipboard surveys with fluorometers to track the thermal plume and to measure rates of dilution. The infrared aerial surveys captured synoptic changes in the rate of dilution with distance by taking photographs that showed the relative difference in water surface temperature at four phases within the tide cycle.

Natural heating and cooling of a large waterbody, such as the Delaware Estuary, results in non-uniform temperatures which may vary by several degrees (°F) across the estuary or over the water column. Thus, no single water temperature defines "ambient". Because the thermal plume from Salem is transported large distances, an ambient temperature study using shipboard surveys and lasting several days was performed to understand the natural spatial variability in the near- and far-field regions of the plume. In addition to the shipboard surveys, moorings were deployed in regions outside the thermal plume for several months to obtain data on the seasonal variation in ambient temperatures.

The marsh surveys were included to address thermal plumes from large tributaries extending into extensive marshlands of the Estuary. Natural heating and cooling processes significantly alter the temperature of the water entering and leaving the marshes. Data from the marsh survey were used to estimate the contribution (either loss or gain) of heat to the Estuary from the marshes.

Finally, an initial condition survey was performed to obtain temperature and salinity distributions along and across the main-stem of the estuary prior to the shipboard surveys and deployment of in-situ moorings. For large estuarine systems, such as Delaware Estuary, an incorrect initial condition may affect predictions by the hydrothermal model well into the simulation period. The results from these studies were used to set the initial conditions to facilitate the calibration and verification of the hydrodynamic model.

Modeling

Practical considerations and costs prohibit deploying a monitoring program that provides all the data needed to fully characterize a thermal plume (i.e. the time-varying distributions of temperatures within the plume). Properly calibrated and verified state-of-the-art hydrothermal models provide missing information or characteristics that cannot be measured (i.e. ΔTs), and are useful tools for assessing combinations of alternative effluent limitations and estuarine conditions (tidal regimes, freshwater flows, etc.) on plume characteristics. The choice of model(s) depends on the required level of temporal and spatial resolution needed for assessing the impacts of thermal exposures on the BIP, the user's familiarity with the model, computational resources, unique features of the thermal plume (such as mixing zones) or estuary, or prior use and acceptance by a regulatory agency. For large once-through cooling water discharges to estuarine waters, time-varying three-dimensional hydrothermal models offer an efficient means of obtaining comprehensive information on the thermal plume.

PSEG's approach to estimating the temporal (intra-tidal and seasonal) and spatial distribution of temperatures and ΔTs is shown graphically in Figure 12-2 for Salem. Numerical models were used to calculate the spatial distribution in ΔTs with distance from the outfall (the upper left panel), and the time varying exposure experienced by a particle drifting through the thermal plume (the upper right panel). A similar approach was used for Hudson. The (total) temperature within the thermal plume under average or warm ambient conditions was estimated by adding the intra-tidal variations in ΔT to estimates of ambient temperature (shown as the upper and lower curves, respectively, in the lower panel of Figure 12-2) for various recurrence intervals (i.e. one year in two, and one year ten).

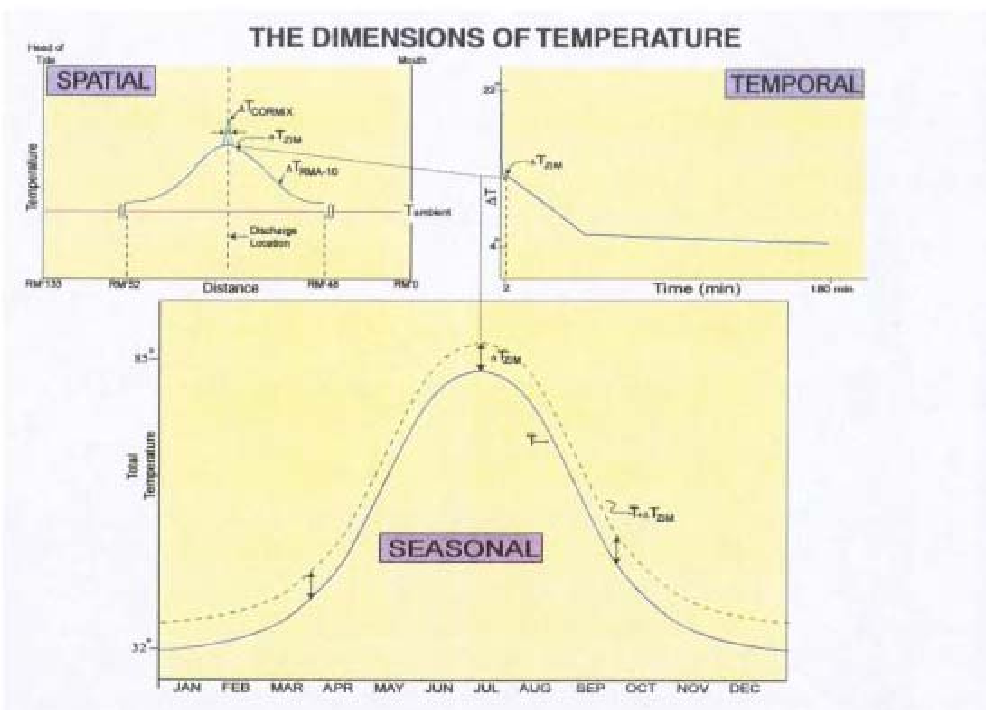

Figure 12-2
Dimensions of the Salem Total Temperature Model

Seasonal variations in ambient water temperature for the Hackensack River (Hudson Station) were calculated using a "response temperature" model developed by J. E. Edinger Associates (Edinger). The response temperature is the temperature a column of water of specified depth would have if surface heat exchange were the only active heat transfer process. Edinger calibrated the response temperature model for the Hackensack River. LMS calibrated the response temperature model for Salem and modified it to include a component that accounted for the influence of the Atlantic Ocean on water temperatures at Salem. In both cases, the calibrated ambient temperature models were used with a long-term (20^+ year) record of meteorological conditions to compute daily and seven-day average ambient temperatures as input to the biothermal assessment.

The intra-tidal spatially varying distributions of ΔT, which define the thermal plume, were computed using time-varying three-dimensional hydrothermal models. For Hudson, Edinger adapted the Generalized Longitudinal, Lateral, Vertical Hydrodynamic Transport (GLLVHT) Model to the Hackensack River and Hudson [4]. The model was calibrated and verified using independent (seasonal) data sets provided by the thermal modeling program. LMS and Woods Hole Group calibrated and verified RMA-10, with an externally linked steady state near-field model (CORMIX), using independent data sets when Salem was operating with one and two units[5, 6]. The GLLVHT and RMA-10 models were selected because they allow irregular computational grids that can be shaped to the bathymetry and thermal plume, allow for unique features of the estuary (such as the wetting and drying of marshes) and simulate the withdrawal of cooling water from the estuary and the discharge of heated water. CORMIX was linked to the RMA-10 model because far-field models, like GLLVHT and RMA-10, cannot simulate the complex mixing processed induced by a submerged high-rate diffuser.

Models like GLLVHT and RMA-10 have their own advantages and disadvantages. RMA-10 is a finite element model that provides considerable flexibility in configuring the computation grid so as to obtain the optimum spatial resolution for characterizing different areas of the thermal plume. In addition, it has been accepted by regulatory agencies for other applications in the Delaware Estuary. RMA-10's primary disadvantages are its complexity and demands on computation resources. Calculations of ΔT using RMA-10 (Version 6.6) require two simulations and subsequent processing of the output data files. Separate simulations are made with and without the thermal discharge. Post-processing to determine the thermal plume requires subtracting the output without the thermal discharge from the output with the thermal discharge for each computational point and time-step. GLLVHT has the distinct advantage of calculating ΔT directly, and was constructed using empirical information on various aspects of estuarine hydrodynamics and mixing. The latter feature simplifies the calibration and verification processes since the adaptation of the model to the estuary is based more on bathymetry than adjustments to empirical coefficients. At the time of its application, GLLVHT was a proprietary model that had not been widely applied in New Jersey, and post-processors for obtaining characteristics of thermal plume were being developed.

A properly calibrated and verified suite of hydrothermal models may be used to investigate the impact of the thermal discharge on the BIP for various combinations of critical hydrodynamic and meteorological conditions and proposed alternate thermal effluent limits. Figure 12-3 shows the integrated application of the output from the ambient temperature model (ATM) and the thermal plume (RMA-10 and CORMIX) to predict the seasonal variation in temperatures within the Salem thermal plume and their frequency of occurrence, and to define the spatial distributions and short-term exposures (associated with intra-tidal variations) of ΔT within the thermal plume. A similar integrated approach was used for Hudson. However, as noted above, the Hudson study did not required a separate near-field model like CORMIX to simulate mixing in the near-field.

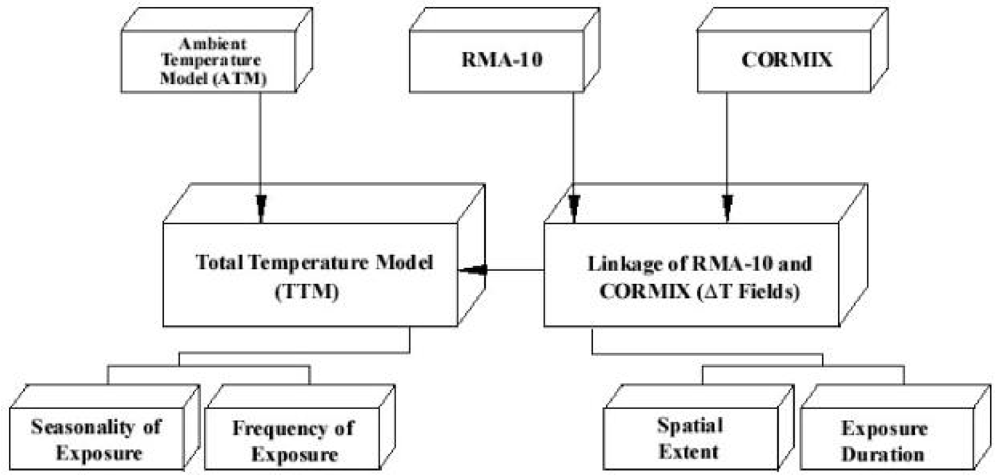

Figure 12-3
Components of the Salem Thermal Modeling Program

The addition of results from the ATM and the far-field yields the seasonal variation in total temperature ($T_{ambient} + \Delta T$), and their likely frequency of occurrence. Similarly, specific information of the characteristics of the thermal plume that are required input to the biothermal assessment are calculated by processing output files of ΔT. For example, spatial distributions of ΔT across various cross sections along the river provide information on blockage to migration. Spatial distributions of ΔT over regions (volumes) of the river allow assessing the potential for exclusion. Spatial distributions of ΔT over the bottom of the river and along the shoreline show the benthic community's exposure to the thermal plume. Time-varying exposures to ΔT by a drifting particle define the acute exposure and are used to address the potential for lethality. Finally, the model is used to determine the potential for cold-shock in the event that the thermal discharge from the station was abruptly terminated, and the instream spatial distribution of the change in DO resulting from the thermal discharge.

316(a) Results: Thermal Monitoring and Modeling

The calibrated and verified models for Hudson and Salem were used to provide various inputs to the biothermal assessment. These inputs include: 1) the spatial and temporal variations of ΔT as a function of the characteristics of the cooling water discharge, the hydrodynamics of the waterbody and the meteorological conditions; and 2) the time series of ΔT along paths that drifting particles, such as ichthyoplankton (fish eggs and larvae) might follow if they were released at the point of discharge of the thermal plume. In addition, calculated daily and weekly averages of ambient temperatures and their frequency of occurrence were estimated. Examples of the results are provided below showing the areal extent of the surface isotherms (Figures 12-4 and 12-5) and a table of the surface area within each isopleth (Table 12-1).

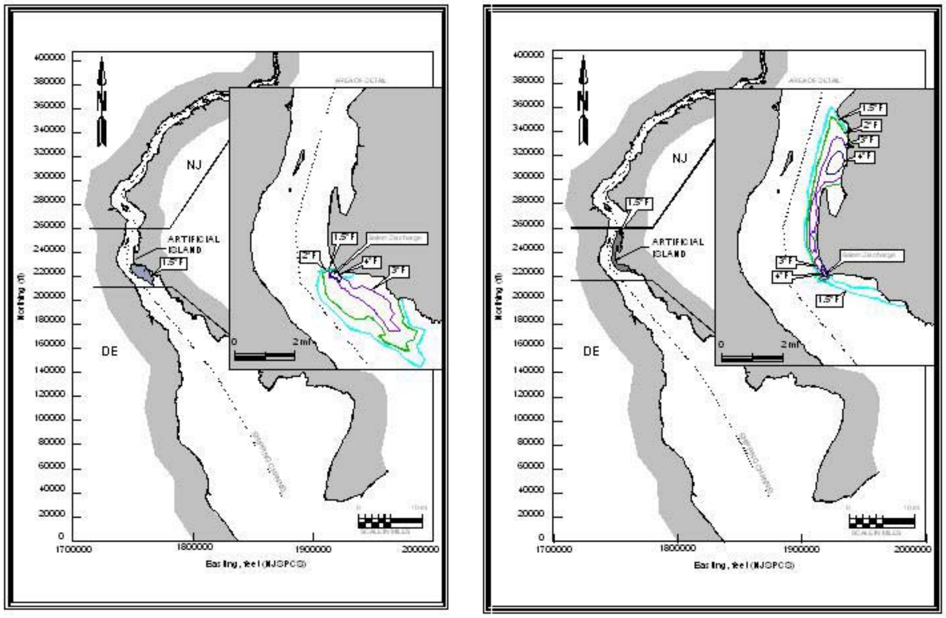

Figure 12-4
RMA 10 Results- Surface ΔT Isotherms for Salem's Longest Plumes (End-of-Ebb 6/2/98 on Left and End-of-Flood 5/31/98 on Right)

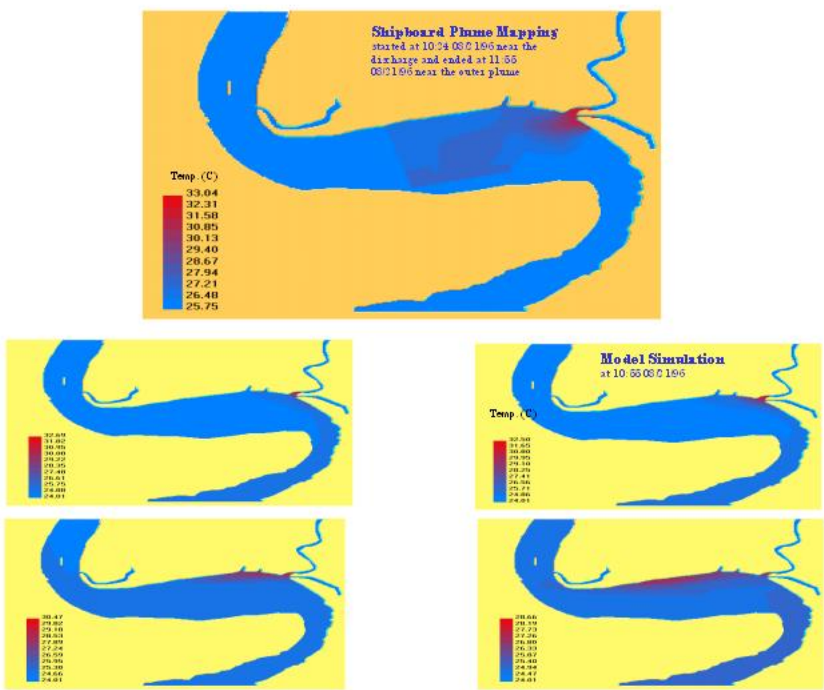

Figure 12-5
Results of GLLVHT Model Compared to Shipboard Plume Mapping for Hudson Generating Station

Table 12-1
Cumulative Surface Area within each ΔT Isopleth of the Salem Thermal Plume

	Ebb: 6/2/1998 at 0830 hrs		End of Ebb: 6/2/1998 at 0000 hrs		Flood: 6/4/1998 at 1630 hrs		End of Flood: 5/31/1998 at 1600 hrs	
ΔT (°F)	Surface Area (acres)	Percent of Estuary Area	Surface Area (acres)	Percent of Estuary Area	Surface Area (acres)	Percent of Estuary Area	Surface Area (acres)	Percent of Estuary Area
>13	0.08	0.00002	0.00	0.00000	0.00	0.00000	0.00	0.00000
>12	0.46	0.00010	0.47	0.00010	0.21	0.00004	0.00	0.00000
>11	0.98	0.00020	2.15	0.00045	0.61	0.00013	0.00	0.00000
>10	1.66	0.00034	2.15	0.00045	1.15	0.00024	0.85	0.00018
>9	2.22	0.00046	2.15	0.00045	1.82	0.00038	1.93	0.00040
>8	3.19	0.00066	2.15	0.00045	2.64	0.00055	1.93	0.00040
>7	4.32	0.00090	5.10	0.00106	3.59	0.00075	1.93	0.00040
>6	5.61	0.00116	11.32	0.00235	4.68	0.00097	1.93	0.00040
>5	36.60	0.00760	21.43	0.00445	56.58	0.01174	2.14	0.00044
>4	150.08	0.03115	45.11	0.00936	245.94	0.05105	205.37	0.04263
>3	631.42	0.13106	739.88	0.15357	585.78	0.12158	920.75	0.19111
>2	1947.91	0.40430	2519.94	0.52303	2212.75	0.45927	2093.04	0.43442
>1.5	3156.56	0.65517	3725.19	0.77319	3703.61	0.76871	3596.95	0.74657

316(a) Methodologies: Biothermal Assessment

The biothermal assessment for Hudson was performed by Beak Consultants, Inc. and the Salem biothermal assessment by EA Science and Technology (1993 assessment) and ASA Communications (1999 assessment). All of the biothermal assessments were performed under the guidance of Dr. Charles Coutant and coordinated by Dr. James Mudge (Civil and Environmental Consultants, Inc.). The general approach used in the biothermal assessment is contained in Figure 12-6.

Using the alternative Type 3 Demonstration assessment methods recommended by USEPA, the biothermal assessment provided both predictive and retrospective evaluations of the potential biological effects of each Station's thermal discharge on the BIP [7, 8, 9]. In addition to the USEPA draft guidance documents, professional practice in prior Section 316(a) assessments at Hudson, Salem and other generating stations; and Guidelines for Ecological Risk Assessment (ERA Guidance) recommending approaches and criteria for assessing impacts from chemical, physical, or biological stressors were also considered[10]. The latter guidance is not specific to biothermal impact assessments, but was used to verify that the design of this assessment and the criteria used for assessing the potential adversity of effects are consistent with current regulatory advice and scientific practice.

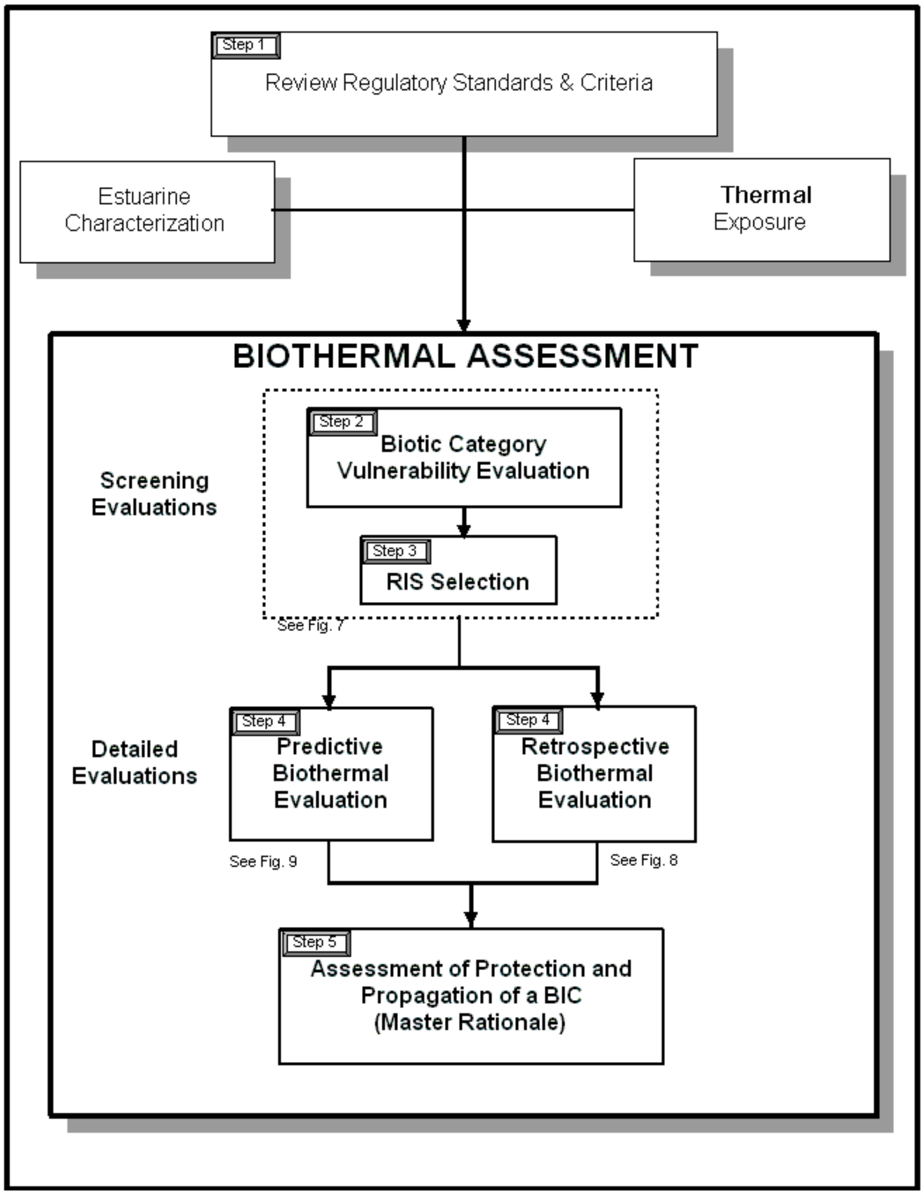

Figure 12-6
Steps in the Biothermal Assessment Process for a 316(a) Demonstration

Four methods of evaluation were used for the biothermal assessment, consistent with draft technical guidance for preparation of Section 316(a) demonstrations. These include two screening methods (Critical Function Zone (CFZ) and Biotic Category (BC) assessments), and two detailed methods (Predictive/Representative Important Species (RIS) and Retrospective/No Prior Appreciable Harm (NPAH) assessments) (Figure 12-6).

The second step in the biothermal assessment process evaluates the potential vulnerability of the BIP and its component biotic categories. Figure 12-7 illustrates the process for Salem’s thermal plume. For purposes of the biothermal assessment, “vulnerability” means either the potential for exposure to the plume and/or level of resistance to impacts from exposure. This screening step identifies the attributes of the thermal discharge design and location that reduce the potential for thermal impacts on the BIP. It also evaluates the relative potential for thermal discharge impacts on the biotic categories, based on the habitat zones they occupy, the importance of their role in ecosystem energy dynamics, or their life-history characteristics. Two screening methods were used in this step of the assessment to evaluate the potential for impact, referred to herein as the Critical Function Zone (CFZ) and Biotic Category (BC) methods (Figure 12-7). Assessment criteria used for both methods were those suggested by the Draft 316(a) Guidance. The vulnerability evaluation screens out those biotic categories that have low potential for impacts from a station's thermal plume (LPI categories), and focuses the detailed predictive RIS assessment and retrospective NPAH assessment on the remaining biotic categories.

The Draft 316(a) Guidance recognizes that it is impractical to study and assess in great detail every species at a site, and it is therefore necessary to select a smaller group to be representative of the balanced indigenous community. These selected species are designated as the RIS. Generally 5–15 RIS are chosen to represent biotic categories that are not classified as LPI. According to the USEPA Draft 316(a) Guidance, RIS are to include species that are:

- Commercially and recreationally valuable;
- Threatened or endangered;
- Critical to the structure and function of the ecosystem (e.g. habitat formers);
- Potentially capable of becoming localized nuisance species; and
- Necessary in the food chain for the well being of species determined above.

Other considerations for RIS selection include the extent of the species involvement with the thermal plume, the species thermal sensitivity, and the quantity and quality of information available for the assessment. Using these criteria, RIS were selected for the biothermal assessment for each Station.

Predictive Biothermal Assessment

For the predictive portion of the biothermal assessment, Draft 316(a) Guidance recommended that the community of organisms that become involved with the thermal plume be divided into several biotic categories. The predictive and retrospective assessments of protection of the balanced indigenous community addressed each of the USEPA-recommended biotic categories, which are:

- phytoplankton;
- habitat formers;
- zooplankton and meroplankton;
- shellfish/macroinvertebrates;
- fish; and
- other vertebrate wildlife.

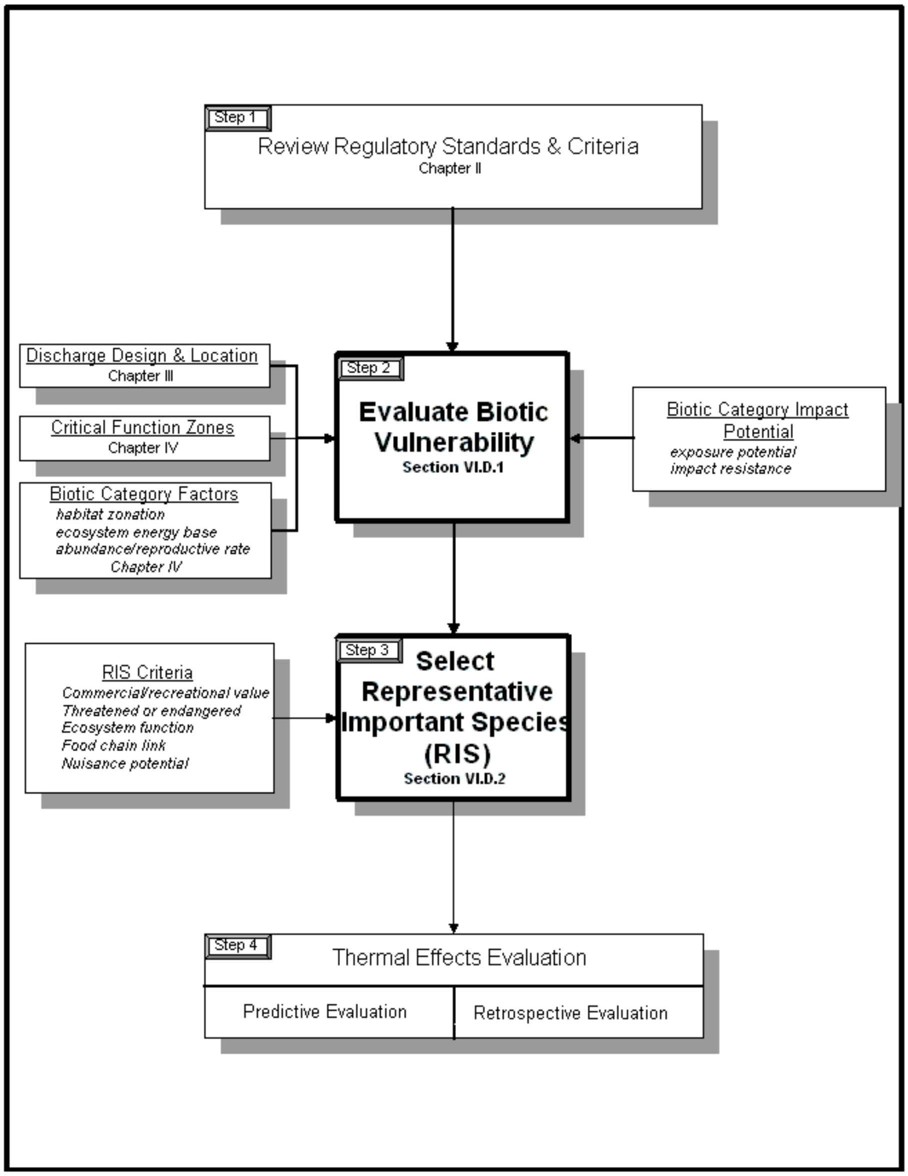

Figure 12-7
Biotic Category Evaluation and Representative Important Species Selection for a Biothermal Assessment

For some individual biotic components of the balanced indigenous community, the area evaluated is smaller or larger than the receiving water body defined. This is appropriate since the species and biotic categories comprising the overall BIP are not uniformly distributed in space and time. The area selected for analysis of biothermal effects therefore also depended on the life history and distribution of the species and community components involved.

The potential for impact was evaluated by first predicting the nature and likelihood of potential thermal effects on individual organisms, and then assessing the significance of those effects on the RIS populations. In the language of USEPA Draft Section 316(a) Guidance, the significance of effects equates to their potential for causing "Appreciable Harm" (Figure 12-8). The nature and likelihood of thermal effects was characterized by comparing the habitat preferences, seasonal occurrence, and temperature requirements or limits of each species to reasonable worst-case thermal plume conditions that could potentially occur as a result of the station's operation.

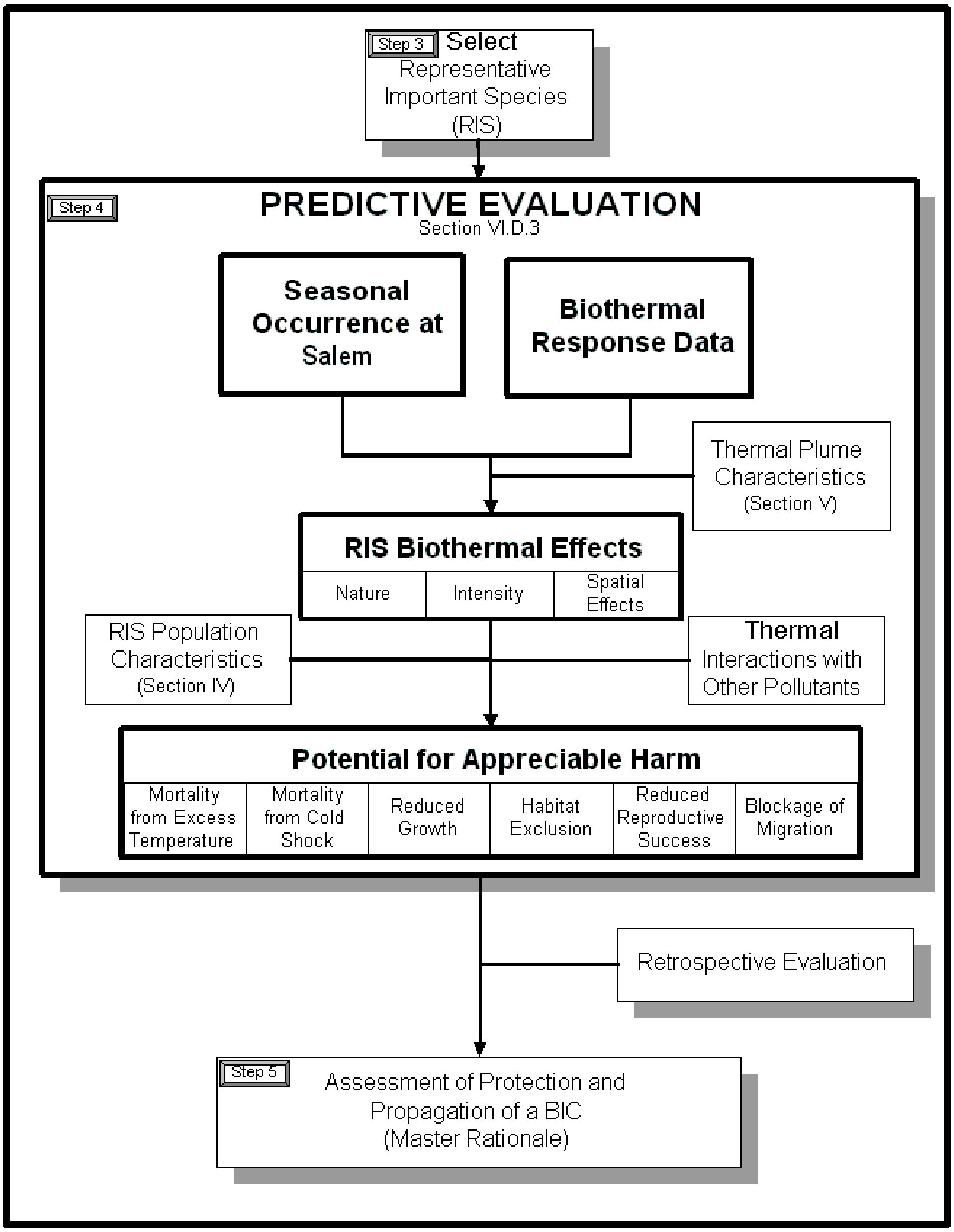

Figure 12-8
Predictive Evaluation Steps in a Biothermal Assessment

An important consideration is that the detailed predictive RIS Biothermal assessment is performed at three levels of protective conservatism. First, excess temperatures (ΔT s) used to characterize exposure of the RIS were based on a "reasonable worst-case" thermal plume. This plume was modeled based on full generating load and low cooling water flow conditions, which result in a maximum ΔT above ambient estuary temperature at the point of discharge in the

Estuary. In addition, the plume was modeled based on hydrological and meteorological conditions that result in higher near-field plume temperatures than would occur during most times of the year.

Second, the total plume temperatures to which the RIS may be exposed at various times of year were characterized based on both "warm" or "cool" (e.g., 1-in-10-year recurrence) and "average" (e.g., 1-in-2-year recurrence) ambient water temperatures in the vicinity of the Stations.

Third, the likelihood of thermal effects on individuals of the RIS exposed to the plume was assessed using highly conservative assumptions about the location and duration of their residence in the thermal plume. Effects on organisms drifting through the plume were initially evaluated based on the highest (centerline) ΔT that could be experienced, beginning from the point of discharge. Chronic effects on more mobile life stages of RIS were initially evaluated based on the highly unlikely case that they could and would choose to maintain position in the highest ΔT fields near the edge of the zone of initial mixing (ZIM). Effects, if any, that were evident from these highly conservative initial evaluations were then examined in more detail to determine their potential for causing appreciable harm.

Water temperatures at a given location in these estuaries vary daily by about 2°F to 5°F and seasonally by about 45°F to 50°F. The dimensions and location of the Station's thermal plume, which are overlaid on these daily and seasonal fluctuations in ambient temperature, change dynamically in response to tidal and meteorological conditions. Except for that very small area referred to as the zone of initial mixing ("ZIM") in the immediate vicinity of the discharge, a given temperature substantially above ambient is unlikely to occupy any specific location for more than a brief and intermittent period during the daily tide cycles. Field measurements and instantaneous model simulations represent momentary, ephemeral snapshots of the physical configuration of this dynamic plume. Since fish and other organisms exhibit a seasonal range of temperatures that they prefer and utilize if available, it is unlikely that organisms would actively follow portions of the plume with temperatures outside their preferred range as the plume shifts with the tide and wind. In addition, the likelihood of all of these three conservative conditions occurring at the same time is extremely low.

The potential effects of the thermal discharge were evaluated for the following five biothermal response categories as recommended in the Draft 316(a) Guidance:

1. thermal shock tolerance (heat and cold) of juveniles and adults;
2. upper temperature tolerance for short-term exposure of planktonic forms of the RIS
3. upper avoidance temperature;
4. temperature requirements for performance and growth; and
5. temperature requirements for spawning and early development.

The potential for the Station's plume to elicit these biothermal responses was analyzed graphically by comparing seasonal occurrence of the RIS in the vicinity of the Stations and biothermal response data for these species obtained in laboratory studies to the predicted seasonal ambient and maximum plume temperatures to which the organisms may be exposed (see results Section below).

The graphical analyses screen for potential effects by relating the occurrence of each life stage in the Station vicinity to potential thermal effects produced by contact of that life stage with the highest accessible plume temperatures. They were used to identify the nature and seasonal timing of the thermal effects expected for each RIS. They clearly identify the conditions that would be unable to elicit particular biological responses (e.g., mortality). For each condition under which a thermal effect is predicted, estimates were made of maximum cross-sectional areas, surface areas, and volumes of the thermal plume in which important biological activities would potentially be limited by increased temperatures. Thus the thermal effects evaluation yielded predictions of the likelihood of thermal effects being caused by the Station's thermal plume, and the nature, spatial extent, and temporal pattern of such effects (Figure 12-8 above).

Under this evaluation, existing empirical information was carefully analyzed to determine whether there is any evidence that the Station's thermal plume has caused appreciable harm to the biological communities over the period of Station operations.

This evaluation was conducted in two parts. First, there was consideration of each biotic category, other than those considered LPI, because of low or no exposure to the thermal plume, i.e. habitat formers and other vertebrate wildlife. This part considered community-level factors such as species composition, structure and overall abundance to reach conclusions about current conditions for each biotic category as a whole compared to that expected had the Stations not operated.

Retrospective Biothermal Assessment

The retrospective assessment considered the condition of the population of each RIS in the Estuary. This focused on long-term trends in abundance of RIS within the Estuary. For most species, these abundance estimates are for the juvenile stages that can be used as an index of annual production of young from spawning and nursery habitats within the Estuary. One of the first signs of a continuing decline in population abundance is a downward trend in recruitment (i.e. young fish produced each year). Therefore, examination of this stage provides both a reliable measure of potential thermal effects on the early life stages of each species as well as an indication of potential effects on the adult stock (Figure 12-9).

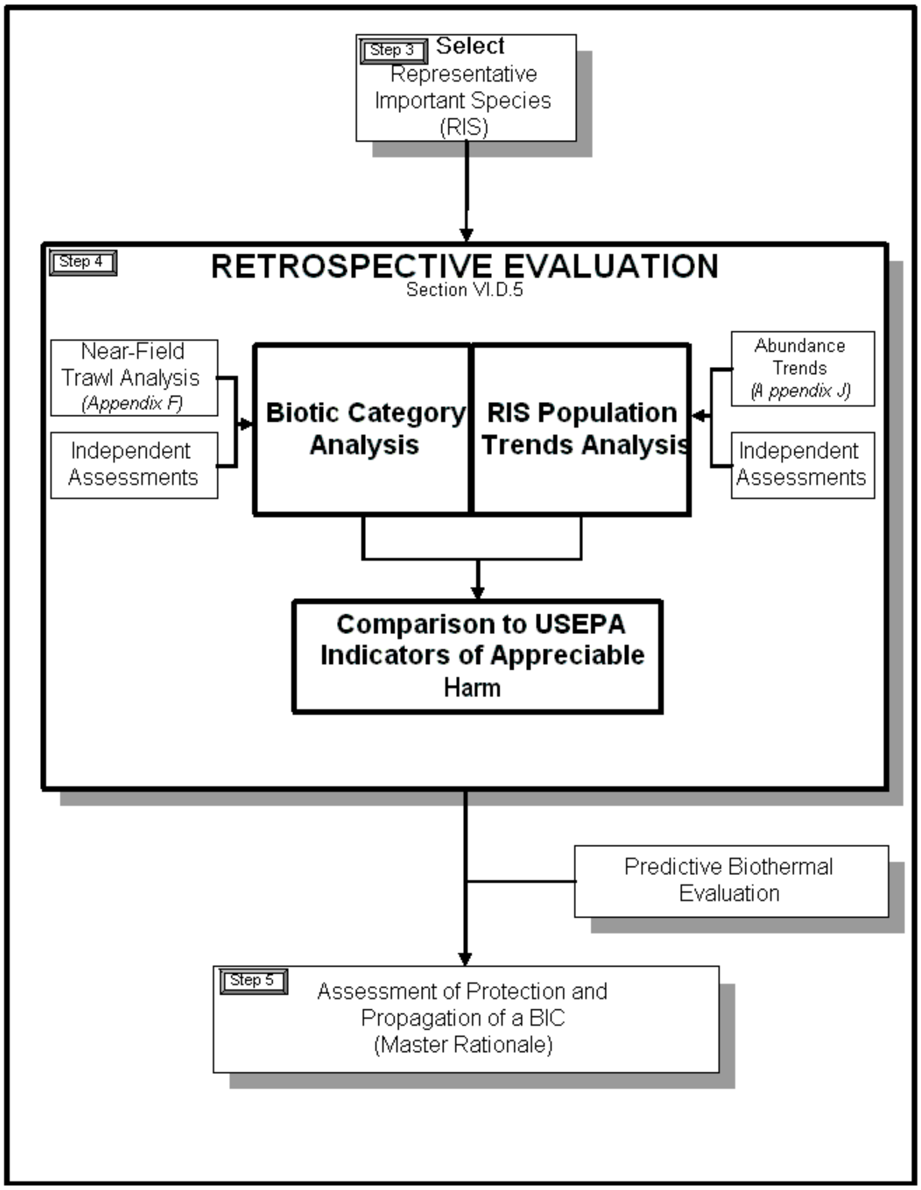

Figure 12-9
Retrospective Evaluation Steps in the Biothermal Assessment

The results of both the biotic category and RIS population level retrospective evaluations were then compared to phenomena identified by USEPA as evidence of appreciable harm to biological communities [9]. The intensity and magnitude of effects observed through this retrospective evaluation was then assessed in light of the potential for the Station's thermal plume to induce such effects.

The potential for the Station's thermal plume to have an appreciable impact on the populations of the Estuary RIS was assessed by evaluating the likelihood, nature, and spatial scale of thermal effects predictions in the context of:

- species life-cycle requirements and characteristics;
- species ranges and distributions;
- dimensions of available habitat in the Estuary and population resilience; and
- potential for reversal of effects.

These evaluations specifically addressed the criteria presented in the Draft 316(a) Guidance dealing with issues of survival, growth, reproduction, and habitat exclusion.

The retrospective assessment included additional documentation to support PSEG's application for a Section 316(a) variance, including data on the abundance of RIS fish in the Estuary that, in the case of Salem, were collected from 1966 to 1990 by environmental and resource management agencies, the University of Delaware, and PSEG[11]. Data from long-term studies of fish abundance in the Delaware Estuary for the period after Salem began operation were compared with the findings of studies conducted before operation, and were examined for trends over time.

At Hudson, there were no surveys and little available baseline information on the aquatic community of the Estuary in the vicinity of the Station prior to the commencement of operation of the station (1968 two unit operation). The retrospective assessment primarily relied on a comparison of the communities in the near-field and far-field regions as they currently exist. The Hudson retrospective assessment also evaluated changes in the community during operations in 1971-73, 1986-88 and 1996-1997. This temporal evaluation provides insight on the re-establishment of a BIP as water pollution control initiatives have improved water quality during the period that Hudson operated.

Information on the thermal plume's characteristics developed from the updated studies was integrated with life-history information and with the temperature requirements of the RIS. The integrated information was used to predict the potential of the thermal plume to cause appreciable harm to indigenous populations. The assessment also included an evaluation of the potential interactive effects of heat and other parameters (e.g., DO, chlorine, and toxic substances).

316(a) Results: Biothermal Assessment

Based on the extensive thermal monitoring and modeling and biothermal assessments for the Stations it was concluded that there was no evidence that the Station s' thermal plumes have caused or will cause an adverse impact to biotic populations and aquatic communities in the Estuaries. Using the USEPA Criteria, the Hackensack in the vicinity of Hudson was determined to be low potential impact site for four biotic categories: phytoplankton, zooplankton, habitat formers/macrophytes and other vertebrate wildlife.

Based on the previous studies conducted by PSEG in the Hackensack River in the vicinity of Hudson and the thermal modeling, PSEG concluded that the thermal discharge would not adversely affect shellfish/macroinvertebrates and fish because:

- The rapid mixing of the thermal plume near the point of discharge does not preclude significant habitat from use;
- A substantial barrier to migration or passage of aquatic organisms is not formed;
- Appreciable adverse impacts to reproduction, growth and survival of aquatic organisms will not be caused because to the warmest temperature occupy a small volume of the Estuary;
- There is little impact on plankton or less mobile, early lifestages (eggs and larvae) of macroinvertebrates and fish because they are able to tolerate the typical temperature regime (except under the rarest circumstances were low levels of mortality (<10-30%) of certain lifestages of certain species might occur near the end of the thermal discharge); and
- Cold shock would not cause an adverse impact.

The Delaware in the vicinity of Salem was determined to be low potential impact site for two biotic categories: phytoplankton and zooplankton. Based on the analysis of existing available scientific data for shellfish and macroinvertebrates in the Delaware Estuary in the vicinity of Salem indicates that this biotic category is similar in composition and overall abundance to that observed before Salem began operation. There is no evidence of any increase in the abundance of any stress-tolerant species since the Station began operation.

The analysis of fish species richness, density, and turnover in the Delaware Estuary clearly demonstrates that there has been no appreciable harm to the fish community that can be attributed to the operation of the Station. Figure 12-10 is an example of the results of one of these analyses.

Results of the trends analysis demonstrate that the operation of Salem, including its thermal discharge, has not caused appreciable harm to the populations of any of the ten RIS evaluated. In fact, statistical analysis of these trends reveals a significant increase in the abundance of seven of these RIS during the period of Station operations (See Figure 12-11 for an example for weakfish).

The predictive assessment concluded that Salem's thermal discharge does not threaten the protection and propagation of a balanced indigenous community of aquatic life because:

- only very small portions of the populations were being exposed to the higher-temperature regions of the plume for more than short (one to two minutes) periods of time (Figure 12-12);
- differential temperatures within the far-field portion of the plume are small (Figure 12-13); the volume encompassed by the instantaneous 1.5°F ΔT isopleth is small, relative to the extensive RIS habitat within the Estuary, and organisms are exposed to the far-field portion of the plume for relatively short periods of time;
- the potential for fish mortality due to cold shock is low, primarily because of the design and location of the discharge; and
- the plume has little potential for adverse population effects due to its mixing with other pollutants, and it does not significantly affect characteristics of the receiving waters such as the level of dissolved oxygen.

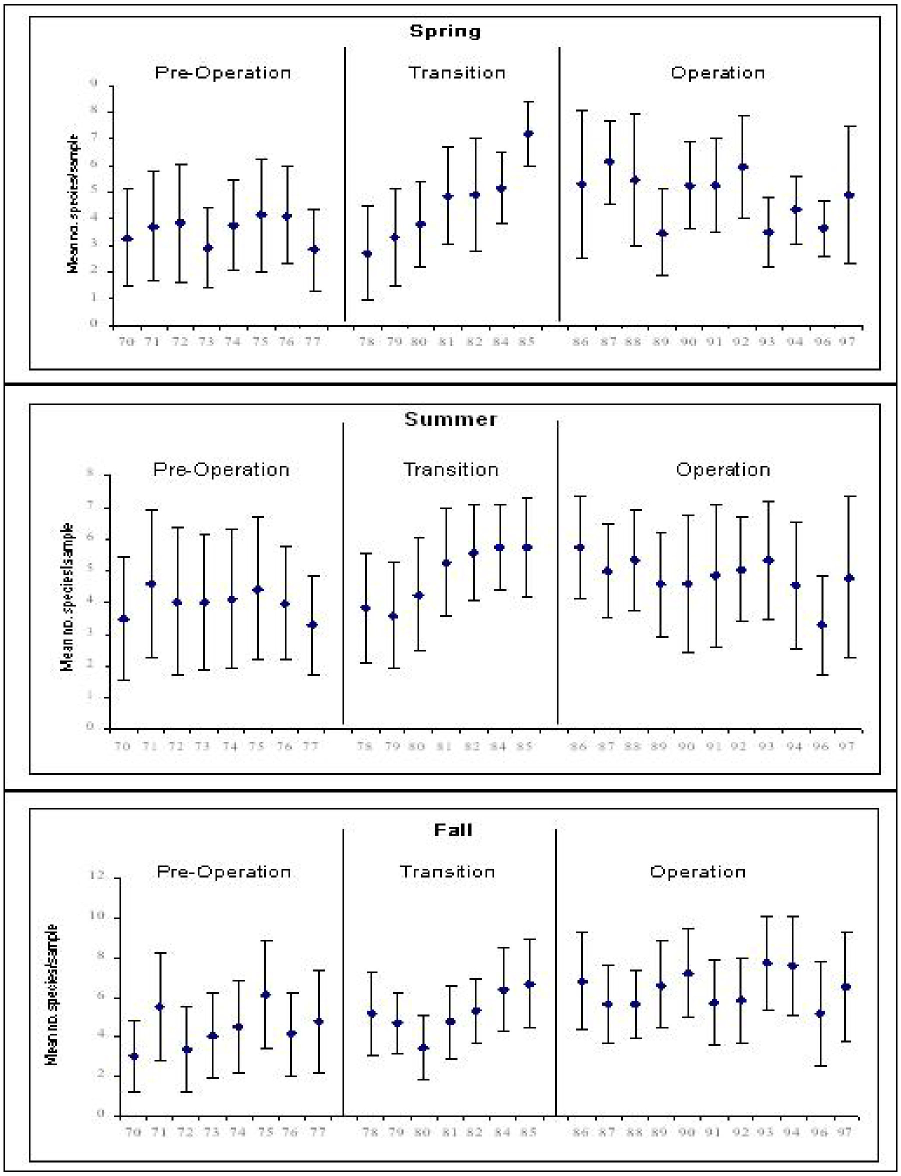

Figure 12-10
Long-Term Trends in Species Density of Fish Collected by Bottom Trawl in the Vicinity of Salem, 1970–1998

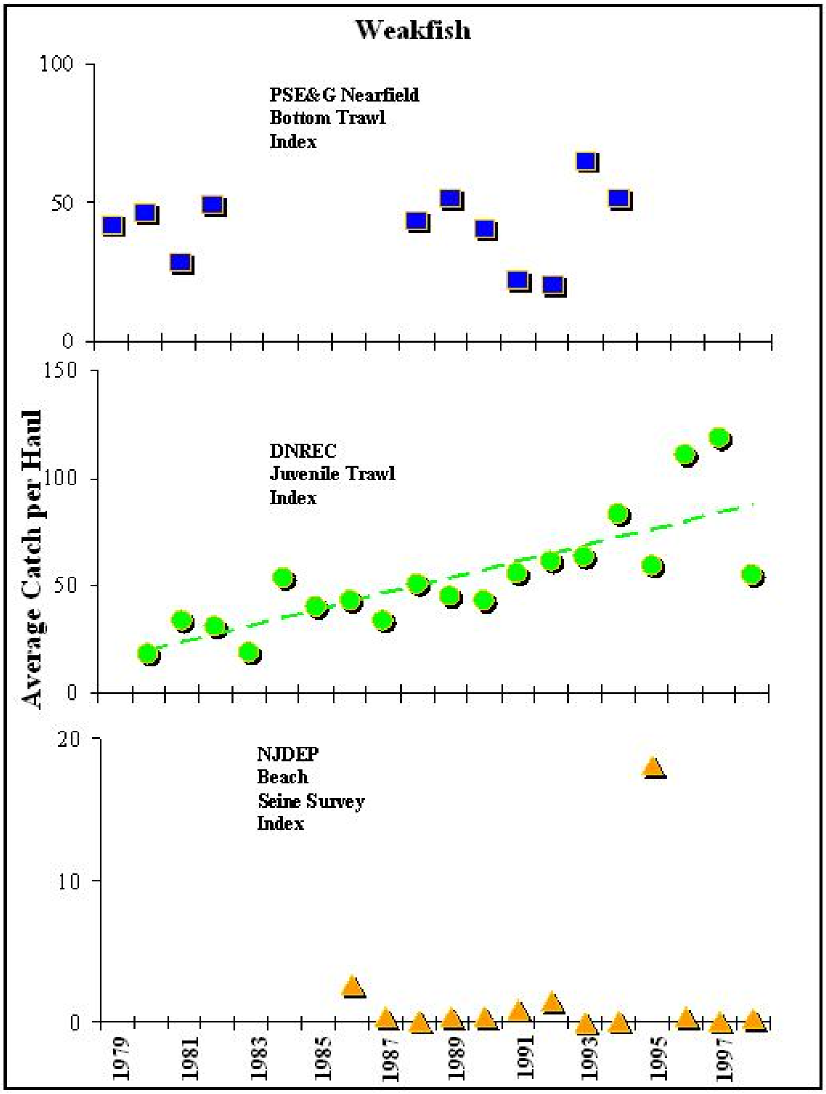

Figure 12-11
Long-Term Trends in the Abundance of Age-0 Weakfish in the Delaware Estuary Based on Three Independent Monitoring Programs, 1979-1998

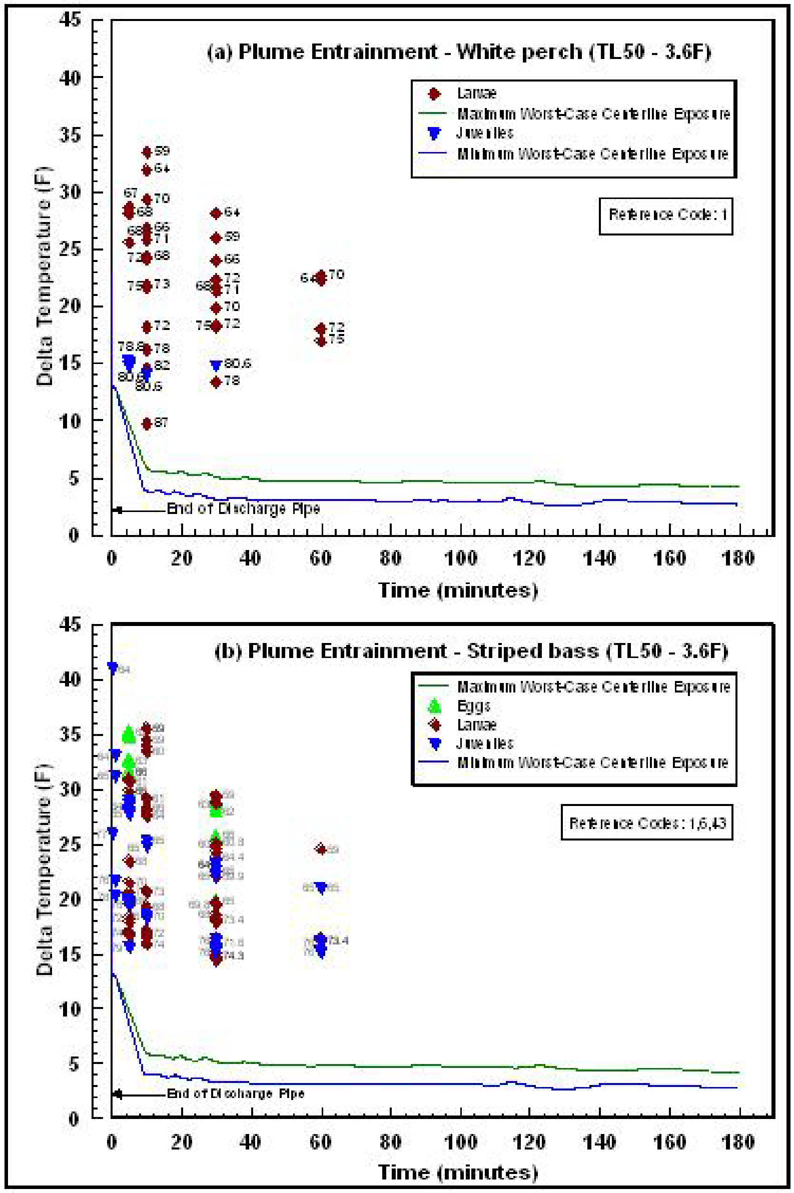

Figure 12-12
Short-Term Thermal Tolerance of Temperate Bass Relative to Temperature Decrease with Time along the Centerline of Salem's Thermal Plume. (Data Labels Indicate Test Acclimation Temperature)

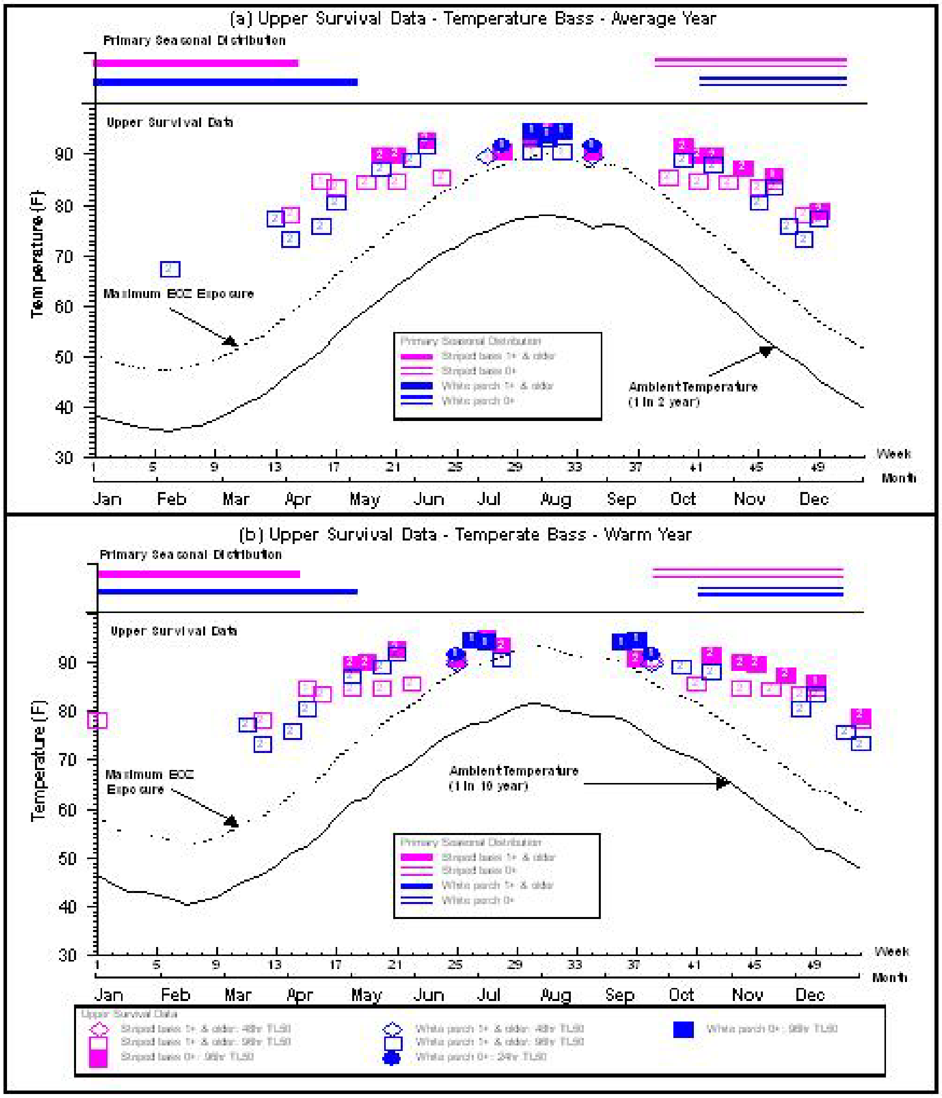

Figure 12-13
Upper Survival Data for Temperate Bass Relative to their Primary Seasonal Occurrence Near Salem, and to the Estimated Ambient Temperature and Maximum Plume Temperature at the Edge of the Zone of Initial Mixing (EOZ). (Reader Caution: Continuous Residence of the RIS at the EOZ Plume Temperature is Highly Unlikely for Various Reasons Discussed in the Text, and the Chart must be Interpreted in that Context.) (Data Labels Indicate Reference Codes)

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13

PROBABILITY–BASED IMPACT ASSESSMENT FOR A §316(A) DEMONSTRATION: AN EXAMPLE FROM ENTERGY NUCLEAR VERMONT YANKEE

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Introduction

On behalf of Entergy Nuclear Vermont Yankee, LLC, a Type III §316(a) Demonstration Report [1] was prepared by Normandeau Associates, Inc. in support of a pending request for a nominal increase in certain temperature limits during the summer period of May 16 through October 14 at the Vermont Yankee Nuclear Power Station (Vermont Yankee). The proposed new limits for the summer period change the existing NPDES permit limits only by adding one degree Fahrenheit to the calculated temperature rise that is presently allowed. No change is proposed during the summer period when ambient Connecticut River water temperatures are above 78°F or below 55°F. The request allows Vermont Yankee to improve power–generation efficiency by increasing operating flexibility, particularly during periods of reduced Connecticut River flow. Vermont Yankee discharges heated non–contact cooling water to the Connecticut River near Vernon, Vermont subject to and with the benefit of a NPDES Permit VT0000264, No. 3–1199, which was issued by the Vermont Agency of Natural Resources, Department of Environmental Conservation (VANR) on 29 August 2001 and expires on 31 March 2006.

Vermont Yankee's Demonstration Report detailed an assessment of the potential effects of the proposed nominal increase in the thermal–discharge limitation, as it relates to ten (10)nine species of fish identified as Representative Important Species (RIS): American shad (*Alosa pseudoharengus*), Atlantic salmon (*Salmo salar*), spottail shiner (*Notropis hudsonius*), white perch (*Morone americana*), smallmouth bass (*Microperus dolomieui),* yellow perch (*Perca flavescens*), walleye (*Sander vitrius*, formerly *Stizostedion vitreum*), largemouth bass (*Microperus salmoides*), fallfish (*Semotilus corporalis*) and white sucker (*Catostomus commersoni*). The first six of these RIS were included in the most recent previous §316(a) demonstration document [2]. These sixeven RIS represent the selected species based upon nearly more than 30 years of monitoring data and, therefore, allow an effective determination. Three (3) additional RIS, largemouth bass, fallfish, and white sucker, were added to the current Demonstration Report at the request of VANR. In addition to these ten (10) RIS, three (3) supplemental fish species were considered in this analysis at the suggestion of VANR: gizzard shad (*Dorosoma cepedianum*), American eel (*Anguilla rostrata*) and sea lamprey (*Petromyzon* *marinus*). Because no threatened or endangered aquatic species are known to exist in the Connecticut River near Vermont Yankee, none are included as RIS.

Vermont Yankee's Demonstration Report was based on an analysis of the hydrological and thermal regime in the Connecticut River during the summer periods (May 16– through October. 14) of 1998–2002. Hourly average flows and water temperature data were analyzed to establish recent river flow conditions, confirm that such conditions accord the historical period of record, and project how thermal conditions might change under the proposed new limits. Projections were made of the predicted increase in Connecticut River water temperature at the downstream compliance monitoring location under existing and proposed new permit limits. Analysis of the probability of occurrence of flow and temperature conditions was used to establish the average case (50% occurrence) and the extreme case (1% occurrence) reference conditions. A three–dimensional time varying hydrothermal model was used to predict the extent of Vermont Yankee's thermal plume in lower Vernon Pool under existing and proposed new summer thermal discharge limits for average and extreme case conditions of flow and temperature. Potential fish habitat changes due to the proposed new thermal regime were quantified based on volume and area in lower Vernon Pool predicted to be warmer or cooler than certain specified summer water temperatures that were derived from the thermal effects literature for the RIS and supplemental fish species.

The objective of this paper is to describe the predictive methods and interpretation used to perform the probability–based impact assessment for Vermont Yankee's §316(a) Demonstration Report. This paper is not intended to present a comprehensive overview of the complete Demonstration Report. As a Type III §316(a) Demonstration, retrospective analyses were used to establish baseline conditions, and predictive analyses described changes in the thermal regime under the proposed new discharge limits. The predicted new thermal regime was then interpreted with respect to thermal tolerance criteria for the RIS and supplemental species. Vermont Yankee's Demonstration Report also addressed other issues of concern by the regulatory agencies that will not be included in this paper, such as the predicted effects, if any, of the proposed new permit limits on fish passage upstream through the nearest downstream dam and fish ladder.

Site and Station Description

Vermont Yankee is located 0.75 miles upriver from Vernon Dam on a reach of the Connecticut River known as Vernon Pool (Figure 13-1). Vernon Pool extends from Vernon Dam upstream about 25 miles to the foot of the Bellows Falls Dam in Bellows Falls, VT and is comprisesd of 2,481 surface acres and 1.3814 billion cubic feet of water retained at a full–pond elevation of 220.13 ft behind the Vernon Dam and Hydroelectric Station. Vermont Yankee withdraws cooling water from the lowermost reach of Vernon Pool. All, a portion, or none of the cooling water may be returned to Vernon Pool as heated effluent, depending on the mode of operation. Under open cycle, Vermont Yankee is operated in a "once through" cooling mode, with all cooling water passing through the condenser cooling system and then discharged to lower Vernon Pool. Under closed cycle, all cooling water is pumped through an array of mechanical draft cooling towers and then returned to the intake area for reuse as cooling water, until a portion is discharged as cooling tower blowdown. There is no cooling water discharge to the Connecticut River under closed cycle operations. Under hybrid cycle, Vermont Yankee may modify the amount of cooling water that passes through the cooling towers and the amount that is recirculated such that the discharge to the Connecticut River may vary in both temperature and volume. The typical range in temperature of the heated effluent discharged to the Connecticut River during the warmer summer months is approximately 80 to 90°F, with a very infrequent extreme case maximum of about 100°F. Discharge volume may vary anywhere between closed cycle volumes of 0 cfs to the maximum once–through cooling water pumping capacity of slightly over 800 cfs.

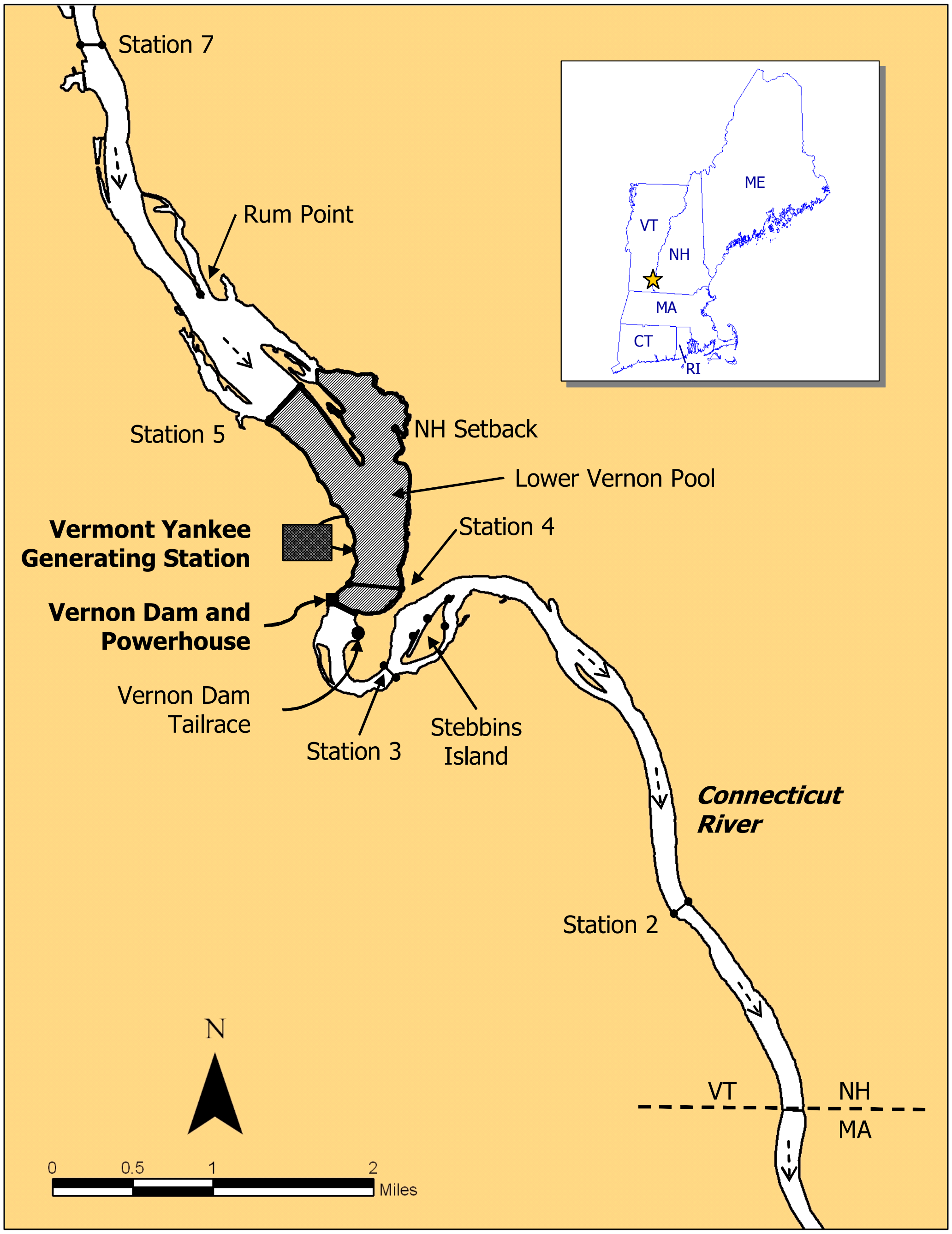


Figure 13-1
Connecticut River in the Vicinity of Vernon Pool

The Connecticut River flows are highly controlled by hydroelectric generation activities both upstream and downstream of Vermont Yankee. There are nine hydroelectric dams and three storage dams on the mainstem Connecticut River upstream of the Vernon Dam, and there are three hydroelectric dams and one pumped–storage facility downstream. Vernon Station, a 26.4 MW hydroelectric generating facility owned and operated by a U.S. Gen New England entity (PGE), is located on the west (VT) side of the 1200–ft long Vernon Dam (Figure 13-1). When river discharge approaches or exceeds station capacity (about 13,280 cfs), Vernon Station generates continuously, and any surplus flow is spilled from crest gates or deep gates. When the Connecticut River discharge is less than Vernon Station's capacity, all of the discharge past Vernon Dam is controlled by the facility. The stipulated minimum flow at Vernon Station is 1,250 cfs or inflow if less than 1,250 cfs. This situation leads to two characteristic patterns of regulated discharge: one of high and gradually varying flow, and one of frequent (two or more flow changes during each 24–hour period) cycling between lower and higher flows characterized by rapid transitions. The duration and magnitude of both the lower and higher flow during periods of cycling is determined largely by the availability of water from upstream sources. Vernon Station has nine hydroelectric units that range in maximum capacity from 1,280 to 1,970 cfs. "Lower" flows are maintained by operating one unit and may likewise vary from 1,250 (the permitted minimum flow) to 1,970 cfs. "Higher" flows are generated by operating multiple units and may vary from 2,560 to 13,280 cfs. Typically, "lower" flows would be maintained for a period of several hours during each day, while "higher" peaking power flows would be maintained the rest of the time. However, under very low flow conditions, PGE may operate Vernon Station may operate continuously at or near 1250 cfs for several consecutive days. Because the amount of heat Vermont Yankee can discharge to the Connecticut River is highly dependent on flow, Vernon Station operations and the upstream "ambient" water temperature determine to a large extent how much heat Vermont Yankee can discharge while maintaining compliance with its NPDES permit.

Temperature limits established in Vermont Yankee's existing NPDES permit correlate to two compliance periods: May 16–October 14 ("summer") and October 15–May 15 ("winter"). Compliance with the thermal limits established for both summer and winter periods is determined by calculating the plant–induced increase in Connecticut River water temperature above ambient conditions using Equation 1, which was initially proposed in the Station's 1978 § 316(a) demonstration [2], and has been the operative formula in every subsequent NPDES permit, including the current NPDES permit. Equation 1 is an elegant regulatory tool that restricts Vermont Yankee's thermal discharge to a fixed proportion of the river flow. Ambient upstream temperatures determine the proportion of waste heat that may be discharged. When the Connecticut River had a low capacity to accommodate additional heat, such as during periods of relatively low flow and high ambient water temperatures, Vermont Yankee is allowed a lower thermal discharge. When the Connecticut River had a high capacity to accommodate additional heat, such as during periods of relatively high flow and low ambient water temperatures, proportionally more waste heat may be discharged.

This compliance equation is given below:

$\Delta Tr = H/(\rho CpQr)$ **Equation 13-1**

Where:

ΔTr = the discharge–induced temperature increase in the Connecticut River

H = the heat rejection rate to the Connecticut River

ρ = the density of water

Cp = the specific heat of water

Qr = the Connecticut River flow rate at Vernon Dam

Ambient Connecticut River water temperature is monitored at upstream Station 7, a location 3.5 miles upriver of Vermont Yankee on the Vermont shore and well beyond any potential cooling water discharge effect (Figure 13-1). The actual change in Connecticut River water temperature due to Vermont Yankee's thermal discharge, as well as atmospheric influences, is monitored at downstream Station 3, located 0.65 miles downstream from the Vernon Dam and 1.4 miles downstream from Vermont Yankee.

During the summer period of May 16 through October 14, the current temperature increase limitation at downstream Station 3 is defined in Vermont Yankee's NPDES permit and shown in Table 13-1 in comparison to the proposed new summer period thermal limits.

Table 13-1
Vermont Yankee's Present and Proposed New Thermal Discharge Limits for the Summer NPDES Permit Period of May 16 through–October 14

Upstream Station 7 Ambient Temperature May 16–October 14	Calculated (by Equation 1) Temperature Increase Above Ambient at Downstream Station 3	
	Present Limits	Proposed New Limits
>78°F	2°F	2°F
>63°F, ≤78°F	2°F	3°F
>59°F, ≤63°F	3°F	4°F
≥55°F, ≤59°F	4°F	5°F
<55°F	5°F	5°F

Therefore, the proposed new limits for the summer period change the existing NPDES permit limits only by adding one degree Fahrenheit to the calculated temperature rise that is presently allowed. No change is proposed during the summer period when ambient Connecticut River water temperatures are above 78°F or below 55°F.

Connecticut River Flow Analysis

Hydrological and thermal conditions in the Connecticut River have been monitored since the late 1960s, providing a significant data set. The present Demonstration Report examined the hydrological and thermal conditions in the Connecticut River during the summer periods (May 16 through October 14) of 1998–2002 to establish recent flow conditions, confirm that such conditions accord the existing data set, and project how in–river thermal conditions might change under the proposed new thermal limits.

The long–term flow record for the Vermont Yankee area was generated from historical data for the North Walpole, NH gauging station[1], located less than 20 miles upstream of Vernon Station and operated by the United States Geological Survey (USGS). Flow data were transformed using log Pearson type III statistical methods, consistent with USGS protocols for developing streamflow statistics. Vernon Dam flow data (generated by Vernon Station) was then compiled to allow comparison with the North Walpole data. Hourly flow data were averaged to produce average daily flow. These data were then corrected based on differences in watershed area between North Walpole and Vernon (5,493 and 6,266 square miles, respectively, for a difference of 773 square miles) to allow direct comparison with the North Walpole data. Since the Connecticut River is heavily regulated, particularly during low flow periods and the 773 square mile tributary area is not, it was decided not to simply prorate flow based on the ratio 5,493/6,266. Instead, we took that portion of the 773 square mile of watershed that is gauged by USGS (West River at Jamaica, VT, which accounts for 179 square miles) and prorated this flow to estimate flow differences between North Walpole and Vernon Dam (772/179 = 4.32; and Vernon flow minus (4.32 times West River flow) = North Walpole flow. The average monthly (*i.e.*, average of the daily averages for the month) Vernon Station flows for each year from 1998–2002 were also assessed for comparison of recent flows to the historical (1973–2001) record.

Monthly and seasonal flows for the period 1998–2002 were generally representative of a wide range of flow conditions found in the historic period of record (Figure 13-2, Table 13-2). Seasonal flows for 1998–2002 were generally normal, with the probability of occurrence clustering around the 50th percentile mark. The exception was 2001, which was exceptionally dry for the summer period, occurring with a frequency that was greater than the 95th percentile (less than 1 year out of 20). This combination of representiveness, in conjunction with the recent examples of extreme low flow, supports the use of actual Connecticut River flow data from the recent (1998–2002) summer periods to examine the potential impact of the proposed new thermal limits, while ensuring confidence in the analysis. The very low flow months were nearly as extreme as can be expected for Vernon Pool, and analyses based on these low–flow conditions should be equally as extreme. Therefore, the use of the recent (1998–2002) Connecticut River flow record was considered conservative with respect to our evaluation of the proposed new permit thermal discharge limits for Vermont Yankee.

[1] A USGS gauging station immediately below Vernon dam was abandoned in 1973 due to backwater effects from downstream hydroelectric operations. Vernon Station also maintains a record of flow, but the majority of these data are not stored in a readily usable format. Consequently, a source of long–term flow data was needed and the USGS gauging station at North Walpole was selected as a surrogate. Although the flow record for the Connecticut River at North Walpole dates from the 1940s, it has been only since 1973 that minimum flows were maintained at 1200 cfs or higher. For that reason, only the flow record from 1973 to 2001 was used to generate the flow duration curves.

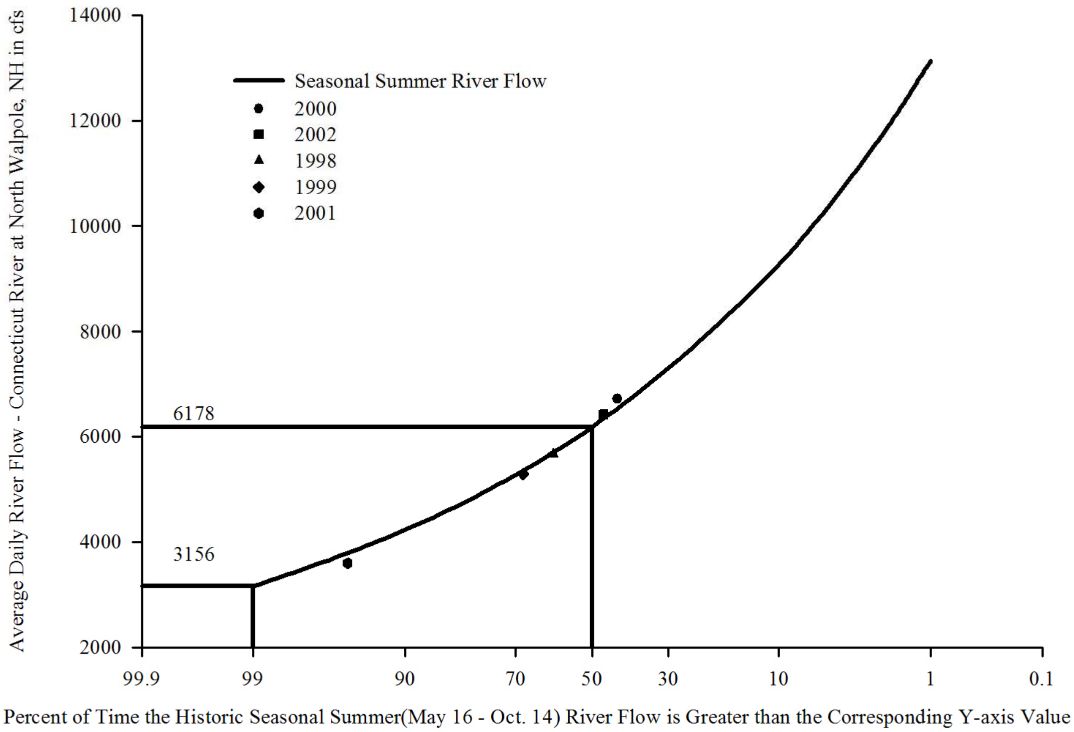


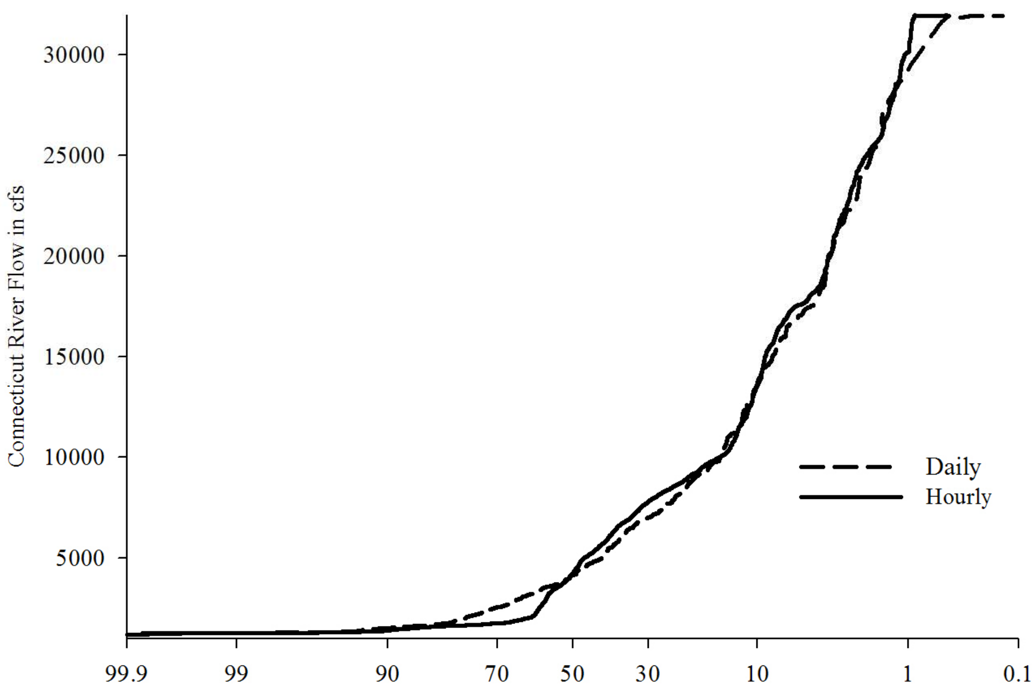
Figure 13-2
Historic Flow Duration Curve for the Connecticut River at North Walpole, NH–Summer Season (May 16 – October 14)

Table 13-2
Probability (Percent of Time) that Average Monthly and Seasonal Connecticut River Flow is Greater than Listed Values during the Summer Period (May 16 – Oct. 14)

Probability that Flow is Greater than Listed Flow (%)	May (16–31)	June	July	August	Sept.	October (1–14)	Seasonal
99	3563	2678	1908	1529	1157	1936	3156
95	4644	3404	2158	1797	1561	2329	3631
90	5998	4311	2470	2133	2065	2820	4225
75	7988	5568	3119	2734	2769	3732	4957
50	11305	7663	4201	3735	3943	5253	6178
25	17476	11297	7534	6340	5881	9427	8108
10	21178	13477	9534	7903	7044	11931	9266
2	30825	18847	18285	13715	9690	21711	11975
1	35164	21190	23705	16976	10784	27351	13141

The analyses of Connecticut River flow data are based on average daily flows, as derived from the North Walpole gauging station and Vernon Dam. Use of average daily flow is consistent with USGS methods. Moreover, in regulated rivers, particularly rivers where flows cycle widely over a 24–hour period, as here, hourly flow provides additional information about the frequency of occurrence of a particular flow. This is especially important for Vermont Yankee, because its thermal discharge is directly linked to flow and its NPDES permit requires hourly reporting.

Figure 13-3 presents the flow duration curve for Vernon Dam based on recent hourly data for the entire summer seasons of 1998–2002. For comparison, the summer season 5–year curve based on average daily flow data is also included. Although these curves largely follow one another, it can be seen that the average daily and the hourly flow duration curves are somewhat different at times, particularly in the 80th to 20th percentile range. The average daily curve reflects the general availability of water within the watershed (and from storage) on a daily basis. The hourly curve reflects that water is manipulated during a 24–hour period to achieve power generation objectives. Hourly flow is actually lower than daily flow for 30% of the time (between the 80th and 50th percentiles) and higher than daily flow for another 30% of the time (between the 50th and 20th percentiles). Thus, the hourly data provide a more accurate presentation of the typical flow constraints under which Vermont Yankee operates.



Percent of Time that Recent Summer Season Daily and Hourly Flows are Greater than the Corresponding Y-axis Values.

Figure 13-3
Recent (1998–2002) Summer Season Hourly and Daily Flow Duration Curves for the Connecticut River at Vernon Dam

Comparison of the probability of occurrence of both daily and hourly Connecticut River flows reveals that Vernon Dam flow was greater than 1275 cfs 99% of the time on both an hourly and daily basis (Table 13-3). However, at the 90% probability level, hourly flow was greater than 1412 cfs, whereas daily flow was 1507 cfs. Similarly, at a 75% probability level, hourly flow was 1688 cfs, while daily flow was 2206 cfs. This disparity between daily and hourly flows is typical of Rivers with the flow regulated for hydroelectric power, and is often most pronounced in those Rivers regulated for peaking power. At both higher and lower probabilities (>95%, < 20%), River flows for both hourly and daily events are more equal since in both ranges of flow, there is less opportunity for manipulating flow to achieve hydroelectric power generation objectives. In conclusion, recent and historic Connecticut River flow patterns are highly variable on hourly, daily, monthly and seasonal bases. However, recent (1998–2002) conditions were similar to those during at least the last 30 years.

Table 13-3
Probability (Percent of Time) that Connecticut River Flow is Greater than Listed Values during the Summer Period

Probability (%)	Hourly Flow (cfs)	Daily Flow (cfs)
99	1275	1275
95	1317	1333
90	1412	1507
75	1688	2206
50	4234	4163
25	8425	7716
10	13725	13550
1	30137	28250

Connecticut River Water Temperature Analysis

Because projections of the proposed increase in permit delta T are based in part on recent (1998–2002) Connecticut River water temperature data, it was appropriate to evaluate these recent data within the context of a longer–term temperature baseline. However, Vermont Yankee has only been operating under its existing allowable delta T values since the summer period of 1991. Further, while Connecticut River water temperature has been recorded at upstream Station 7 and downstream Station 3 since the late 1960s (Figure 13-1), much of these data are not in a readily usable format for this analysis. Consequently, a preferred alternative method for evaluating temperature was selected. Because Connecticut River water temperature is directly related to ambient air temperature and because air temperature is widely available, historic average seasonal and monthly air temperatures were developed for a nearby meteorological station.

Vernon Dam, cooperatively operated by Vernon Dam personnel and the National Climate Center, was selected, due to nearby location and to the length of the temperature record. Daily temperatures were available for 1952–1997. Vermont Yankee air temperature data were compiled for the summer period (May 16 through October 14) for 1998–2002, for comparison with the historic data.

Figure 13-4 presents the historic and recent seasonal data in the form of probability versus temperature, in much the same way the flow data were presented. The average seasonal temperature for each recent year is plotted on the temperature probability line to facilitate the determination of how seasonal air temperature in recent years compared to the historical temperature record. As is evident in Figure 13-4, average seasonal temperature for the last five years is well distributed along the historic frequency occurrence curve. Seasonal temperatures range from about 95% (historic seasonal temperatures were greater in 95% of the years) to approximately 20% (historic seasonal temperatures were greater in only about 20% of the years). This means that recent seasonal temperatures ranged from being quite cold (2000) to quite warm (1999) with respect to historic seasonal temperatures. The other recent seasonal temperatures were well distributed between the two extremes, which demonstrates that recent seasonal air temperature were representative of the range of air temperatures documented in nearly fifty years of historic record.

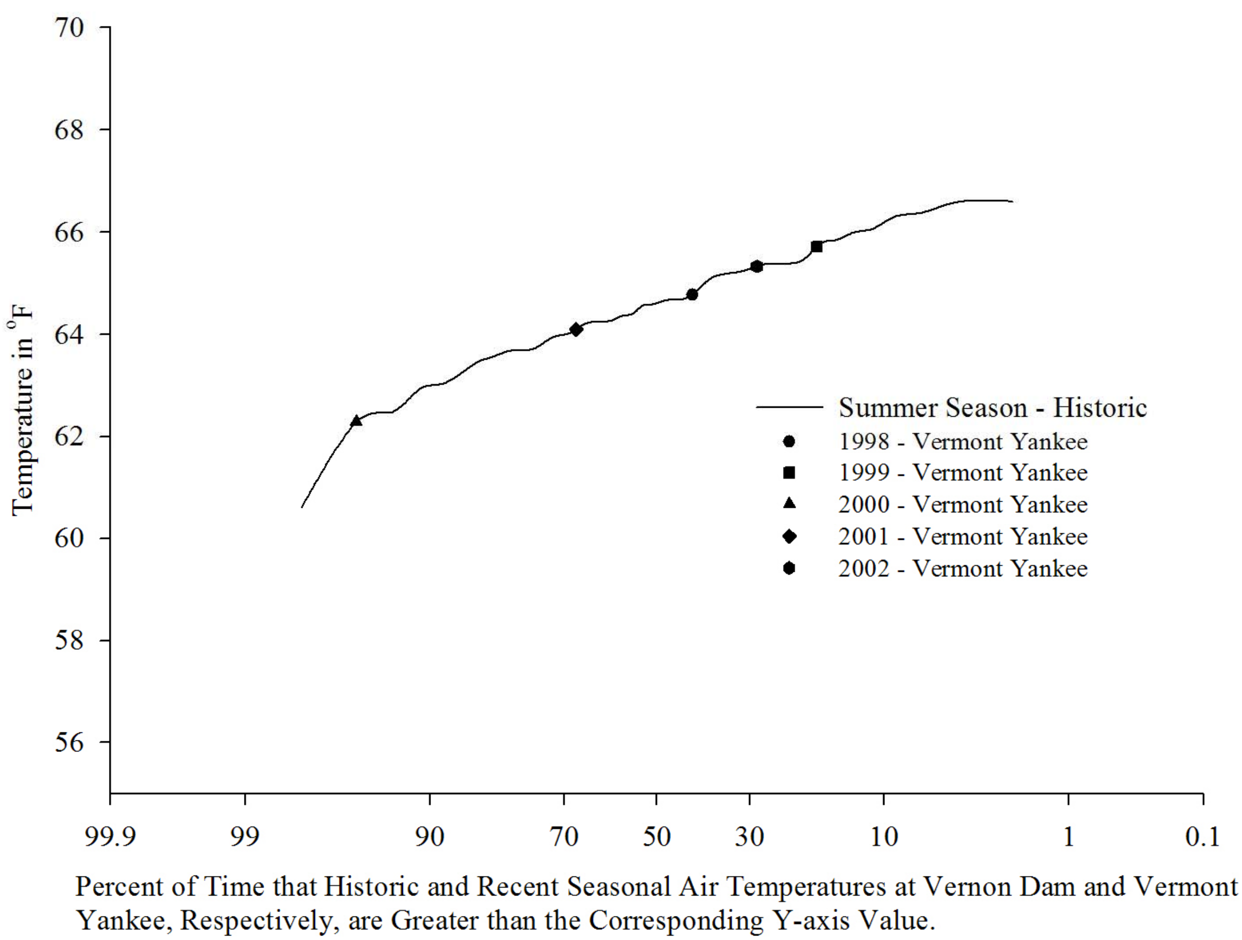


Figure 13-4
Historic Average Summer Season (May 16 through–October 14) Air Temperature Duration Curve at Vernon Dam

Based upon analysis of the air temperature data, we concluded that monthly temperatures experienced during the last five years are representative of a wide array of historic monthly temperatures. Again, this combination of representiveness, in conjunction with occasional exceptionally warm months, further supports the use of actual Connecticut River flow data from the recent (1998–2002) summer periods to examine the potential impact. The very high temperature month observed in August 2001 was nearly as extreme as can be expected for lower Vernon Pool, and analyses based at least in part on these high temperature conditions should be equally as extreme. As with Connecticut River flow, the use of the recent (1998–2002) river temperature record is conservative with respect to our evaluation of the proposed new permit thermal discharge limits for Vermont Yankee.

Pursuant to its NPDES permit, Vermont Yankee's compliance with NPDES permit thermal limits is determined by calculating the temperature rise that would result after complete mixing of the discharge with the Connecticut River flow, using Equation 3-1. During the summer period, the present allowable temperature increase in °F is given in Table 13-1. Ambient Connecticut River water temperatures are measured continuously to the nearest 0.1°F at the upstream Station 7, located 3.5 miles upriver of (Figure 13-1). Although not strictly related to compliance, Connecticut River water temperatures are also measured continuously to the nearest 0.1°F after complete mixing at downstream Station 3, located approximately 0.65 miles downstream from Vernon Dam and 1.4 miles downstream from the Vermont Yankee thermal discharge (Figure 13-1).

During the summer period, measured temperatures at downstream Station 3 are almost always higher than at upstream ambient Station 7 (Figure 13-5). More importantly, they are usually higher than can be explained solely by the addition of Vermont Yankee's discharge (upstream Station 7 plus permitted Delta T). Figure 13-5 presents a comparison of the observed or measured delta T at downstream Station 3, representing how much warmer the measured Connecticut River temperature is below Vernon Dam, as compared to the water temperature measured upstream from Vermont Yankee at upstream Station 7. Figure 13-5 also present the upstream ambient Station 7 hourly average water temperatures with the actual waste heat discharge rate from Vermont Yankee added to it under the existing permit limits and Connecticut River flow (calculated using Equation 3-1), for 2002 as a representative year. The measured hourly average river water temperature at downstream Station 3 is always higher than the upstream ambient (Station 7) temperature plus the permitted delta T due to the heat rejection from Vermont Yankee's discharge. This is consistent with historic monitoring data throughout the 1970s and 1980s, when Downstream Station 3 was observed to be typically 1–2°F higher than Upstream Station 7, even though, during the 70's, Vermont Yankee was not discharging heat to the Connecticut River during the summer period and, during the 80's, heat was discharged only experimentally [3] [4]. These data further support the observation that some other source of heat is contributing to downriver temperatures. Because there are no other thermal discharges between Vermont Yankee and downstream Station 3, the only other source of heat is atmospheric. Differences are generally greatest during the earlier (June and July) part of the summer season, the time when solar insolation reaches its annual maximum, which provides further evidence that atmospheric inputs heavily influence the observed downstream Station 3 temperatures.

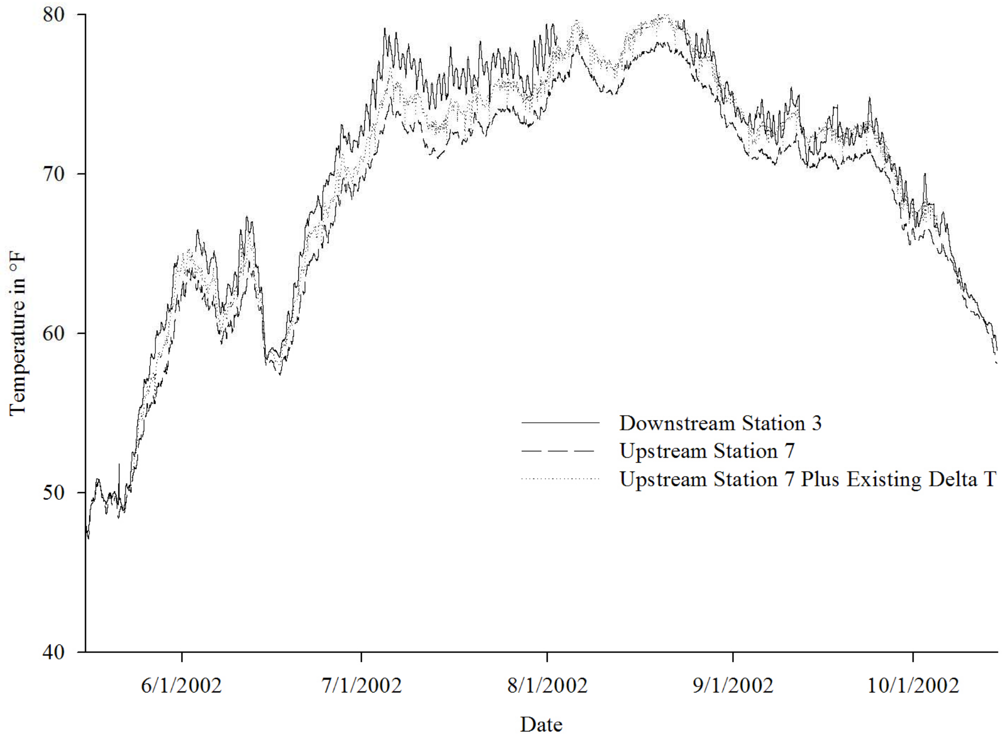


Figure 13-5
Hourly Temperature at Selected Monitoring Stations in the Vicinity of Vernon Dam During the 2002 Summer (May 16 through–October 14) Period

A key measure of the historic (and anticipated) thermal regime is a statistical determination of probability of occurrence of a particular temperature, often expressed as the probability of exceedance. Table 13-4 and Figure 13-6 present the probability of exceedance statistics for the recent past (1998–2002 summer seasons combined) for the observed hourly average Connecticut River water temperatures at both the upstream ambient (Station 7) and downstream mixed (Station 3) monitoring locations. The right–hand pair of columns in Table 13-4 presents the probability of exceedance statistics for Vermont Yankee's existing and proposed new thermal compliance limits. All probability determinations are based on hourly data for the entire five years of recent record, which means that there were more than 17,000 data points for each station location.

Connecticut River water temperature did not exceed 80°F at upstream Station 7 at any time during the last five years (Table 13-4, Figure 13-6). Upstream Station 7 temperatures were higher than 75°F for 10.19% of the time and higher than 70°F for 49.18% of the time. If the effects of the Vermont Yankee discharge were added to the temperature probabilities, Connecticut River water temperature (after complete mixing of the discharge and the Connecticut River flow) would have been greater than 80°F for 0.87% of the time during the last five years. Similarly, the Vermont Yankee discharge would be expected to cause the Connecticut River water temperature to be higher than 75°F for 19.04% (as compared to 10.19% of the time with no thermal discharge), and 70°F for 57.58% of the time (as compared to 49.18% of the time with no thermal discharge).

Table 13-4
Probability that Hourly Average Connecticut River Water Temperatures will be greater than Each Listed Temperature (°F) during the combined 1998-2002 Summer Seasons (May 16 through October 14) under Existing and Proposed Permit Thermal Discharge Limits for Vermont Yankee.

Temp(F)	Probability (%) and *Hours* of Exceedance							
	Upstream Ambient Measured at Station 7		Downstream Mixed Measured at Station 3		Calculated Downstream Station 3 with Existing Permit Delta T		Calculated Downstream Station 3 with Proposed Permit Delta T	
	Percent	*Hours*	Percent	*Hours*	Percent	*Hours*	Percent	*Hours*
86	Never greater than		Never greater than		Never greater than		Never greater than	
85	Never greater than		Never greater than		Never greater than		Never greater than	
84	Never greater than		0.06	*2*	Never greater than		Never greater than	
83	Never greater than		0.30	*11*	Never greater than		Never greater than	
82	Never greater than		1.61	*59*	Never greater than		Never greater than	
81	Never greater than		4.03	*147*	0.11	*4*	0.39	*14*
80	Never greater than		7.58	*277*	0.87	*32*	2.78	*101*
79	0.39	*14*	10.90	*398*	2.14	*78*	5.61	*205*
78	1.74	*64*	15.43	*563*	4.31	*157*	10.19	*372*
77	2.81	*103*	21.76	*794*	7.69	*281*	15.87	*579*
76	5.61	*205*	28.48	*1,039*	13.03	*475*	24.19	*882*
75	10.19	*372*	36.04	*1,315*	19.04	*695*	32.63	*1,190*
74	15.88	*579*	42.99	*1,568*	28.47	*1,039*	41.85	*1,527*
73	24.19	*882*	47.39	*1,729*	37.72	*1,376*	49.17	*1,794*
72	32.63	*1,190*	51.94	*1,895*	46.09	*1,681*	54.63	*1,993*
71	41.86	*1,527*	57.95	*2,114*	51.42	*1,876*	61.60	*2,247*
70	49.18	*1,794*	64.73	*2,361*	57.58	*2,101*	64.67	*2,359*
69	54.64	*1,993*	67.71	*2,470*	62.79	*2,291*	68.64	*2,504*
68	61.61	*2,248*	70.15	*2,559*	66.51	*2,426*	71.32	*2,602*
67	64.68	*2,360*	72.08	*2,629*	69.72	*2,543*	72.53	*2,646*
66	68.65	*2,504*	74.38	*2,713*	71.83	*2,620*	74.61	*2,722*

Table 13-4
Probability that Hourly Average Connecticut River Water Temperatures will be greater than Each Listed Temperature (°F) during the combined 1998-2002 Summer Seasons (May 16 through October 14) under Existing and Proposed Permit Thermal Discharge Limits for Vermont Yankee. (Continued)

Temp(F)	Probability (%) and *Hours* of Exceedance							
	Upstream Ambient Measured at Station 7		Downstream Mixed Measured at Station 3		Calculated Downstream Station 3 with Existing Permit Delta T		Calculated Downstream Station 3 with Proposed Permit Delta T	
	Percent	*Hours*	Percent	*Hours*	Percent	*Hours*	Percent	*Hours*
65	71.33	*2,602*	76.53	*2,792*	73.50	*2,681*	77.48	*2,826*
64	72.54	*2,646*	79.16	*2,888*	75.95	*2,771*	81.01	*2,955*
63	74.62	*2,722*	82.39	*3,006*	79.08	*2,885*	84.77	*3,092*
62	77.49	*2,827*	86.44	*3,153*	83.52	*3,047*	88.31	*3,222*
61	81.02	*2,956*	89.59	*3,268*	87.14	*3,179*	91.37	*3,333*
60	84.78	*3,093*	92.87	*3,388*	90.07	*3,286*	94.29	*3,440*
59	88.32	*3,222*	94.51	*3,448*	92.90	*3,389*	95.39	*3,480*
58	91.38	*3,334*	95.70	*3,491*	95.04	*3,467*	96.06	*3,504*
57	94.30	*3,440*	96.23	*3,510*	95.96	*3,501*	96.76	*3,530*
56	95.40	*3,480*	97.15	*3,544*	96.73	*3,529*	98.47	*3,592*
55	96.07	*3,505*	98.60	*3,597*	97.88	*3,571*	99.36	*3,625*
54	96.77	*3,530*	99.07	*3,614*	98.64	*3,598*	99.67	*3,636*
53	98.42	*3,590*	99.68	*3,636*	99.17	*3,618*	99.98	*3,647*
52	99.37	*3,625*	Always greater than		Always greater than		99.83	*3,642*
51	99.68	*3,636*	Always greater than		Always greater than		99.89	*3,644*
50	99.99	*3,648*	Always greater than		Always greater than		Always greater than	
49	Always greater than		Always greater than		Always greater than		Always greater than	

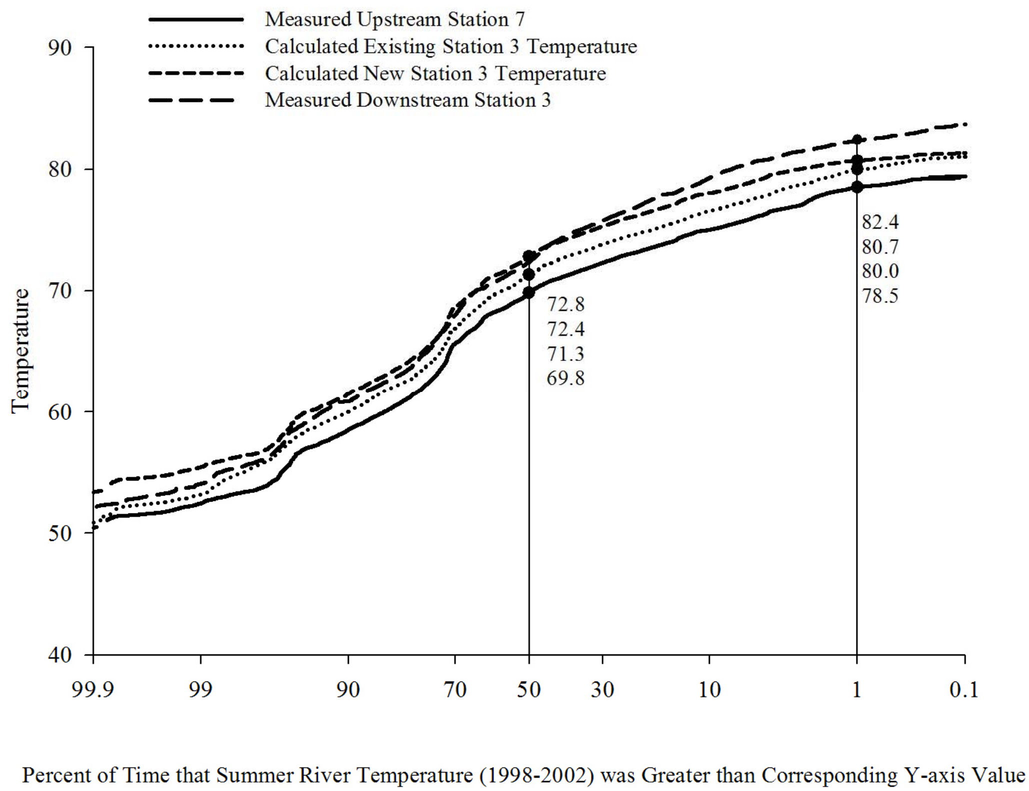


Figure 13-6
Comparison of Projected Temperature Response in the Connecticut River for Background, Existing and Proposed Vermont Yankee Discharge Permit Conditions

Actual hourly average Connecticut River water temperatures, as measured at downstream Station 3, were considerably higher than would have been predicted by taking the upstream ambient Station 7 water temperatures and adding the temperature rise attributed to Vermont Yankees thermal discharge (Table 13-4). The measured temperature at downstream Station 3 was greater than 80ºF for 7.58% of the time or about 6.71% more of the time than accounted for by Vermont Yankee's discharge. The observed maximum temperature at downstream Station 3 actually was greater than 84ºF only 0.06% of the time (about 2 hours per year during the summer period), and never exceeded 85ºF. Similarly, the probability of exceedance at downstream Station 3 for all temperatures listed was substantially higher than could be explained by Vermont Yankee's discharge. For example, downstream Station 3 was greater than 75ºF for 36.04% of the time (versus a calculated 19.04% of the time) and was greater than 70ºF for 64.73% of the time (versus an expected 57.58%).

The existing thermal discharge from Vermont Yankee increased the amount of time that Connecticut River water temperature was greater than naturally occurring values by no more than 10% for the temperature range expected during the summer season (Table 13-4) and generally by considerably less. In fact, at temperatures above 76°F, predicted increases are 5% or less. The observed maximum temperature for the most recent 5–year record was 84°F at downstream Station 3, and this maximum temperature was observed for just 2 hours/year during the 5–year summer period.

Vermont Yankee's proposed new summer permit limits would increase the hourly average Connecticut River water temperature during the summer season after complete mixing only by as much as 1°F (Table 13-4 and Figure 13-6). With allowable delta T's as defined by the proposed new permit limits, the calculated maximum complete mixed temperature in the Connecticut River (upstream ambient plus proposed new Delta T) would be expected to be similar to existing conditions (i.e., maximum temperature would exceed 81°F but would not exceed 82ºF). However, the frequency of occurrence of temperatures greater than 81°F would be slightly higher under proposed conditions (0.39%), compared to 0.11% of the time for existing conditions (14 hours per season versus 4 hours). The calculated temperature would exceed 80°F for 2.78% of the time. This compares to 0.87% of the time for existing conditions and represents an increase of about 69 hours/year for the summer period. It is further predicted that the calculated maximum temperature would be greater than 75°F for 32.63% of the time and greater than 70°F for 64.67% of the time, compared to 19.04% and 57.58% of the time, respectively, under existing conditions.

In summary, the proposed increase in delta T of as much as 1°F (depending on ambient temperature) would result in only slightly higher summer temperatures in the Connecticut River downstream from Vernon Dam. The maximum calculated temperature based on upstream Station 7, plus the proposed new Delta T, would only slightly exceed 81°F, which is virtually identical to existing conditions. Under the proposed new summer period permit limits, the other expected temperatures (50ºF–85ºF) at Downstream Station 3 would be exceeded by varying amounts, depending on the actual temperature, but the increase would be no greater than 14% and generally much less. For example, the proposed new NPDES permit change in summer period Delta T would result in the occurrence of mixed Connecticut River water temperatures at or above 75ºF about 13.6% more of the time compared to the existing permit Delta T. A significantly lesser effect would exist for temperatures below 69ºF and above 77°F, with less than a 6% increase in the amount of time a given temperature is greater than 77°F and less than 69°F.

Thermal Plume Model

A three–dimensional time–varying hydrothermal model (BFHYDRO, [5]) was developed, calibrated, confirmed and used to predict the extent of Vermont Yankee's thermal plume in lower Vernon Pool of the Connecticut River under existing and proposed new summer (16 May–14 October) thermal discharge limits.

The objectives of hydrothermal modeling were to:

- Forecast changes in the Connecticut River thermal regime of the lower Vernon Pool under existing and proposed new summer thermal discharge limits,
- Quantify the gain or loss of fish habitat with respect to the forecasted thermal regime changes, and
- Predict the effects, if any, of the proposed new thermal discharge limits on water temperatures in the Vernon Dam fishway.

The hydrothermal model was developed to predict changes within the entire 25 miles of Vernon Pool between Vernon Dam and Bellows Falls Dam (Figure 13-1). However, the relevant predictions are for the Connecticut River in the vicinity of Vermont Yankee in lower Vernon Pool. Lower Vernon Pool was defined as the 1.4 mile–long segment upstream from Vernon Dam, which includes Vermont Yankee's intake and discharge (Figure 13-1). The existing permit summer limits and the proposed new permit summer limits were modeled to provide a forecast of changes in the thermal regime in lower Vernon Pool under average case (50% exceedance) and extreme case (1% exceedance) conditions. Probability of occurrence of hourly average Connecticut River flows and temperatures were used to define the average and extreme case conditions with respect to input for the hydrothermal model. For additional conservatism in the model predictions, the average and extreme case flow and water temperature values were selected from the warm July–August period, not from the entire summer period as defined by the current NPDES permit (*i.e.*, 16 May–14 October).

The average case represented the hourly Vernon Dam flow and hourly upstream ambient (Station 7) Connecticut River water temperature at the exact mid–point among all of the observed hourly flows and temperatures during the recent (1998–2002) five July–August summer periods (Table 13-5). Half of the hourly flows and half of the hourly upstream ambient (Station 7) water temperatures fall above, and half fall below, the specified average (50%) probability of occurrence values. The extreme case flow and temperature conditions for hydrothermal modeling were defined as the lowest flow and warmest ambient water temperature with a frequency of occurrence of 1% during July–August. The selected hourly average Connecticut River flow for the extreme case was so low that nearly all (99%) of the hourly flows in the recent (1998–2002) five summer periods were greater than this value (Table 13-5). The selected extreme case upstream Station 7 water temperature was similarly so high that nearly all (99%) of the hourly temperature observations in the recent (1998–2002) five July–August periods were less than this value. Conservatism was also incorporated into the modeling projections by assuming that the discharge flow from Vermont Yankee was always at 100°F, even though this rarely occurs. Another conservative assumption was that the amount of waste heat discharged from Vermont Yankee is based on thermal discharges occurring exactly at the NPDES permit limits, which rarely occurs because Vermont Yankee typically operates the plant cooling system with a margin of about 0.2°F or more below the permit limit in an attempt to accommodate rapid changes in Connecticut River flow.

Fish habitat changes due to the proposed new thermal regime were quantified based on the volume or area in lower Vernon Pool predicted to be at or above a specified summer water temperature. The thermal plume temperature contours in lower Vernon Pool, derived from predictions based on the existing permit summer limits, provide the baseline for evaluation of habitat change. The increase in Connecticut River volume or Connecticut River bottom area predicted by the model for the proposed new permit summer limits quantifies the change from this baseline due to the anticipated increase in thermal discharge from Vermont Yankee under average and extreme case scenarios.

Table 13-5
Connecticut River Flow and Upstream Temperature, and Vermont Yankee Discharge Flow and Temperature Defining Average (50%) and Extreme (1%) Case Hydrothermal Modeling Scenarios for July–August

Parameter	Average (50% Exceedance) Case		Extreme (1% Exceedance) Case	
	Existing Permit Limit (2°F Delta T)	Proposed New Permit Limit (3°F Delta T)	Existing Permit Limit (2°F Delta T)	Proposed New Permit Limit (3°F Delta T)
River Flow (cfs)	3420	3420	1275	1275
Upstream Temperature (°F)	73.5	73.5	79.0	79.0
Discharge Flow (cfs)	258	387	121	182
Discharge Temperature (°F)	100.0	100.0	100.0	100.0

The increase in Vermont Yankee's thermal plume volume in lower Vernon Pool under the existing and proposed new permit conditions is illustrated in Figure 13-7 for the average (50% exceedance) and extreme (1% exceedance) cases. Plume volumes for the average case remain indistinguishable under both existing and proposed new thermal limits until the water temperature approached and exceeded 73°F. The average case plume volume for the proposed new thermal limits exhibited increases of between 0.1% and 5.0% for each one-degree temperature contour from 73°F to 82°F when compared to the existing permit conditions. Connecticut River water temperature in lower Vernon Pool never got above 82°F for the average case existing or proposed new permit discharge limits, based on the resolution of the hydrothermal model. Plume volumes for the extreme case exhibited a pattern similar to the average case (Figure 13-7), with volumes remaining indistinguishable under existing and proposed new thermal limits until the water temperature approached and exceeded ambient (79°F). The extreme case plume volume for the proposed new permit limits exhibited increases of between 0.3% and 10.8% for each temperature contour from 79°F to 86°F compared to the existing permit conditions. The Connecticut River water temperature in lower Vernon Pool was never warmer than 87°F for the extreme case existing or proposed new permit discharge limits, based on the resolution of the hydrothermal model.

Slight changes were also observed in the Connecticut River bottom area in contact with the Vermont Yankee thermal plume under the proposed new permit limits compared to existing conditions for both average and extreme cases (Figure 13-8). Bottom areas for the average (50% exceedance) case remain indistinguishable under both existing and proposed new thermal limits until the water temperature approached and exceeded 73°F. The average case bottom area for the proposed new thermal limits exhibited increases of between 0.0% and 4.8% for each one-degree temperature contour from 73°F to 82°F when compared to the existing permit conditions. The Connecticut River bottom in contact with the thermal plume in lower Vernon Pool was never warmer than 83°F for the average case existing or proposed new permit discharge limits, based on the resolution of the hydrothermal model. Vermont Yankee's plume bottom areas for the extreme (1% exceedance) case exhibited a similar pattern as seen for the average case, with bottom areas remaining indistinguishable under existing and proposed new thermal limits until the water temperature approached and exceeded ambient (79°F), and then the predicted bottom areas for the proposed new limits increased between 0.1% and 7.7% from the existing conditions for each one-degree temperature contour from 79°F to 86°F (Figure 13-8). The benthic substrate in the lower Vernon Pool never got above 86°F for the extreme case existing or proposed new permit discharge limits, based on the resolution of the hydrothermal model.

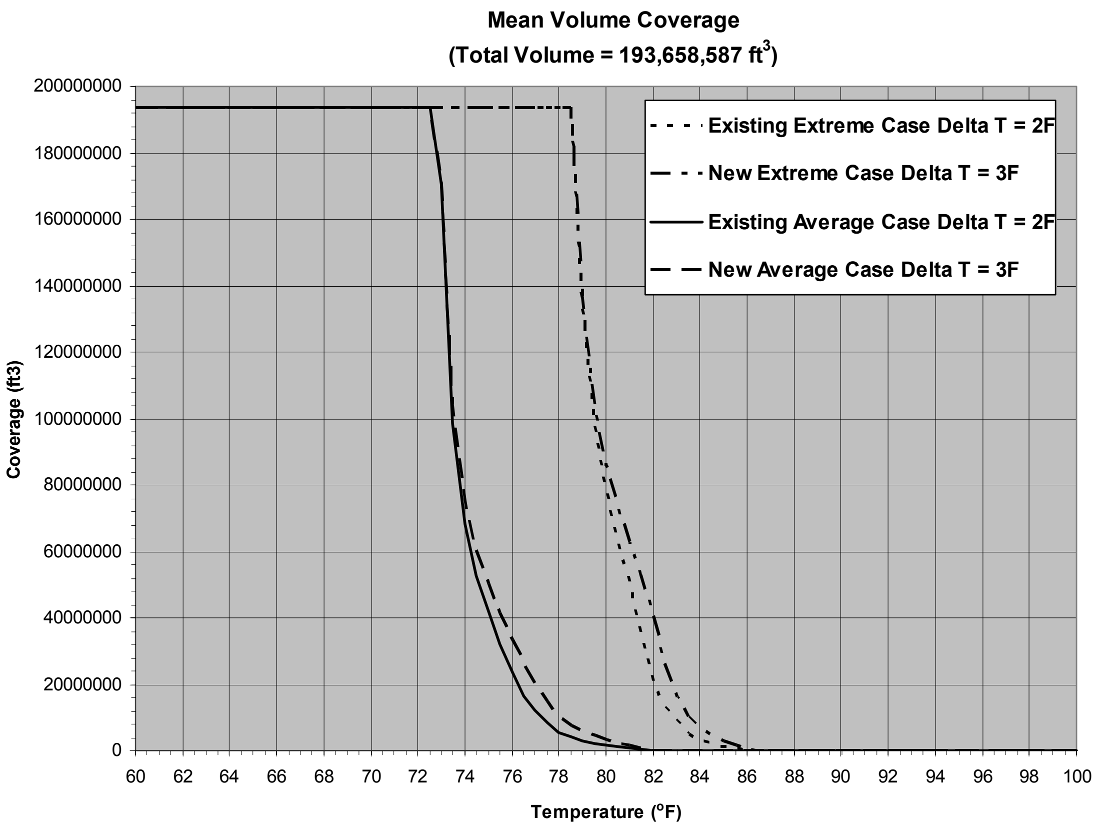


Figure 13-7
Cumulative Mean Volume in Lower Vernon Pool Predicted to be at or above each One Degree Temperature Contour for the Average Case (50% Occurrence of Connecticut River Flow and Upstream Ambient Temperature) and Extreme Case (1% Occurrence) August Conditions for Existing and Proposed New Vermont Yankee Permit Thermal Limits

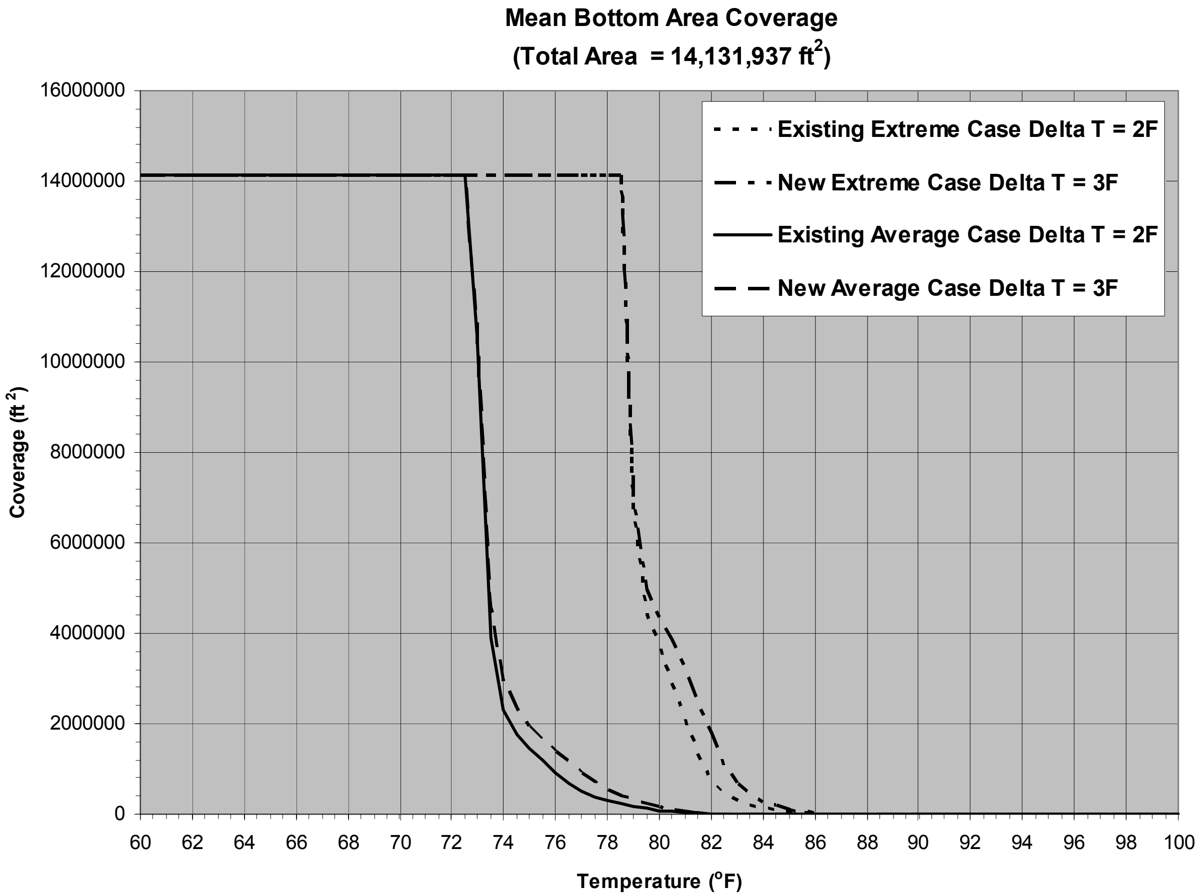


Figure 13-8
Cumulative Mean Bottom Area in Lower Vernon Pool Predicted to be at or above each One Degree Temperature Contour for the Average Case (50% Occurrence of Connecticut River Flow and Upstream Ambient Temperature) and Extreme Case (1% Occurrence) August Conditions for Existing and Proposed New Vermont Yankee Permit Thermal Limits

Fish Habitat Impact Assessment

The predicted changes in habitat suitability for each of the nine RIS within lower Vernon Pool of the Connecticut River were characterized for their selected life history or thermal response functions under proposed thermal limits for the average case (occurs 50% of the time) and extreme case (occurs 1% or less of the time) hydrothermal model scenarios. This evaluation relied on the criteria response temperatures drawn from the available published literature. Habitat suitability in lower Vernon Pool was quantified in terms of volume and bottom area that changed between existing and proposed new permit limits due to predicted river water temperatures occurring in excess of the criteria temperatures. A data table for each RIS of fish summarized the habitat changes relative to the percentage of total volume (19,366 x 10^4 cubic feet) and total bottom area (324 acres) that may exceed the thermal effects criteria under the existing and proposed new summer temperature limits for the average and extreme case hydrothermal model scenarios. As a measure of the predicted changes in the tailrace water immediately downstream from Vernon Dam, these tables also show the percent and hours of the summer period (16 May through 14 October) during 1998 through 2002 that the calculated mixed water temperature at Station 3 is equal to or higher than the criteria temperatures.

The temperature response data used for each of the nine RIS of fish are presented in Table 13-6, and include: (1) maximum temperature for summer survival and/or upper incipient lethal temperature, (2) optimum temperature for growth, (3) avoidance temperature, and (4) preferred temperature, (5) spawning, and (6) early life history.

Table 13-6
Water Temperatures (°F) for Thermal effects Parameters Applied to Vermont Yankee's epresentative Important Species (RIS) of Fish

RIS Species of Fish and Primary Literature References	Thermal Effects Parameter and Temperature in °F					
	Maximum (UILT) (*Exclusionary*)	Avoidance (*Exclusionary*)	Optimum for Growth (*Indicator*)	Preferred (*Indicator*)	Spawning (*Indicator*)	Early Life History (*Indicator*)
American shad [6, 7, 8, 9, 10, 11, 12]	90	86	70	65	65	70
Atlantic salmon [13]	82	78	na	na	na	na
Spottail shiner [14]	95	95	86	86	64	70
Smallmouth bass [12, 20, 21, 22]	98	95	90	81	63	70
Yellow perch [14, 23, 24]	90	83	74	77	50	65
Walleye [14, 17, 24, 25]	89	76	74	72	48	54
Largemouth bass [16, 17, 21, 26, 27]	95	90	83	86	70	75
Fallfish [12, 14, 28]	90	82	68	na	60	65
White sucker [14, 17, 21, 29, 30]	88	86	81	81	60	65

There are fundamentally two different classes of thermal effects parameters among the six categories; exclusionary and indicator temperature limits. The maximum temperature and the avoidance temperature are considered exclusionary thermal effects because the fish species will not be found in habitat where the water temperature is at or above the reported values for any sustained period of time. The fish species is, therefore, excluded from use of the portion of the habitat for the time that the habitat is at or above the maximum or avoidance temperatures.
The remaining four categories of thermal effects parameters, optimum, preferred, spawning and early life history, are considered indicator parameters because they are water temperature values that coincide with the physiological or life history events represented by the thermal effects parameters. For example, a given fish species is not likely to change its distribution in response

to the water temperature in the habitat occupied that is not at the optimum or preferred temperature. The fish species is likely to remain exposed to water temperatures that are different than the optimum or preferred temperatures for different periods of time under existing or predicted new thermal discharge limits rather than actively search for optimum or preferred conditions. Likewise, the spawning and incubation or larval development thermal effects parameters describe the water temperatures occurring during those life history events.

The "maximum temperature" for summer survival is generally regarded as a peak temperature during the warmest time of the year that can be tolerated by a species for brief time periods, and is therefore considered exclusionary. The maximum temperature is higher than the indicator temperatures. The maximum temperature is routinely derived from field observations. The "upper incipient lethal temperature" or UILT is
a lethal threshold temperature obtained from laboratory experiments in which fish are removed from a temperature they are acclimated to and placed in a range of other temperatures that typically result in a range of survival from 100% to 0%. The ultimate upper incipient lethal temperature or UUILT is the temperature beyond which no increase in lethal temperature results from increase in acclimation temperature. Fish will avoid water temperatures that exceed the avoidance temperature when escape routes are available and will not succumb to lethal temperatures unless trapped. "Optimum temperatures for growth" are developed from field observation of feeding behavior, which usually yield a range of temperatures, or more precisely from physiological experiments. A commonly used temperature criterion for growth is the maximum weekly average temperature or MWAT. The MWAT is considered the highest temperature that will maintain growth of the organism at levels necessary for sustaining actively growing and reproducing populations. The MWAT is calculated as a temperature that should not exceed one–third of the range between the optimum temperature for growth and the UUILT of the species. For many species, the final preferred temperature has been found to be coincident with optimum temperatures for growth and is used as a surrogate for optimum growth temperature when the latter is unavailable. Since fish are motile, behavior responses to a thermal variation include avoidance, preference or merely a physiological adjustment as they pass through or remain exposed to it. Determination of temperatures that are avoided and preferred is usually based on laboratory experiments, but field collection data provided useful information when reported in the literature. The mid–range of the observed and reported temperatures for spawning and for egg incubation and larval development were selected as the indicator temperatures for these life history events. While the effects of all variables that regulate fish populations are incompletely understood, direct effects of temperature are accepted and allow evaluation of the potential impacts of the predicted temperature regime from the proposed new thermal limits by Vermont Yankee during the summer period.

The following section illustrates how the predicted changes in the Connecticut River thermal conditions due to Vermont Yankee's proposed new thermal discharge limits were quantified and interpreted for one of the RIS, yellow perch. For brevity of this paper, similar interpretations will not be provided for the remaining eight RIS in Vermont Yankee's 316(a) Demonstration. Yellow perch was selected because it represents an abundant resident fish species in Vernon Pool of the Connecticut River that is considered to be intermediate in its reported tolerance to pollution.

Impact Assessment for Yellow Perch

Yellow perch (*Perca flavescens*) has a circumpolar distribution in fresh waters of the northern hemisphere [12]. Within North America, yellow perch are widespread and very adaptable. They are found in a variety of warm– to cool–water habitats, and have historically occupied a range from Nova Scotia to South Carolina along the east coast extending northwesterly through the Great Lakes states into Alberta, Canada. Yellow perch has been successfully introduced into nearly all states west and south of its historical range [12]. They are often common in clear open water habitats with moderate vegetation, typically less than 30 feet deep [34].

Yellow perch represents the lentic insectivore trophic guild of fish species that are reported to be intermediate in their tolerance to pollution (Table 13-7). Some researchers consider yellow perch to be piscivorous or a generalist forager, however these alternate trophic guilds undoubtedly apply to different age classes, with general foraging occurring in the earlier life stages, a predominance of piscivory in the older and larger individuals, and insectivory occurring throughout their life. The relatively high abundance of yellow perch in lower Vernon Pool, and their much lower abundance in the Vernon Dam tailwaters, supports this trophic guild classification and representation within the Connecticut River fish community as a lentic insectivore.

Yellow perch shares the same trophic guild and pollution tolerance classification as two other Vermont Yankee RIS, the spottail shiner and American shad (Table 13-7). Although some researchers classify the spottail shiner as a generalist forager that is intolerant of pollution, the predominant classification is the same as for yellow perch. American shad have also been classified in the filter feeder tropic guild, which undoubtedly applies to the ability of juveniles to feed on *Daphnia* sp. and other freshwater planktonic crustaceans, however both spottail shiner and juvenile American shad are also reported to feed on insect larvae (chironomids) if abundant [12]. Both spottail shiner and juvenile American shad remain as forage fish during their presence in lower Vernon Pool, while yellow perch can only be classified as forage during their larval and juvenile life stages. As a RIS, yellow perch also represents other non–RIS lentic insectivores with intermediate tolerance found in the Vernon Pool fish community, including the closely–related tessellated darter, two minnow species (common shiner and spotfin shiner), and three centrarchids (redbreast sunfish, pumpkinseed sunfish, and bluegill, Table 13-7). Therefore, conclusions about the interaction of yellow perch with the existing and proposed new Vermont Yankee thermal limits embodies USEPA's requirements for the RIS, and are also sufficiently representative of the other members of the fish community within the same habitat guild, trophic guild, and tolerance classification.

Throughout over 30 years of monitoring at Vermont Yankee, juvenile and adult yellow perch have been numerically important components of the Connecticut River fish community sampled by electrofishing and trap nets. They are found throughout Vernon Pool including habitats exposed to the thermal effluent. In lower Vernon Pool, yellow perch comprised about 16% of the catch in 1968–1980, 24% in 1981–1990, and 39% in 1991–2002. In upper Turners Falls Pool (study area downstream from Vernon Dam), yellow perch relative abundance consistently has been lower than in Vernon Pool, ranging between 8% and 9% of the total catch in each of the three review periods. The lower relative abundance of yellow perch below Vernon Dam probably reflects its preference for open water habitats with moderate vegetation, which are found upstream in lower Vernon Pool, and avoidance of the relatively turbulent riverine habitat in the Vernon Dam tailrace.

The annual catch of yellow perch by electrofishing from 1991–2002 in lower Vernon Pool (Table 13-8) was highest in 2001 (114 fish per hour) and lowest in 1992 (32 fish per hour). Catch per unit effort (CPUE) of yellow perch by electrofishing was lower downstream from Vernon Dam where annual CPUE was highest in 1999 (17 fish per hour) and lowest in 2002 (0.4 fish per hour). The yellow perch annual CPUE by trap netting in lower Vernon Pool between 1991 and 1999 was highest in 1997 and 1998 (10 fish per day) and lowest in 1992, 1993, 1996, and 1999 (4 fish per day). The annual CPUE by trap netting downstream from Vernon Dam was lower than in Vernon Pool, and ranged between a high of 3 fish per day in 1995 and 1999 and a low of 1 fish per day in 1992, 1994, 1996 and 1998.

Table 13-7
Trophic Guilds and Tolerance Classifications for Connecticut River Fish Species Present in the 1991–2002 Fish Samples from Lower Vernon Pool and the Vernon Dam Tailrace

() Representative Important Species	Habitat Guild[1]	Trophic Guild[2]	Trophic Exceptions[3]	Tolerance	Tolerance Exceptions[3]
American shad	Lentic and Lotic	Insectivore	Filter feeder	Intermediate	
Atlantic salmon (parr and smolts)	Lotic	Insectivore		Intermediate	Intolerant
Spottail shiner	Lentic and Lotic	Insectivore	Generalist	Intermediate	Intolerant
Fallfish	Lotic	Generalist		Intermediate	
White sucker	Lentic and Lotic	Omnivore	Insectivore, Generalist	Tolerant	
Smallmouth bass	Lotic	Piscivore	Insectivore	Intermediate	Intolerant
largemouth bass	Lentic	Piscivore	Insectivore	Intermediate	Tolerant
Yellow perch	Lentic	Insectivore	Piscivore, Generalist	Intermediate	
Walleye	Lentic and Lotic	Piscivore		Intermediate	
Other Fish Species Present at VY	**Habitat Guild[1]**	**Trophic Guild[2]**	**Trophic Exceptions[3]**	**Tolerance**	**Tolerance Exceptions[3]**
Sea lamprey (ammocetes)	Lentic	Filter feeder		Intermediate	
American eel	Lentic	Piscivore	Generalist	Intermediate	Tolerant
Blueback herring	Lentic	Filter feeder		Intermediate	
Gizzard shad	Lentic	Omnivore	Filter feeder; Herbivore	Intermediate	Tolerant
Goldfish	Lentic	Omnivore	Generalist	Tolerant	
Common carp	Lentic	Omnivore	Generalist	Tolerant	
Eatern silvery minnow	Lentic	Herbivore	Omnivore	Intermediate	Intolerant

Table 13-7
Trophic Guilds and Tolerance Classifications for Connecticut River Fish Species Present in the 1991–2002 Fish Samples from Lower Vernon Pool and the Vernon Dam Tailrace (Continued)

() Representative Important Species	Habitat Guild[1]	Trophic Guild[2]	Trophic Exceptions[3]	Tolerance	Tolerance Exceptions[3]
Other Fish Species Present at VY	**Habitat Guild[1]**	**Trophic Guild[2]**	**Trophic Exceptions[3]**	**Tolerance**	**Tolerance Exceptions[3]**
Common shiner	Lentic and Lotic	Insectivore	Generalist	Intermediate	
Golden shiner	Lentic	Omnivore	Insectivore, Generalist	Tolerant	
Spotfin shiner	Lentic	Insectivore		Intermediate	Tolerant
Mimic shiner	Lentic	Insectivore	Generalist	Intolerant	Intermediate
Yellow bullhead	Lentic and Lotic	Insectivore	Omnivore, Generalist	Tolerant	Intermediate
Brown bullhead	Lentic	Insectivore	Generalist	Tolerant	Intermediate
Northern pike	Lentic	Piscivore		Intermediate	Intolerant
Chain pickerel	Lentic	Piscivore		Intermediate	
Brook trout	Lentic and Lotic	Piscivore	Insectivore	Intermediate	Intolerant
Banded killifish	Lentic	Insectivore		Tolerant	Intermediate
White perch	Lentic	Piscivore	Insectivore	Intermediate	
Rock bass	Lotic	Piscivore	Insectivore	Intermediate	Intolerant
Redbreast sunfish	Lotic	Insectivore	Generalist	Intermediate	
Pumpkinseed	Lentic	Insectivore	Piscivore, Generalist	Intermediate	
Bluegill	Lentic	Insectivore	Generalist	Intermediate	Tolerant
Black crappie	Lentic	Piscivore	Insectivore, Invertivore	Intermediate	
Tessellated darter	Lentic	Insectivore		Intermediate	

[1] [15].
[2] Appendix C in [34].
[3] Exceptions were taken when there was disagreement in one or more of the seven references regarding the trophic guild or tolerance classification of a species; the alternatives are shown.

Table 13-8
Annual Mean Catch per Unit of Effort for Yellow Perch in Vermont Yankee's NPDES Monitoring Program Conducted in the Connecticut River near Vernon, Vermont, 1991–2002

Annual Mean Catch per Unit of Effort for Yellow Perch in Vermont Yankee's NPDES Monitoring Program				
Year	**Electrofishing in Lower Vernon Pool (fish/hr)**	**Electrofishing in the Vernon Dam Tailwaters (fish/hr)**	**Trap Netting in Lower Vernon Pool (fish/24 hr)**	**Trap Netting in the Vernon Dam Tailwaters (fish/24 hr)**
1991	665	5	5	2
1992	321	2	4	1
1993	45	4	4	2
1994	61	3	7	1
1995	45	1	7	3
1996	1002	75	4	1
1997	81	10	109	23
1998	834	78	10	1
1999	5960	176	4	3
2000	741	1	ns	ns
2001	11408	56	ns	ns
2002	4443	<1	ns	ns

A nonparametric Mann–Kendall test was used to examine the 1991–2002 annual time series of yellow perch CPUE for significant increasing or decreasing trends [356]. The field sampling design has consistently required sampling at the same stations with the same gear during the same months in each of the twelve consecutive years in the electrofishing time series, and for each of the nine consecutive years in the trap net time series (1991–1999), making annual mean CPUE the appropriate response variable in the time series analysis. The Mann–Kendall test is robust with respect to parametric assumptions of data normality and variance heterogeneity [356][367], and was performed on untransformed annual mean CPUE. The null hypothesis was that there is no statistically significant ($p<0.05$) trend in yellow perch abundance during the period analyzed as measured by the Kendall Tau b correlation coefficient. If a statistically significant negative (decreasing) trend wasis observed, it waswill be interpreted with respect to whether the plant thermal discharge may be a contributing factor. Finding no significant trend over time or finding a significant increasing trend wasill be considered to statistically support a finding of "no prior appreciable harm".

Source: Appendix C in: Barbour et al. 1999 [35].
Note: exceptions in classification were taken when there was disagreement among one or more of the 7 references. regarding the trophic guild or tolerance of a species; the alternatives are shown.

No statistically significant negative (decreasing) trends were observed in yellow perch annual mean CPUE during the 1991–2002 period, supporting a finding of “no prior appreciable harm” due to Vermont Yankee’s existing (baseline) summer period permit limits. The time series of annual mean electrofishing CPUE from lower Vernon Pool exhibited a Kendall’s Tau b of 0.212 with a probability level of p=0.273333 (Figure 13-9Aa), while the electrofishing time series from the Vernon Dam tailrace area exhibited a Kendall’s Tau b of 0.0300 with a probability level of p=10.000891 (Figure 13-9bC). Kendall’s Tau b correlation coefficient for the annual mean trap net CPUE time series from lower Vernon Pool was 0.222 with a probability level of 0.404 (Figure 13-9Bc), and the correlation coefficient for the trap net data from the Vernon Dam tailrace was 0.56111 with a probability level of 0.835677 (Figure 13-9dD).

The thermally influenced portion of lower Vernon Pool of the Connecticut River (located from Vermont Yankee’s discharge weir downstream to Vernon Dam) is represented by 324 acres of bottom habitat and 0.19366 billion cubic feet of volume out of a total of 2,481 acres and 1.3814 billion cubic feet of volume contained in the entire Vernon Pool between Vernon Dam upstream to the foot of Bellows Falls Dam. Based on the two limiting or exclusionary thermal effects threshold temperatures cited in Table 13-6 for yellow perch (maximum and avoidance), and the predicted plume temperature contours for the average case and extreme case occurrence of flow and temperature (Figure 13-7 and Figure 13-8), the increase in river water temperature due to the new permit limits would exclude yellow perch from using between 0 zero and 9 nine acres of existing benthic habitat (0.0% to 2.7% of 324 acres) under the proposed new Vermont Yankee thermal limits that they presently have access to under the existing permit limits (Table 13-9). No habitat exclusion is predicted for the maximum survival temperature with either modeling scenario because the thermal plume never reaches 90°F near the river bottom. The excluded 9 nine acres of bottom habitat is predicted to occur for the avoidance temperature of 83°F modeled under the extreme case (1% occurrence) low flow and upstream temperature conditions, and is located near and immediately downstream from the plant discharge weir on the west side of lower Vernon Pool. When put in perspective with the entire Vernon Pool, 9 nine acres of bottom habitat represents 0.4% of the total aquatic habitat area available.

The exclusion of yellow perch from up to 9 nine acres of benthic habitat in lower Vernon Pool describes the spatial extent of the predicted impact for the extreme (1% occurrence) case with respect to the avoidance temperature for yellow perch (Table 13-9), but the temporal aspect during which the exclusion is predicted to occur should also be considered to fully understand the extent of the predicted impacts. During one percent of the summer period (36 hours), 9 nine additional acres of habitat will be warmer than the reported avoidance temperature under the proposed new thermal limits compared to the existing limits. The diel cycle of solar insolation limited the occurrence of these 36 hours to a few consecutive hours in late afternoon of the warmest days, and not in one continuous block of time. For the average case summer period conditions modeled, which are predicted to occur 50% of the time or 1,824 hours, there is no increase (0.0%) in the extent of the thermal plume area above 83°F because the entire plume never reaches 83°F for both the proposed new limits and for the existing conditions. It should be noted that Station 7, upstream from Vermont Yankee’s discharge, is never at or above 80°F during the summer period (Table 13-4), so Vermont Yankee’s thermal discharge never reaches the avoidance temperature of 83°F under the existing permit delta T (+2°F) or under the proposed new summer period delta–T (+3°F).

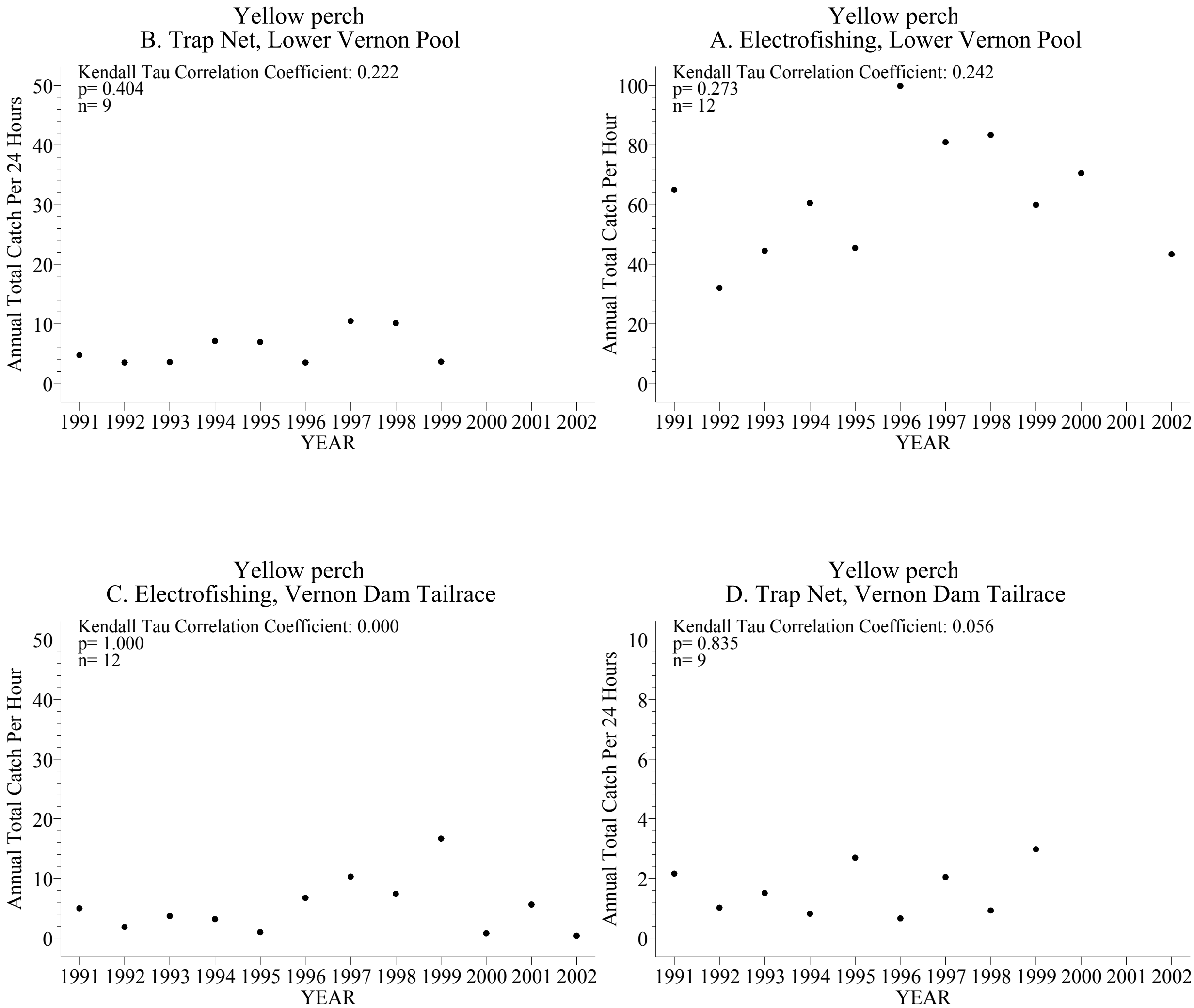


Figure 13-9
Scatter Plots Comparing Yellow Perch Annual Mean Catch per Hour for Electrofishing and Annual Mean Catch per 24 Hours for Trap Nets During 1991 Through 2002 in Lower Vernon Pool and the Vernon Dam Tailrace of the Connecticut River near Vernon, Vermont

With respect to the indicator thermal effects parameters for yellow perch, there is no meaningful change in the predicted habitat volume or bottom area under the proposed new summer temperature limits compared to the existing discharge conditions for both average case and extreme case conditions (Table 13-9). Yellow perch is considered to be a relatively thermally sensitive RIS with respect to its indicator thermal effects parameters. The optimum temperature for growth is reported as 74°F, the preferred temperature is 77°F, the mid–range of the reported spawning temperature is about 50ºF, and the mid–to upper incubation temperature for egg and larval development is 65°F (Table 13-6, Table 13-9). The spawning indicator temperature for yellow perch is naturally exceeded in Vernon Pool prior to 16 May, and the plume is presently at or above the early life history indicator temperature of 65ºF during most of the summer period under both the existing and proposed new thermal discharge limits, therefore no change is predicted for either indicator thermal effects parameter under the conditions modeled. The predicted changes in habitat exposure with respect to the preferred indicator temperature in the thermally effected portion of lower Vernon Pool are relatively small in plume volume (–4.2% or about –817 $ft^3 * 10^4$) and in plume area (–2.9% or –9 acres) compared to the total available habitat (19,366 $ft^3 * 10^4$ or 324 acres) in lower Vernon Pool. The predicted changes in habitat exposure to the indicator temperature for optimum growth in the thermally effected portion of lower Vernon Pool (Table 13-9) due to the proposed new thermal limits occur for the average case modeling scenario, and are relatively small in both plume volume (–3.8% or about 728 $ft^3 * 10^4$) and plume area (–4.8% or 16 acres) compared to the total available habitat (19,366 $ft^3 * 10^4$ or 324 acres) in lower Vernon Pool. Therefore, the thermal plume affects only a small portion of the habitat because the highest plume temperatures typically occur at the surface near the Vermont Yankee discharge weir, habitat not particularly favored by yellow perch.

There is no predicted increase in the time the mixed Connecticut River water in the Vernon Dam tailrace down to Station 3 will be at or above the maximum temperature for summer survival (UILT) for yellow perch under the proposed new thermal discharge limits (Table 13-9), because this water temperature is never reached. The avoidance temperature of 83°F is exceeded during 65 more hours or 1.8% more of the summer period time under the proposed new thermal limits compared to the existing limits. It is likely that yellow perch will shift their distribution in the Vernon Dam tailrace to avoid being there during the hours when the water temperature is predicted to exceed 83°F. With respect to the indicator temperatures, the optimum temperature for growth is predicted to be exceeded for 241 hours or 6.6% more of the time under the proposed new limits compared to the existing limits, the predicted increase in time at or above the preferred temperature is 10.9% or 397 hours, there is no change predicted for the time at or above the spawning temperature because this temperature occurs before the summer period, and the predicted increase in time the Connecticut River water at Station 3 will be at or above the incubation and larval development temperature under the proposed new discharge limits is 4.6% or 167 hours compared to the existing limits Table 13-9).

There is little potential under the proposed new temperature limits for the Vermont Yankee thermal plume to adversely affect the spawning of yellow perch, since spawning occurs in mid–April through mid–May, a period prior to the onset of summer permit limits when water temperatures are low and Connecticut River flows are generally high. In fact, no difference was calculated in the available plume volume or bottom area under either average or extreme case thermal plume conditions for yellow perch spawning or early life history (Table 13-9). In the Vernon Dam tailrace area, the indicator thermal effects temperature of 65ºF for incubation and larval development would be exceeded 4.6% more of the time (167 hours) under the proposed new permit limits compared to the existing limits, however tailrace habitat it is not a preferred spawning habitat. Yellow perch typically spawn in Vernon Pool from late–April through early–May, as evident by the first appearance of their larvae in the Vermont Yankee nearfield ichthyoplankton collections in lower Vernon Pool between 1 May (the start of permit–required sampling) and 21 May, depending on the year. Furthermore, the eggs are laid in a semi–buoyant mass or string that is deposited on the river bottom, and becomes attached to the substrate or vegetation [12]. Although wind and wave action or strong currents can dislodge the egg mass, the demersal and semi–adhesive nature of the mass would serve to limit their exposure to potential thermal impact from contact with the warmest portion of the surface plume in lower Vernon Pool.

Table 13-9
Comparison of Predicted Habitat change in Vernon Pool of the Connecticut River for Yellow Perch Between the Existing and the Proposed New Summer Permit Limits

A. Percent Difference						
Thermal Effects Parameter	**Temp (°F)**	**Average (50%) Case**	**Extreme (1%) Case**	**Average (50%) Case**	**Extreme (1%) Case**	**Increase in % Time[1] Tailrace Water is at or above Temp °F**
Exclusionary Temperatures		**Change in % Plume Volume ≥ Temperature**		**Change in % Bottom Area ≥ Temperature**		
Max. for summer survival, or UILT	90	0.0	0.0	0.0	0.0	0.0
Avoidance	83	0.0	–3.8	0.0	–2.7	1.8
Indicator Temperatures		**Change in % Plume Volume ≥ Temperature**		**Change in % Bottom Area ≥ Temperature**		
Optimum for growth	74	–3.8	0.0	–4.8	0.0	6.6
Preferred	77	–4.2	0.0	–2.9	0.0	10.9
Spawning	50	0.0	0.0	0.0	0.0	0.0
Early life history	65	0.0	0.0	0.0	0.0	4.6

B. Numeric Difference						
Thermal Effects Parameter	**Temp (°F)**	**Average (50%) Case**	**Extreme (1%) Case**	**Average (50%) Case**	**Extreme (1%) Case**	**Increase in Hours[2] Tailrace Water is at or above Temp °F**
Exclusionary Temperatures		**Change in Plume Volume ($ft^3 * 10^4$) ≥ Temperature**		**Change in Bottom Area (acres) ≥ Temperature**		
Max. for summer	90	0	0	0	0	0
survival, or UILT						
Avoidance	83	0	–731	0	–9	65
Indicator Temperatures		**Change in Plume Volume ($ft^3 * 10^4$) ≥ Temperature**		**Change in Bottom Area (acres) ≥ Temperature**		
Optimum for growth	74	–728	0	–16	0	241
Preferred	77	–817	0	–9	0	397
Spawning	50	0	0	0	0	0
Early life history	65	0	0	0	0	167

[1]Increase in % time = Station 3 proposed % exceedance–Station 3 existing % exceedance
[2]Increase in hours = increase in % time * 3648 hours in summer period of 16 May through 14 October

In conclusion, the proposed new summer temperature limits for the Vermont Yankee thermal plume are expected to have no adverse effects on the lentic insectivore trophic guild of intermediate tolerant members of the fish community that are represented by yellow perch in lower Vernon Pool and the Vernon Dam tailrace waters of the Connecticut River.

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14
THE 316(A) RENEWAL PROCESS: FULL THERMAL DEMONSTRATIONS MAY NOT BE NECESSARY

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Introduction

Over the last three years Allegheny Energy's coal-fired generating stations in three states (Pennsylvania, Maryland, West Virginia) were scheduled to renew their National Pollutant Discharge Elimination System (NPDES) Permits. During the application process the State Environmental Agencies where these facilities were located notified us that additional studies would be required to justify continuing each facility's 316(a) thermal variance. Allegheny entered into negotiations with the Agencies to discuss the site-specific circumstances of each facility and determine what the Agency would need to justify a continuance of the facility's variance. Through these negotiations, Allegheny's experience for complying with 316(a) variance renewal requirements varied from quick, simple, and inexpensive paper studies to complex, lengthy, and expensive demonstration studies. Three case studies are presented here to substantiate our position that, regardless of regulatory requirements, either level of study will show that conditions of the initial 316(a) studies are substantially similar to current conditions and the original 316(a) study conclusions are still valid. Therefore, we recommend that the lower effort paper study should be used initially to compare past and present conditions. Only when present conditions have significantly changed should full demonstrations be conducted.

Different Stations/Different Requirements

The case studies to be discussed represent three distinctively different sets of environmental settings. The R. Paul Smith Power Station in Maryland is a small station with once-through cooling and is located on a small river; the Willow Island Power Station in West Virginia is also a small station with once-through cooling, but is located on a large river; and the Harrison Power Station in West Virginia is a large station with cooling towers and is located on a small river. For each of these stations special conditions were added to their reissued NPDES permits requiring "thermal studies" to justify a continuance of the facility's 316(a) thermal variance. However, there is little consistency to the language stating the objectives of the required studies and the methods to be employed. Allegheny negotiated with state agencies to establish specific study requirements in each case, yet, the permit conditions for each remained distinct.

R. Paul Smith Power Station

The R. Paul Smith Power Station is a 116 MW coal-fired plant located along the Potomac River in Maryland. The average once-through cooling water discharge rate is 47 MGD with an average temperature differential of 18.5°F. The permit language reads as follows:

- "Within six months after the effective date of the permit, the permittee shall submit to the Department for approval a study plan and schedule for determining compliance with state water quality standards for thermal discharges for Outfall 001. The study plan shall be implemented and the final results provided to the Department no later 24 months after approval of the study plan by the Department.
- In lieu of submitting a study plan, the permittee may submit a re-evaluation of the existing study no later than six months after the effective date of the permit. If approved, the Department shall rescind the requirement to submit a study plan and perform a new study."

Allegheny elected to re-evaluate the 1981 316(a) thermal study. The approach was to compare critical evaluation parameters of the 1981 report with existing conditions. If critical parameters had not changed, then the 1981 evaluation would be considered valid. Data sources included Allegheny Energy and the Maryland Department of Natural Resources. The findings of the study were that the cooling system operating parameters had not substantially changed, receiving water flow conditions were within the range used for the 1981 study, and recent biological data supported the finding of the 1981 report that thermal discharges do not significantly affect periphyton, benthic macroinvertebrates and finfish.

State agencies accepted this study after minor revisions consisting primarily of the addition of data from a report not included in the submittal.

Willow Island Power Station

The Willow Island Power Station, located on the Ohio River in West Virginia, consists of two coal-fired units rated at 55 MW and 188 MW. The maximum design discharge rate is 203 MGD at a temperature differential of about 16.7°F. The special condition in the reissued permit read:

"As justification for the continuance of the Station's Clean Water At (CWA), Section 316(a) Thermal Variance, Allegheny Energy shall provide, within nine months of the NPDES permit's effective date, current data to verify the following:

a. There are no substantive changes to the Station's cooling water discharges, or other discharges in the plant site area which could significantly alter the previous 316(a) determinations, and

b. There are no changes in the biotic community of the receiving stream (based on existing available data) which would significantly impact the previous 316(a) determination.

c. In the event that any of the above conditions have changed, or continuance of the variance cannot be adequately justified, a plan of study and schedule of implementation shall be submitted to the State for approval within 120-days of determining that additional studies are required."

Allegheny developed a strategy to conduct a paper study to demonstrate conditions "a" and "b" by comparing current conditions with those at the time of the previous 316(a) determination studies in 1976. Data sources included information from Allegheny Energy, U.S. Army Corps of Engineers, Ohio River Valley Water Sanitation Commission, and U.S. Environmental Protection Agency. The submittal compared plant conditions of 2000–2002 with 1976, assessed Ohio River hydrology (1985–2003) and chemistry (1976–2001), took into account other discharges in the plant site area, and reviewed the benthic invertebrate (1971–1998) and fish (1968–1999) communities.

The study concluded that current thermal discharge characteristics are similar to those observed in 1976 and there have been no changes in the biotic community that would impact the previous 316(a) determination conclusion of no appreciable harm.

The study is still under review by the State Agencies. To date, they have not yet made their determination as to whether the submitted study provides sufficient justification to continue Willow Island's 316(a) thermal variance.

Harrison Power Station

The Harrison Power Station is comprised of three coal-fired units operating at a maximum of 640 MW each on the West Fork River in West Virginia. The maximum design discharge rate is 19.23 MGD. The permit special conditions required a full demonstration study as specified below:

"Permittee shall demonstrate that thermal discharges do not cause appreciable harm to a balanced indigenous aquatic life and wildlife in and on the receiving stream. Demonstration shall be [in] accordance with 40 CR 125, Subpart H. "Appreciable Harm" is damage to the balanced, indigenous community or to community components, which result in the following phenomena:

- Substantial increase in abundance or distribution of any nuisance species or heat tolerant community not representative of the highest community development achievable in receiving waters of comparable quality;
- Substantial decrease of formerly indigenous species, other than nuisance species;
- Changes in community structure to resemble a simpler successional stage than is natural for the locality and season in question;
- Unaesthetic appearance, odor or taste of the waters;
- Elimination of an established or potential economic or recreational use of the waters;
- Reduction of the successful completion of life cycles of indigenous species, including those of migratory species; or
- Substantial reduction of community heterogenisity of trophic structure.

The demonstration shall be based on the following:

- Development of thermal plume profiles and dissolved oxygen profiles.
- A survey of fishery resources to assess any adverse impacts of the thermal discharge.
- Quarterly macroinvertebrate surveys by placing Hester-Dendy multiplate samplers at a minimum of three monitoring locations (1) upstream of the thermal discharge (2) within the influence of the thermal discharge (3) downstream of the thermal discharge."

After negotiations with the state regulators regarding site-appropriate study methodologies, Allegheny implemented a one-year monitoring plan. New data were collected that included four seasonal macroinvertebrate surveys, two fisheries surveys, and in situ water quality and temperature measurements collected during each of the biotic sampling events. In addition, the dimensions of the thermal mixing zone associated with the station were modeled using the CORMIX thermal model. Comparisons with historical data were also made.

The findings of the year-long monitoring program were:

- The thermal discharge does not significantly affect water temperature or dissolved oxygen.
- The water quality is impacted by mine drainage.
- The macroinvertebrate community is stressed from mine drainage.
- The predominant fish species are intermediate in tolerance.
- The fish and macroinvertebrate communities are indicative of improving water quality conditions.
- The distribution of the existing biotic community does not show consistent trends that would indicate adverse impact on taxa richness or community composition from power station discharge.
- The thermal plume (delta Δ T – 5°F isotherm) is shallow, positively buoyant, does not contact the channel bottom, and may attach to the near shore.
- The thermal discharge is not expected to result in fish mortality nor affect passage of fish, nor will it significantly alter macroinvertebrate assemblages.

As a result of these findings, Allegheny was able to conclude the following:

- The existing aquatic community is impaired from mine drainage.
- There is no evidence the thermal discharge is adversely influencing the abundance, composition, or community structure of the aquatic community.
- Thermal discharge does not cause appreciable harm to a balanced indigenous community.

The State Agencies were satisfied that Harrison's study fulfilled the requirements of the special permit condition. They concurred with the study's conclusions and raised no objections with the renewal of Harrison's 316(a) thermal variance.

Conclusions Based On Our Experience

Regulatory agencies will place special conditions on the renewal of NPDES permits for facilities holding thermal variances that require evidence to support continuing the original thermal variances. As the case studies presented here illustrate, the formal regulatory language and specific requirements for permit renewal are not consistent. We have conducted thermal investigations for a variety of levels of effort and under a variety of discharge and receiving water conditions. In each instance, we found current conditions and conclusions to be substantially similar to the initial 316(a) studies. Agency approval hinges on that finding. Therefore, we support the strategy of initially conducting paper studies to compare present to past conditions. Full demonstrations should only be initiated when the paper study indicates that present conditions have changed significantly since the original 316(a) study was completed.

15
ECONOMIC AND ENVIRONMENTAL IMPACTS OF THERMAL DISCHARGE REQUIREMENTS

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Introduction

This paper summarizes the results of three studies conducted in the early 1990s by Argonne National Laboratory [1, 2, 3]. The studies were conducted for the U.S. Department Energy (DOE) in response to a bill proposed in the U.S. Senate that would have significantly modified the way in which large thermal discharges are regulated. The data were presented and published within the first few years after they were collected [4]. However, the data complemented the scope of the Electric Power Research Institute's (EPRI's) October 2003 workshop on issues relating to Section 316(a) of the Clean Water Act, and many of the workshop participants had not previously seen the data. At the request of EPRI staff, the data were presented again at that workshop. A brief description of the data and a related discussion are provided. Readers are encouraged to review the original three reports for greater details.

Background

Section 316(a) of the Clean Water Act (CWA) allows the states or the U.S. Environmental Protection Agency (EPA) (whichever has the authority to issue National Pollutant Discharge Elimination System [NPDES] permits) to establish alternative thermal limits if the discharger can demonstrate that the otherwise applicable thermal effluent limits are more stringent than necessary to protect the organisms in and on the receiving water body and that other, less stringent effluent limitations would protect those organisms.

In 1991, 250,466 MW were generated nationwide by electric power plants using once-through cooling systems. Once-through cooling systems withdraw water from a surface water body, use the water for cooling, then discharge the water back to the same or a nearby water body. About 75% of the generating capacity using once-through cooling systems operated under §316(a) variances [5]. Section 316(a) variances are renewed along with the facility's NPDES permit, approximately every five years.

In many cases, thermal discharge requirements for these plants are based on state water quality standards for temperature that must be met outside of designated mixing zones. Each state determines the size and shape of allowable mixing zones differently. Thermal mixing zones often were very large in size because discharges of both cooling water and heat from once-through cooled plants were huge. Typically, once-through plants are sited on large bodies of waters that could accommodate large mixing zones.

Overview of the Studies

In 1991, the U.S. Senate attempted to reauthorize the CWA through Senate Bill S. 1081. Among many other changes to the existing CWA, S. 1081 would have limited mixing zones (thermal and otherwise) to no more than 1,000 feet. In addition, the bill would have repealed Section 316(a) of the CWA, eliminating the specific provision allowing variances from thermal limits. Either one of these actions could have had a serious impact on the electric power industry. The U.S. Department of Energy (DOE) therefore requested Argonne National Laboratory to prepare some preliminary estimates of the magnitude of the impacts.

Argonne collected information from a sample of power plants in different parts of the country that use once-through cooling. In separate surveys, selected power companies were asked what each plant would do if a) it had to meet thermal limits within a 1,000-foot mixing zone, or b) the §316(a) variance were no longer available. The power companies also were asked to provide cost estimates, when available, for constructing new facilities and equipment to meet the changed requirements. Responses were received from 13 companies representing 79 plants for the mixing zone study and from 14 companies representing 38 plants for the §316(a) variance study.

The data from the power companies were used to develop capital cost rates in terms of dollars per kilowatt ($/kW). To estimate national capital costs, these cost rates were multiplied by the national affected capacity in megawatts (MW). The affected capacity was assumed to consist of those generating units that were currently using a §316(a) variance and those generating units that would be unable to meet limits based on a 1,000-foot mixing zone. This methodology assumes that the limited sample of plants providing data is proportionately representative of the nationwide power industry.

Capital Costs: §316(a) Study

Approximately 680 units would have been affected if the §316(a) variance had been lost. These units had a combined generating capacity of roughly 189,000 MW, which represented 33% of the total steam electric generating capacity in the United States. Of those 189,000 MW, approximately 43,000 MW were attributable to nuclear plants and approximately 146,000 MW were attributable to fossil-fuel plants [6].

The 14 power companies that provided information reported that they would retrofit cooling towers at nearly all of their 38 plants now operating under §316(a) variances. Reference 1 shows the capital cost estimates provided by the power companies to retrofit cooling towers as a function of power production capacity. Because costs for construction at a nuclear plant are nearly always higher than those for construction at a fossil-fuel plant, data were presented separately for the two fuel types. The reported cost rates ($/kW scaled to 1992 dollars) for fossil-fuel plants range from $32/kW to $346/kW, with an average of $108/kW for 31 plants. The cost rates for nuclear plants range from $102/kW to $234/kW, with an average of $171/kW for 7 plants [1]. Linear regression analysis was performed on the data. The resulting regression equations and correlation coefficients (r) are shown below.

Fossil-fuel plants	$y = 0.105x + 2.2$	$r = 0.77$	**Equation 15-1**
Nuclear plants	$y = 0.151x + 31.4$	$r = 0.53$	**Equation 15-2**

where y = millions of 1992 dollars and x = MW

National cooling tower retrofit costs were estimated by multiplying the appropriate cost rates by the affected capacity (146,000 MW for fossil-fuel plants and 43,000 MW for nuclear plants). For fossil-fuel plants, both the average fossil-fuel cost rate ($108/kW) and the slope of the fossil-fuel regression line ($105/kW) were used. Since the slope of the regression line for nuclear plants is not a reliable indicator of the data set, two other approaches—the average nuclear cost rate ($171/kW) and the median nuclear cost rate ($201/kW)—were used to develop the national retrofit cost for nuclear plants.

The results of this analysis show that if §316(a) of the CWA were eliminated and all plants currently operating under §316(a) variances were retrofitted with cooling towers, the estimated national capital cost would range from $15.3 billion to $15.8 billion for fossil-fuel plants and from $7.4 billion to $8.6 billion for nuclear plants. The combined total ranges from $22.7 billion to $24.4 billion in 1992 dollars [1].

A similar but separate study, using a different methodology, estimated the capital cost to the power industry of losing the §316(a) variance at $28.9 billion in 1992 dollars [6]. That estimate is based on two hypothetical plants, one fossil-fuel and one nuclear, with the costs scaled up to all affected plants. The relatively close agreement of the two estimates using mostly independent methodologies suggests that the estimates are at least in the right order of magnitude.

Capital Costs: Mixing Zone Study

The data collected from the 13 power companies indicate that 24 of the 79 plants for which data were provided may already be able to meet thermal standards within a 1,000-foot mixing zone. These plants represent 20,085 MW of capacity [4]. However, four of the plants are located in Wisconsin, which at the time reference 4 was published did not have any thermal water quality standards. The remaining 58 plants, representing 57,964 MW of capacity, would not be able to meet thermal standards within a 1,000-foot mixing zone and would have to find an alternative mode of operation. Operators of these 58 plants selected primarily two alternatives for compliance—cooling towers and diffusers (mechanical devices added to the end of discharge pipes to promote rapid mixing.) Diffusers would be added at six plants, cooling towers at 39 plants, and both diffusers and cooling towers at eight other plants. At two plants, helper towers (towers used to supplement once-through cooling systems) would be converted to full closed-cycle cooling. One company said it would consider either cooling towers or spray systems to enhance evaporation at three of its plants. One plant would be retired, and new replacement generating capacity would be constructed elsewhere. To simplify calculations, all sample data alternatives involving retrofitting cooling towers (adding cooling towers or a spray system, converting helper towers to full-time closed-cycle cooling, or adding cooling towers and diffusers) were combined with the one plant that would be retired, thereby creating a single category (52 plants). The six plants using just diffusers constitute a second category.

The cooling tower cost rates used to calculate a national cost estimate are at the higher end of the ranges from reference 1—$108/kW for fossil-fuel plants and $201/kW for nuclear plants. For diffusers, the capital cost rates from reference 7 were modified to equal $43/kW for fossil-fuel plants and $59/kW for nuclear plants. National estimates are based on consideration of the selected alternatives, the percentage of capacity anticipated to use each alternative, the estimated capacity (in MW) nationwide that would use each alternative, the cost rates, and the total capital costs. The estimated national capital cost for retrofitting plants that cannot meet thermal standards within a 1,000-foot mixing zone is up to $21.4 billion.

Costs Associated with the Energy Penalty

Retrofitting cooling towers or diffusers to existing power plants results in a reduction of plant output known as the energy penalty. The energy penalty is caused by increases in turbine back pressure that result in less efficient power generation, additional power requirements for pumping recycled water to the top of a natural-draft cooling tower or operating the fans at a mechanical-draft cooling tower, and increased pump head requirements due to the restricted flow through a diffuser. Power companies have several options for dealing with the energy penalty. They can operate the plant at lower net power output or, in some cases, they can run it more frequently or at a higher temperature. The latter option requires that additional fuel be burned to maintain output. In either case, there is an energy cost associated with retrofitting the cooling towers.

Cooling towers result in an energy penalty for fossil-fuel plants ranging from 1.1 to 4.6%, with most of the data falling between 1.5 and 2.5%. The cooling tower energy penalty for nuclear plants ranges from 1.0 and 5.8%, with the selected data falling between 2 to 3% [2]. The energy penalty from diffusers is 0.02% for fossil-fuel plants and 0.028% for nuclear plants [7].

The cost of compensating for the energy penalty has two components—the cost of generating replacement energy and the capital cost of building new generating capacity. The replacement energy cost is a function of the cost per kilowatt-hour, historical capacity factors, and the percent energy penalty. Annual energy costs ranged from $370 million to 670 million. The levelized 20-year energy costs ranged from $11.4 to 18.4 billion for loss of the §316(a) variance and $10 to 16.2 billion for reducing mixing zones to 1,000 feet.

In addition to the fuel costs for providing the extra energy, some power companies would need to construct new generating capacity. National cost estimates for this additional capacity range from $1.2 to $5.3 billion.

Total Economic Impacts

The total economic impact of the two changes proposed to the CWA's thermal discharge requirements are shown in Table 15-1. The total for losing the §316(a) variance ranges from $35.5 billion to $48.1 billion. The total for restriction of thermal mixing zones to 1,000 feet ranges from $32.6 billion to $42.2 billion. These are expressed in terms of 1992 dollars. For comparison to 2003 dollars using the Construction Cost Index, multiply those numbers by a factor of 1.36.

Table 15-1
National Estimate of Economic Impact of Two Changes in Thermal Discharge Policy
(Costs are expressed in billions of 1992 dollars)

Change in Policy	Capital Cost to Retrofit Cooling Tower	20-Year Fuel Cost	Capital Cost to Construct Replacement Capacity
Loss of 316(a) thermal variance	22.7–24.4	11.4–18.4	1.4–5.3
1,000-ft mixing zone	21.4	10–16.2	1.2–4.8

Sources: [1, 2, and 3].

Environmental Impacts

In addition to the direct costs associated with a 1,000-foot mixing zone limit, there are other secondary environmental impacts. The DOE made some preliminary calculations of the impact of a conservative 1% energy penalty on carbon dioxide emissions and evaporative loss. For each MW of generating capacity converted from once-through cooling to cooling towers, 46 tons per year of additional carbon dioxide would be emitted to the atmosphere and 15 gallons per minute of additional water would be evaporated [8]. If a higher percentage energy penalty is used (e.g., up to 5.8%, as noted above), the resulting impacts would be proportionally higher. This is estimated to increase carbon dioxide emissions by an estimated 8.3 million tons per year and, even more significantly, increase evaporation by about 2.7 million gallons per minute. Construction of new generating units would cause environmental impacts such as changes in land use, runoff characteristics, and wildlife habitat, and increased solid waste production.

Conclusions

The main conclusion of this paper is that whenever changes are proposed to national thermal discharge requirements, the impacts and costs can be far-reaching and quite significant. Given the survey approach used to estimate costs and the uncertainties surrounding the estimates, the projected costs should be considered as rough approximations. Regardless of whether actual costs would have been $20 billion, $40 billion, or higher, they are very large sums. Further, any proposed changes geared toward reducing water quality impacts should be carefully evaluated to ensure that they do not inadvertently introduce other types of environmental impacts that also have significant adverse effects.

Each Congress lasts for two years. Proposed bills that are not passed by both the House and Senate during that period are dropped. Whether as a result of compelling evidence included in the Argonne studies or presented to the Senate by other interested parties, or for some other reason, S. 1081 was never passed by the 102nd Congress. These or similar changes to the CWA have never been proposed again.

Acknowledgments

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16
THERMAL MODELING AND APPLICATIONS: FROM THE 1960S TO NOW (AND BEYOND)

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Introduction

Modeling of water temperature regimes in surface waters, also called thermal modeling, has been ongoing for at least 40 years. Models have been developed and applied to unidirectional flowing and tidally influenced rivers, small and large reservoirs, the Great Lakes, estuaries, and coastal systems. One impetus for the development of these models is the discharge of waste heat from thermoelectric power plants into surface waters, and the need to predict the extent of elevated temperatures and specified isotherms. While extensive monitoring programs were sometimes developed in attempts to map the temperature fields, it was realized that monitoring could not span the large range of climatological conditions and extreme events that would be of concern to scientists and regulators. Thus models became important predictive tools that addressed these concerns. The value of using models in a predictive capacity was documented many years ago in the 316(a) guidance manual [1]. This manual contains guidance on conducting studies to determine whether a 316(a) variance from water quality based standards or technology based limits would be warranted. The variances would be justified if a "balanced, indigenous community" would be maintained in the presence of the thermal discharge. In addition to thermal discharges, other factors were known to influence water temperatures, such as riparian shading, and models were soon developed to address these other issues as well [2].

Scope of this Paper

This paper is intended to provide an overview of thermal modeling over the past 40 years, and a glimpse of new thermally-related issues that may be on the horizon. Based on the many recent references in the literature on thermal modeling, interest in this subject, rather than diminishing, has in fact been growing over the years. A brief overview of thermal modeling that was performed in the 1960s to 1970s is given directly below. Then an overview of models developed in the past decade will be provided, followed by a review of recent thermal model applications. The paper ends with a look to the future of thermal modeling.

Early Developments in Thermal Models: 1960s to 1970s

Computer models that predict water temperatures were developed in the 1960s with the advent and availability of high-speed mainframe computers. Prior to electronic computers, more limited thermal modeling was performed using manual or nomographic techniques (e.g., [3]). Much of the early research on thermal modeling was conducted at eastern universities, such as MIT's Ralph Parsons Laboratory, Johns Hopkins University, and Vanderbilt University, or by organizations such as the Tennessee Valley Authority (TVA) or the U.S. Environmental Protection Agency's (EPA's) environmental research laboratories. Jirka, Abraham, and Harleman [4] developed some of the first techniques to evaluate waste heat discharges. Their analyses were applicable to both inland and coastal water bodies, as well as to submerged and surface discharges. At about the same time, Shirazi and Davis of the US EPA's Thermal Pollution Branch in Corvallis, Oregon, produced a workbook with nomographic solutions for the prediction of thermal plumes [5]. One key to the eventual success of generalized and accurate thermal models was the development of techniques to perform ambient heat budgets for the water bodies being simulated. In 1972, TVA [6] completed a series of heat exchange studies in reservoirs and, from those studies, developed heat budget algorithms for computer models. Those algorithms are still used today. By the late 1970s and early 1980s, computer modeling in general, and thermal modeling in particular, was rapidly becoming recognized as a valuable research and applied tool. New research institutions, such as the Electrical Power Research Institute (EPRI), which was formed in 1973, supported the development of thermal models. In 1979, for example, EPRI released a five-volume report (with associated software) designed to evaluate impacts of multiple power plants on a single water body [7].

Models Developed Over The Past Twenty Years

From the early 1980s to present, a number of organizations have sponsored the development of thermal (and other) models designed for a diversity of applications, and within the past few years several comprehensive model reviews have been completed. In this paper a summary is provided of representative thermal models applicable to different water body types. This summary is not intended to be comprehensive, but illustrative of the types of models in wide use today. Then an overview of recent reviews is provided, where over 150 models were examined.

Table 16-1 summarizes representative thermal models used for inland rivers and streams, small lakes and reservoirs, and watersheds. Also included are the sponsoring organization, and a summary of model technical specifications. Most, if not all of these models have been updated over the years, as the need for expanded capabilities became apparent. The model with the oldest lineage is HSPF, which was originally the Stanford Watershed Model developed in the late 1960s by Professor Ray Linsley and his graduate student Norman Crawford. At that time, the model was designed to route runoff from the San Francisquito Creek watershed near Stanford University. The USEPA supported the generalization of the model beginning in the 1970s and has continued further development to the present. For inland rivers (unidirectional in flow direction) traditionally one-dimensional models have been used. An exception is EPRI's RIVRISK model that can perform three-dimensional simulations. Several reservoir models are included in Table 16-1, with only one model being two-dimensional (CEQUAL-R2). That model can simulate temperature changes not only in the longitudinal direction (typically the flow direction), but also vertically within reservoirs. All the models in the table are presently supported by the sponsoring organizations shown, and URLs are provided as sources for more information about each model.

Table 16-1
Summary of Representative Thermal Models for Inland Rivers and Reservoirs

Model	Water Body	Sponsor	Dimensions	Hydro-dynamics	Time Step	Heat Budget	URL
CEQUAL-RIV1	River	USACE	1-D	Dynamic	Sub-daily	Both full heat budget and equilibrium method	http://www.wes.army.mil/el/elmodels/index.html#wgmodels
HSPF	River/Watershed	USGS/EPA	1-D	Channel Routing	Sub-daily	Full heat budget	http://www.epa.gov/ceampubl/hspf.htm
QUAL2E	River	EPA	1-D	Steady State	Sub-daily	Full heat budget	http://www.epa.gov/ceampubl/softdos.htm and http://www.epa.gov/OST/BASINS/bsnscocs.html
SNTEMP	River	USGS	1-D	Steady State	Daily to monthly	Equilibrium temperature method	http://www.mesc.usgs.gov/rsm/more_temp.html
WQRRS	River/Reservoir	USACE		Dynamic	Sub-daily	Linearized heat budget, or equilibrium temperature approach	http://www.wrc-hec.usace.army.mil/software/index.html
RIVRISK	River	EPRI	3-D	Daily Average flow rates	Daily	Equilibrium temperature method	Contact Dr. Bob Goldstein at rogoldst@epri.com
WARMF	Stream/Watershed	EPRI	1-D	Channel routing	Sub-daily	Heat budget	Contact Dr. Bob Goldstein at rogoldst@epri.com
CEQUAL-R1	Reservoir	USACE	1-D	Water Balance	Sub-daily	Full heat budget with some linearization	http://www.wes.army.mil/el/elmodels/index.html#wgmodels
CEQUAL-R2	Reservoir	USACE	2-D	Dynamic	Sub-daily	Equilibrium temperature method	http://www.wes.army.mil/el/elmodels/w2info.html

Table 16-1
Summary of Representative Thermal Models for Inland Rivers and Reservoirs (Continued)

Model	Documentation	Description
CEQUAL-RIV1	USACE. 1995. CEQUAL-RIV1: A dynamic, one-dimensional (longitudinal) water quality model for streams: User's Manual. Instructional Report EL-95-2, U.S. Army Corps of Engineer Waterways Experiment Station, Vicksburg, MS.	CE-QUAL-RIV1 is a longitudinal fully dynamic hydraulic and water quality simulation model intended for modeling unsteady stream flow conditions, such as associated with peaking hydroelectric tailwaters. The model also allows simulation of branched river systems with multiple control structures such as navigation locks and dams. The model has two parts, hydrodynamics and water quality. Temperature is a primary constituent that can be modeled, but other water quality constituents include, dissolved oxygen, biochemical oxygen demand, nitrogen and phosphorous species and transforms, coliform bacteria, dissolved iron and manganese, and the effects of algae and macrophytes.
HSPF	Bicknell, B.R., Imhoff, J.C., Kittle, J.L., Donigan, A.S., and Johanson, R.C., 2001. Hydrological Simulation Program – Fortran: User's Manual for version 12. U.S. Environmental Protection Agency, National Exposure Research Laboratory, Athens, GA. EPA/600/R-97/080, 831pp.	HSPF simulates the hydrologic, and associated water quality, processes on pervious and impervious surfaces and in streams and well-mixed impoundments for extended periods of time. HSPF uses continuous rainfall and other meteorological records to compute streamflow hydrographs and pollutographs. HSPF simulates interception soil moisture, surface runoff, interflow, base flow, snowpack depth and water content, snowmelt, evapotranspiration, ground-water recharge, temperature, as well as a host of other water quality parameters.
QUAL2E	Brown, L.C. and Barnwell, T.O. 1987. The Enhanced Stream Water Quality Models QUAL2E and QUAL2E-UNCAS: Documentation and User's Manual. EPA/600/3-87/007, EPA Environmental Research Laboratory. MAY (NTIS PB87-202-156).	QUAL2E permits simulation of several water quality constituents in a branching stream system using a finite difference solution to the one-dimensional advective-dispersive mass transport and reaction equation. It is intended as a water quality planning tool for developing total maximum daily loads (TMDLs) and can also be used in conjunction with field sampling for identifying the magnitude and quality characteristics of nonpoint sources.
SNTEMP	U.S. Fish and Wildlife Service. 1984. Instream Water Temperature Model. Instream Flow Information Paper: No. 16. In cooperation with the U.S. Soil Conservation Service. FWS/OBS-84/15. September	SNTEMP is a mechanistic, one-dimensional heat transport model that predicts the daily mean and maximum water temperatures as a function of stream distance and environmental heat flux. The heat transport model is based on the dynamic temperature-steady flow equation and assumes that all input data, including meteorological and hydrological variables, can be represented by 24-hour averages.
WQRRS	U.S. Army Corps of Engineers, Hydrologic Engineering Center (USACE-HEC). 1986. WQRRS Water Quality for River-Reservoir Systems, User's Manual. Hydrologic Engineering Center. October 1978, revised 1986.	The WQRRS package consists of three programs that interface with each other. The Stream Hydraulics Package and the Stream Water Quality programs simulate flow and quality conditions for stream networks that can include branching channels and islands. The Reservoir Water Quality program evaluates the vertical stratification of physical, chemical and biological parameters in a reservoir. The hydraulic computations can be performed optionally using input stage discharge relationships, hydrologic routing, kinematic routing, steady-flow equations, of the full unsteady flow St. Venant equations (finite element method).

Table 16-1
Summary of Representative Thermal Models for Inland Rivers and Reservoirs (Continued)

Model	Documentation	Description
RIVRISK	Lew, C.S. and W.B. Mills. 2000. User's Guide for RIVRISK Version 5.0: A Model to Assess Potential Human Health and Ecological Risks from Power Plant and Industrial Facility Releases to Rivers. EPRI, Palo Alto, CA. 2000. 1000733	RIVRISK is a modeling tool that provides evaluation of a power plant's effects on its receiving river. Releases addressed by RIVRISK include direct discharges, atmospheric emissions, groundwater seepage, and cooling water, including thermal recirculation. Both two-dimensional and three-dimensional simulations can be performed. Simulations are guided by an online database containing key parameters for 126 organic and inorganic chemicals common to power plant and industrial releases.
WARMF	Chen, C.W., Herr, J., Ziemelis, L., 1998. Watershed Analysis Risk Management Framework – A Decision Support System for Watershed Approach & TMDL Calculation. Documentation Report, EPRI, Palo Alto, CA. TR110709	WARMF is a dynamic watershed simulation model that calculates daily runoff, ground water flow, hydrology and water quality of river segments and stratified reservoirs. The data module contains meteorology, air quality, point source, reservoir release, and flow diversion data. It also contains a decision support system to assist in the consensus building process.
CEQUAL-R1	USACE. 1986. CEQUAL-R1: A Numerical One-Dimensional Model of Reservoir Water Quality, User's Manual. Instruction Report E-82-1, Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.	CEQUAL-R1 simulated the vertical distribution of thermal energy and chemical and biological materials in a reservoir through time. The models are used to study water quality problems and the effects of reservoir operations on water quality. Water quality conditions that can be addressed include prediction and analysis of thermal stratification, location of withdrawal ports required to meet downstream temperature objectives, analysis of storm events, upstream land use changes, or reservoir operational changes on in-pool release water quality.
CEQUAL-R2	USACE. 1995. CE-QUAL-W2: A Two- dimensional laterally averaged, hydrodynamic and water quality model, Version 2.0, User Manual. Instructional Report NE-86-5, Environmental Laboratory, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.	CEQUAL-W2 was developed for reservoirs but has also been applied to rivers and estuaries. The two-dimensional model determines the vertical and longitudinal distributions of thermal energy and selected biological and chemical materials in a system through time. The models provide capabilities for assessing the impact of reservoir design and operations on the water quality variables. The model determines in-pool water volumes, surface elevations, densities, vertical and longitudinal velocities, temperatures, constituent concentrations as well as downstream release concentrations. The unsteady hydrodynamic model accommodates variable density effects on the flow field. The water quality model simulated the dynamics of up to 20 constituents in addition to temperature.

Typically, complex three-dimensional thermal models have not been widely used on small to medium sized rivers or on small reservoirs. However, as regulatory and scientific interests and concerns focus more on the near-field around thermal discharges (i.e., the hydrodynamic and thermal mixing zones), there will be a need to simulate the three-dimensional thermal structure of the water, since plumes typically travel some distance downstream before they behave in a one-dimensional or two-dimensional manner. To simulate plumes in three-dimensions, sophisticated models have been developed during the past decade, and such model development still continues today. Typically, the three-dimensional models focus on complex natural systems, such as estuaries, coastal waters, or large lakes where the three-dimensional nature of the plume cannot be ignored. Such models can be used for thermal discharges on rivers or reservoirs as well when complex hydrodynamics must be simulated to accurately predict the plume's behavior (for example, to simulate plume recirculation). Table 16-2 provides a summary of numerous organizations that have developed three-dimensional hydrodynamic models that could be used to simulate thermal discharges. The table includes a worldwide perspective, since interest in applications of these models is quite varied. Note that the last three rows of that table are devoted to the more simplified near field plume models. Table 16-3 summarizes many three-dimensional hydrodynamic models capable of thermal simulations. The table includes both proprietary and public domain models. Over the past two decades, model developers have preferred to use finite-difference solution techniques rather than finite element techniques which were more widely used in the 1980s. This may be partly because finite difference codes now use curvilinear boundary fitted coordinate systems, which increases their ability to more accurately simulate spatially complex systems, and partially because of more efficient solution techniques.

Below, three recent model review reports are summarized. All the reviews are relevant to thermal models. The reports consist of the following:

- A review of over 150 water quality models, many of them thermal models, applicable to both surface and ground water [8].
- A review of thermal models, with emphasis on applications in the Central Valley of California [9].
- A review of models applicable to coastal water bodies [10]. Many of these models are also applicable to large lake applications, such as the Great Lakes.
- The WEF review evaluated models in the following classes: pollutant runoff, hydraulic or hydrodynamic models, receiving water models, and groundwater models. The documentation is contained on a CD, which also includes model evaluation software, so that users can easily assess the capabilities of specific models, or compare the capabilities of selected models. The second review [9] provides an overview of stream and reservoir water temperature modeling focused on the Central Valley of California. The report also covers theoretical modeling considerations, such as heat budget concepts, model data requirements, and model calibration/validation. This report also delineates the steps that would be executed in a thermal modeling study. The last of the three studies [10] provides an overview of models that have been recently developed for coastal circulation applications. All of the models account for the effect of temperature on water density, so that water temperature is one of the parameters always simulated. Coastal ocean applications also pose a unique problem of having an open-ocean boundary, where it may be difficult to accurately specify appropriate boundary conditions.

Table 16-2
Representative Organizations that Develop and Apply Hydrodynamic and Water Quality Models

Organization Name: Hydrodynamic/Water Quality Models	Type of Organization	Affiliated Organizations	General Description of Models	Contact
HydroQual	Consulting	WL Delft Hydraulics; Worldwide association of research groups	Circulation, water quality, thermal discharges, sediment transport	http://www.hydroqual.com/
Applied Science Associates	Consulting	University of Rhode Island	Circulation, water quality, thermal discharges, sediment transport, oil spills	http://www.appsci.com/
Dynalysis	Research	Princeton University	Circulation	http://www.dynalysis.com/
USACE WES	U.S. Army	RMA; Brigham Young	Circulation, water quality, storm surge	http://hlnet.wes.armyh.mil/software/tabs/
Danish Hydraulic Institute/VKI	Consulting and research	Danish Academy of Technical Sciences	Circulation, water quality	http://www.dhi.dk/
WL | Delft Hydraulics	Consulting and research	Hydroqual	Circulation, water quality, storm surge	http://www.wldelft.nl/
Resources Management Associates	Consulting	University of California at Davis	Circulation, water quality, sediment	jfdegeorge@rmanet.com
Tetra Tech, Inc.	Consulting	Virginia Institute of Marine Science	Circulation, water quality, sediment	ham@visi.net
NOAA	Government al agency	Works jointly with many agencies; Universities such as Rutgers, Princeton, Rhode Island, Maryland	Global circulation models; transport of biota; oil spill software; tsunamis, nowcasts/forecasts	http://www.pmel.noaa.gov/; http:chartmaker.ncd.noaa.gov/csdl/op/welcome .html
U.S. Coast Guard	Government al agency	Works jointly with many agencies, particularly NOAA	Spill response software	http://www.rdc.uscg.mil/

Table 16-2
Representative Organizations that Develop and Apply Hydrodynamic and Water Quality Models (Continued)

Organization Name: Hydrodynamic/Water Quality Models	Type of Organization	Affiliated Organizations	General Description of Models	Contact
Naval Research Laboratory	U.S. Navy	Works jointly with many agencies, particularly NOAA	Ocean circulation, nowcast/forecast, coastal seas, swell and coastal waves, sea ice	http://www7320.nrlssc.navy.mil/html
U.S. Geological Survey	Governmental agency	Works jointly with many agencies and universities (such as Stanford University)	Circulation models	http://sfports.wr.usgs.gov/sfports.html; http://crusty.er.usgs.gov/
Oregon State University	University	University of Miami	Circulation models	http://posum.oce.orst.edu/
University of Miami, FL	University	Office Naval Research, DOE, NSF	Circulation models	http://panoramix.rsmas.miami.edu/micom/
Dartmouth	College	Northestern Universities, USGS	Circulation	http://www-nml.dartmouth.edu/circmods/gom.html
Rutgers University	University	UCLA	Circulation	http://marine.rutgers.edu/
UCLA	University	Rutgers	Circulation, biogeochemical, particulate transport	http://www.ioe.ucla.edu/research/integrmodresearch.html
HR Wallingford	Consulting (United Kingdom)	Laboratoire National d'Hydraulique (France)	Circulation, water quality, sediment transport, wave dynamics	http://www.hrwallingford.co.uk
Athens GA Environmental Research Laboratory	U.S. EPA ORD laboratory	—	Visual Plumes (Beta Version)	http://www.epa.gov/AthensR/
Georgia Institute of Technology	University	—	Roberts-Synder-Baumgartner(RSB)	http://www.gatech.edu/
Oregon Graduate Institute	Research Institute	—	CORMIX	http://ese.ogi.edu/doneker.html

Table 16-3
Representative Three-Dimensional Hydrodynamic Models

Model Name	Reference	Affiliated Organization	URL	Solution Technique
POM	Blumberg and Mellor (1987)[11]	Princeton University	http://www.aos.princeton.edu/WWWPUBLIC/htdocs.pom/	Finite Difference
ECOM-si	See URL	HydroQual	http://crusty.er.usgs.gov/ecomsi.html	Finite Difference
DPOM	See URL	Dynalysis	http://www.dynalysis.com/	Finite Difference
MOM	Pacanowski (1996)[12], Bryan (1969)[13]	Geophysical Fluids Dynamics Lab, Princeton University, NOAA	http://www.gfdl.gov/MOM/MOM.htlm	Finite Difference
POP	McClean et al. (1997)[14]	Los Alamos National Lab	http://gnarly.lanl.gov/Pop/Pop.html	Finite Difference
TRIM3D	Gross (1997)[15]	U.S.G.S in Menlo Park, CA, or Vincenzo Casulli, University of Italy in Naples	casulli@science.unitn.it	Finite Difference
SEOM	Curchitser et al. (1996)[16]	Rutgers University	http://marine.rutgers.edu/po/models/index.html	Finite element/Spectral
SCRUM	Song and Haidvogel (1994)[17]	Rutgers University	http://marine.rutgers.edu/po/	Finite Difference
SPEM	Haidvogel et al. (1991)[18]	Rutgers University	http://marine.rutgers.edu/po/	Finite difference
ROMS	See URL and ROMS information	Rutgers University	http://marine.rutgers.edu/po/	Finite difference
ROMS-modified	See ROMS	Dr. Keith Stolzenbach, UCLA	http://www.ioe.ucla.edu/research/integrmodresearch.html	See ROMS

Table 16-3
Representative Three-Dimensional Hydrodynamic Models (Continued)

Model Name	Reference	Affiliated Organization	URL	Solution Technique
QUODDY	Lynch et al. (1996)[19]	Dartmouth College	http://www-nml.dartmouth.edu/Software/quoddy/quoddy4/Release_2.1/	Finite element
EFDC	Hamrick and Wu (1997)[20]; Hamrick and Mills (1999)[21]	Dr. John Hamrick, Tetra Tech	ham@visi.net	Finite Difference
RMA10/RMA11	King and DeGeorge (1995)[22]	RMA	jfdegeorge@rmanet.com	Finite element
TELEMAC-3D	See URL	HR Wallingford	http://www.hrwallingford.co.uk	Finite element
WQMAP	Spaulding et al. (1999)[23]	Applied Science Associates, University of Rhode Island	http://www.appsci.com/	Finite Difference
CH3D	Sheng (1989)[24], Johnson et al. (1991)[25]	U.S. Army Corps of Engineers, Waterways Experiment Station	http://hlnet.wes.armyh.mil/software/tabs/	Finite Difference
MIKE3	See URL	DHI Inc.	http://www.dhi.dk/	Finite Difference
Delft3D	See URL	WL | Delft Hydraulics and HydroQual	http://www.wldelft.nl/	Finite Difference
MICOM	Bleck (1998)[26]	University of Miami FL; Los Alamos National Lab	http://panoramix.rsmas.miami.edu/micom/	Finite Difference
Princeton West Coast Model (PWC)	See URL	Navy Research Laboratory, Stennis Space Center, MS	http://www7300.nrlssc.navy.mil/html/mel-home.html	Finite Difference
POSUM	See URL	Oregon State University	http://posum.oce.orst.edu/	Finite Difference

Recent Case Studies

Below, a number of thermal modeling case studies are summarized. All of the case studies are recent (within the past five years), and were chosen to reflect current topics of interest to the modeling community. For additional case studies, see the three modeling reports described above [8, 9, 10]. In addition, several of the companion papers in this report provide recent thermal modeling applications. Further, several of the papers presented in this report discuss 316(a) thermal modeling studies. Those papers are entitled:

- "Overview of CWA Section 316(a) Evaluations of Power Plants with Thermal Discharges in Maryland" where case studies of mixing zone analyses were performed on a large estuary, a small estuary, and a riverine facility. (Chapter 5)
- "PSEG 316(a) Study Experience at Salem and Hudson Generating Stations", where near field and far field models were used to predict thermal plume behavior. (Chapter 12)

Platt River Modeling Study [27]

A 128 km section of the Platt River downstream of two dams was modeled to determine the relationship between summer water temperatures and river flow rates. The stretch of the river studied is shown in Figure 16-1. The Kingsley and Keystone Diversion Dams are hydropower facilities. In the study reach downstream, over 500,000 sandhill cranes and millions of ducks and geese use the river as habitat, and feed on the fish in the river. Elevated water temperatures are of concern because the fish may be adversely affected which in turn would impact the bird population. The regulatory community believes that dam operations may cause severe habitat degradation; specifically that proposed operating alternatives for the dams would cause persistent exceedances of the water quality temperature standard of 32°C in the study area.

The one-dimensional model MNSTREM [28] was applied to the river. The model was calibrated using measured water temperature data for June and July of 1994, and weather data at two stations. The model was then verified by comparing the standard errors for the calibrated period to those of the verification period, which consisted of summertime data for four years. The standard errors were comparable for calibration and verification, and ranged from 0.8°C to 1.8°C. A sensitivity analysis was also performed on weather parameters. High sensitivities were found to the weather parameters solar radiation and air temperatures. For the alternatives analysis, MNSTREM simulated hourly temperatures under different flow regimes based on alternative dam releases and found that higher downstream temperatures were associated with lower flow rates. Heat exchange with the streambed was found to be an important factor in causing elevated daytime temperatures, especially during low flow conditions. Four different meteorological zones, representing an elevation change of about 300 m over the study reach, were used to account for the spatial variability in weather conditions. Model predictions typically ranged to within 1.4°C to 1.9°C of measured values, considered good by the investigators since the ambient temperature range was as high as 18°C during many summer days.

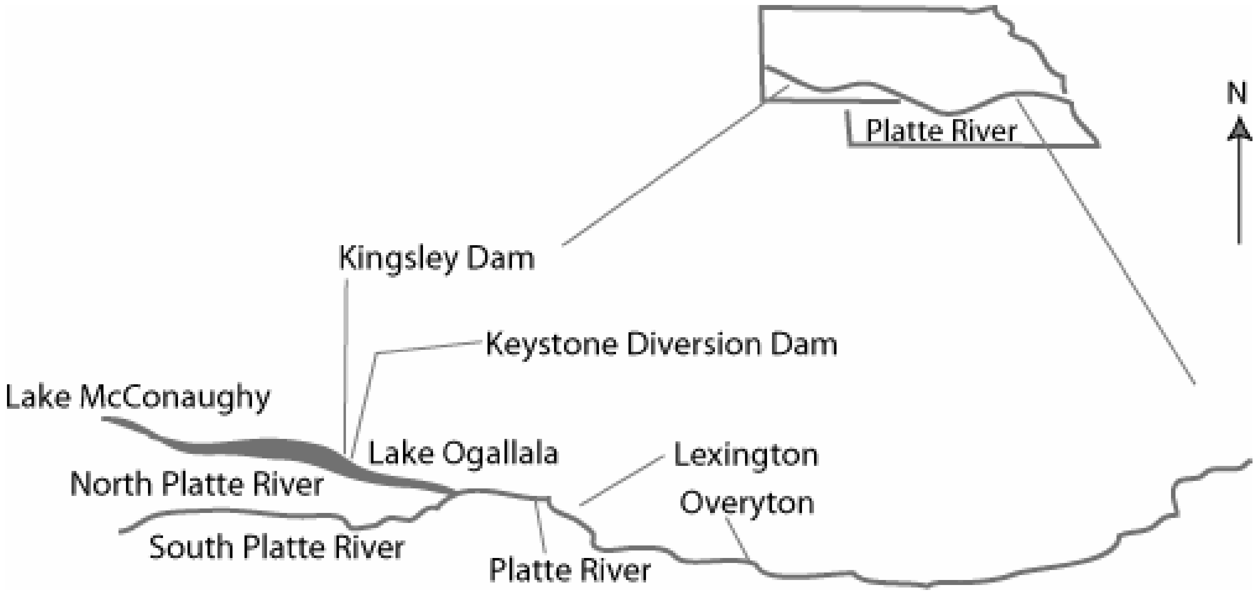


Figure 16-1
Spatial Extent of Water Temperature Modeling on the Platte River, NE

Columbia River Study [29]

Over 1240 km of the Columbia River system extending from the Canadian border to the Pacific Ocean was simulated to determine thermal TMDLs developed to prevent water temperatures from exceeding state water quality standards. The primary causative factors associated with the temperature exceedences were 15 dams and numerous point and nonpoint sources of thermal energy located in this length of river. The water quality standards on the river system are expressed as prescribed increases above natural temperatures (typically 0.14°C to 0.3°C above natural). Since the natural temperatures were defined as the water temperatures in the absence of the dams and other anthropogenic sources and since data on the natural temperatures had never been collected, a model was applied to the river system in the pre-industrial state to determine those temperatures. The computer model used was the River Basin Model 10 (RBM-10) [30] and is a one-dimensional dynamic model, similar to MNSTREM discussed above. When water temperatures under present conditions were compared to those under natural conditions, the temperature increment was as much as 6°C, well above the water quality standard (see Figure 16-2). The elevated temperatures were most evident below large dams, particularly in the fall of the year when water releases from the dams were warmer than the water otherwise would have been. To perform these predictions, the river was divided up into a series of 21 reaches, and each reach was subdivided into computational elements on the order of 2 km in length . RBM-10 was run with 30 years of meteorological data covering the period 1970 to 1999. The daily average temperatures at each site were averaged over the 30 years to estimate a long-term average temperature for each day of the year at each location along the river. By removing the dams, and repeating the simulations, the natural temperatures of the system were estimated.

Chattahoochee River Temperature TMDL [31]

The Georgia Environmental Protection Division identified a 10 mile (17 km) stretch of the Chattahoochee River near Atlanta GA that was not supporting its designated use due to exceedences of water quality standards. In particular, to comply with the water quality standards, if the river temperatures became elevated by more than 5°F above intake temperatures, the water quality standard is violated. As shown in Figure 16-3, several power plants that discharge near RM 300 caused water temperature to increase well above this limit on each for the three profiles shown that represent August 1971, October 1994, and January 1972 conditions.

The approach chosen to develop a thermal loading allocation scheme was to develop a heat budget upstream and down stream of the major discharges (the power plants). Heat fluxes from the other sources in the impaired reach were shown to be negligible in comparison. Thus, river modeling of downstream temperatures was not performed, and the heat budget approach is very straightforward and requires a relatively small amount of data. By selecting the critical meteorological, discharge, and flow conditions for use in the heat balance, and imposing a margin of safety, the allowable discharge temperatures from the major thermal sources were then calculated.

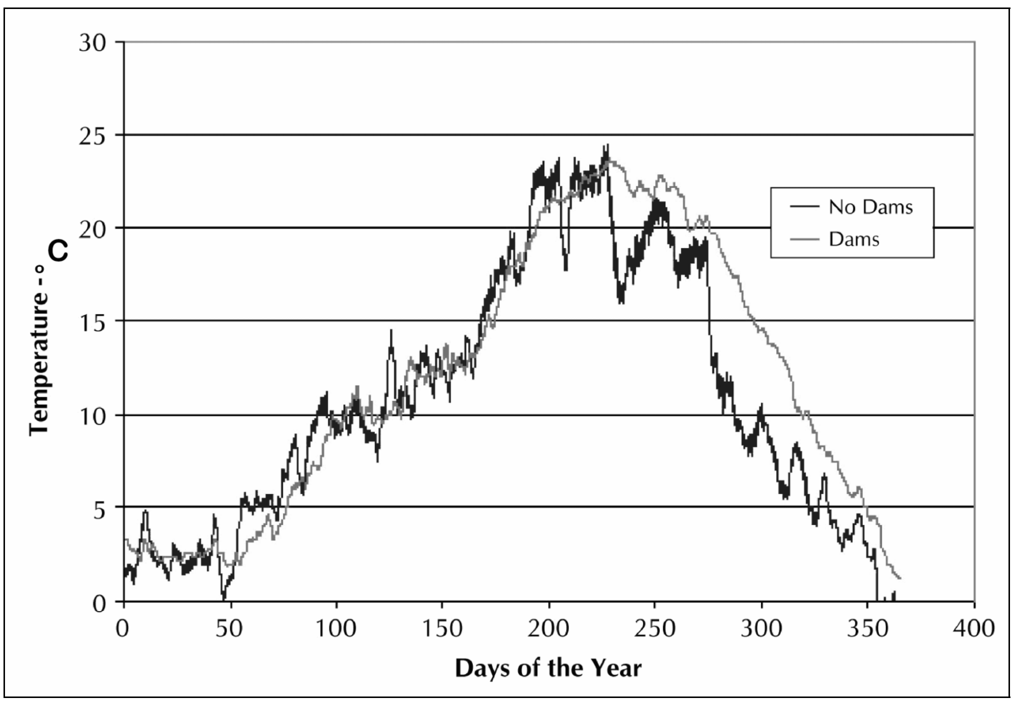


Figure 16-2
Comparison of Predicted Water Temperatures at Ice Harbor Dam on the Snake River with the Dams in Place (Current Conditions) and with the Dams Removed (Natural Conditions) During 1990

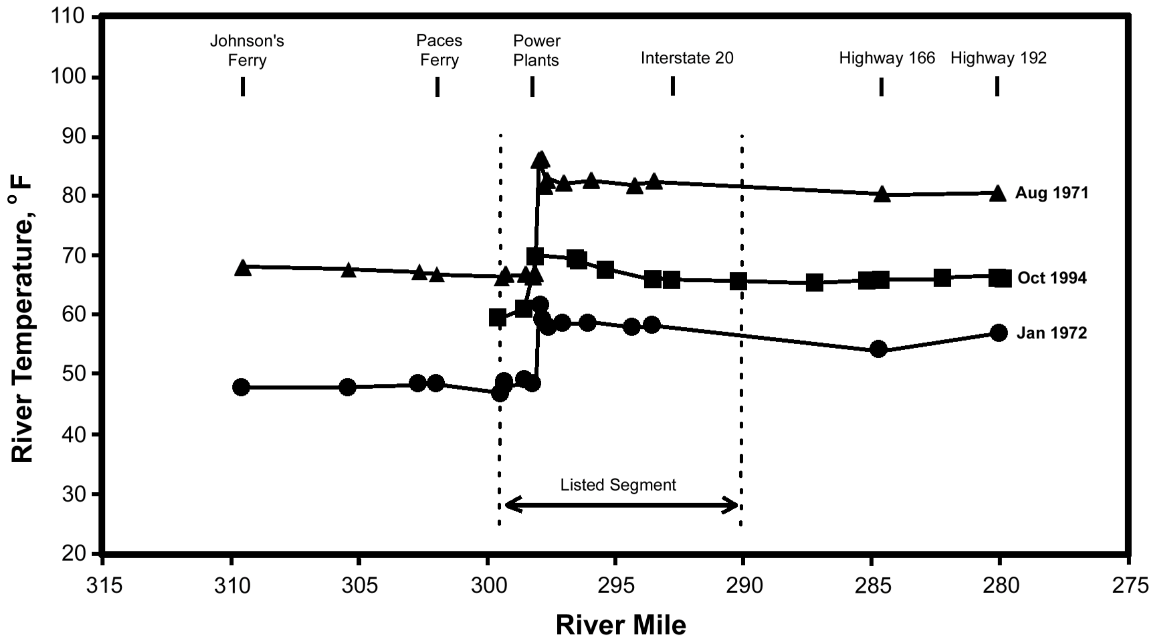


Figure 16-3
Three Water Temperature Profiles on the Chattahoochee River in Georgia

Three-Dimensional Analysis of Water Temperatures in Conowingo Pond, PA [21]

Both physical and numerical surface water hydrodynamic and transport models have been historically applied to predict power plant thermal impacts under design conditions. As the technology for numerical modeling matured in the 1980s and the cost of physical modeling increased, numerical models became the accepted tool for power plant thermal impact analysis. Because of historical computational limitations, numerical modeling analyses were often limited to thermal analysis of design conditions and separate models were used, somewhat independently, to analyze near- and far-field conditions. The current need to understand both thermal impacts and receiving water biogeochemical impacts and associated ecological and health risks under highly variable transient conditions on seasonal to annual time scales necessitates the use of predictive multi-dimensional modeling systems. Over the last decade, three-dimensional hydrodynamic and reactive transport modeling has matured from a research subject to a practical analysis technology. Simultaneously, computational requirements for realistic three-dimensional modeling have changed from super computers and high-end workstations to economical personal computers.

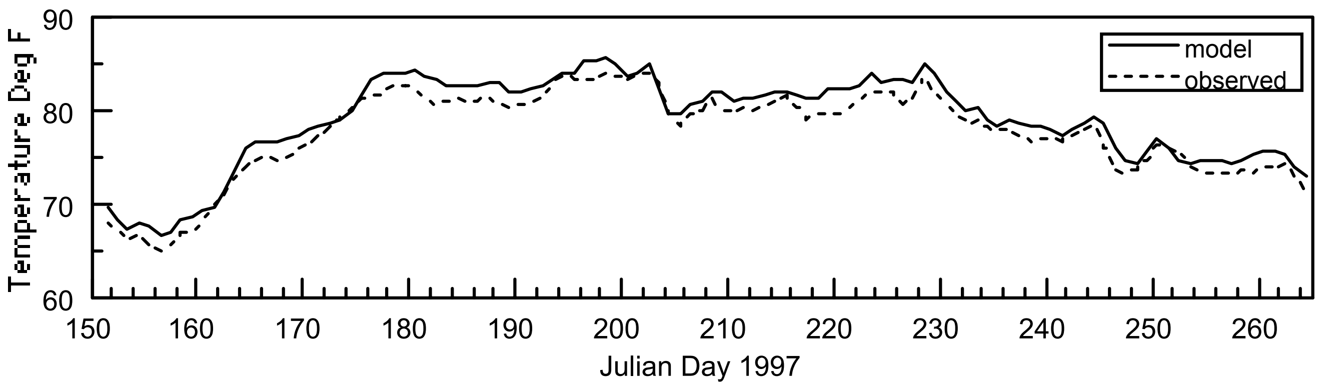
This application describes a three-dimensional surface water model system, the Environmental Fluid Dynamics Code (EFDC), capable of addressing a variety of power plant impact issues. The EFDC model was configured to simulate circulation and temperature in Conowingo Pond on the Susquehanna River in Pennsylvania and Maryland. Conowingo Pond serves as a cooling reservoir for the Peach Bottom Atomic Power Station (PBAPS). The pond, which contains about 240,000 acre-feet of water is approximately 23 km long and is bounded to the north and south by Holtwood and Conowingo Dams, respectively. Widths range from approximately 7 km with depths as great as 15 m upstream of the Conowingo Dam. A 145-day period, spanning from May through the middle of September 1997, was selected for model simulation. A curvilinear-orthogonal boundary fitted horizontal grid, containing 954 cells was developed to represent the pond. Horizontal grid resolution ranged from approximately 100 m in the vicinity of the PBAPS to 2 km near the upstream and downstream dams. Model simulations were conducted using eight layers in the vertical. Thermal forcing for the model included inflow temperature at Holtwood Dam, temperature rise through the PBAPS condenser, and atmospheric thermal exchange. Atmospheric data necessary for thermal simulation included air temperature, pressure, relative humidity, direct rainfall, and wind speed and direction.

The EFDC model, configured as described in the preceding section, was used to simulate thermal transport and the temperature distribution in Conowingo Pond during the summer of 1997. The model simulation began on 1 May, with a uniform initial temperature distribution of 57°F. The model simulated the three-dimensional temperature distribution throughout the reservoir at each grid point for each time step during the simulation. Figure 16-4a shows a comparison of observed and predicted intake temperatures over the period of simulation. Generally the temperature matched to within 2°F. Figure 16-4b shows predicted surface temperature isotherms at 4 pm on July 16, 1997. Upstream of the discharge, typical surface temperatures were between 87°F and 88°F. The influence of the thermal discharge is clearly seen by the downstream temperature that ranged between 89°F and 92°F. This three-dimensional model application clearly demonstrated that a complex numerical thermal model could be applied, with a reasonable effort, to accurately predict the three-dimensional temperature profile from a large thermal discharge into an impounded water body.

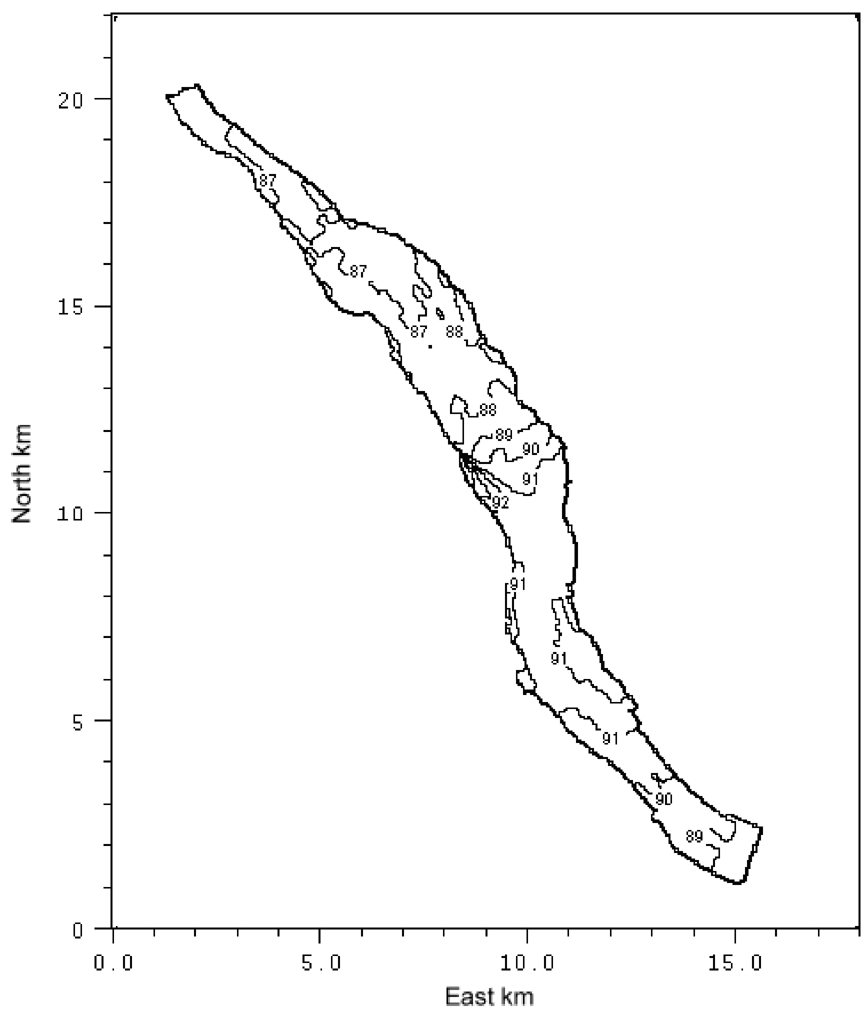
Use of Computational Fluid Dynamics (CFD) for Simulating Flow Fields Around Cooling Water Intake Structures

The natural hydrology of a water body, and its relationship to plant hydraulics, are important factors in evaluating the potential of a Cooling Water Intake Structure (CWIS) to entrain organisms. For an organism to become entrained, it must enter the hydraulic zone of influence (HZI) of a CWIS. Thus, while the proximity of a primary spawning and/or nursery area to a CWIS can be an important influence on the fraction of population potentially entrained for any individual species, other factors interact with proximity to determine actual susceptibility to entrainment. EPA acknowledges the importance of the HZI in its proposed Clean Water Act Section 316(b) and defines the HZI as “ that portion of the source water body hydraulically affected by the cooling water intake structure withdrawal of water.”

In the past, the HZI of CWIS have been inferred from the results of field sampling programs. Today, however, advanced hydraulic modeling techniques can be used to define the HZI of a CWIS using personal computers. One of six HZI case studies is presented here in.



a) Model predicted and observed temperatures at the PBAPS cooling intake.



b) Model predicted surface temperature on July 16, 1997 at 4:00pm.

Figure 16-4
Example Model Results on Conowingo Pond, PA

Chalk Point is located on the Patuxent River and withdraws cooling water from a shallow bay off the main channel (Figure 16-5). The bay from which cooling water is withdrawn is about 3 to 4.5 meters deep with the plant using about 31 m^3/s. Water approaches the plant through an intake canal and is returned to the Patuxent River through the discharge channel. The water is generally discharged upstream of the intake.

The flow direction in the Patuxent River changes with the tide; during low tide water flows out, and during high tide it flows inland. The water is a mix of the salty ocean water and fresh water from the Patuxent River. In shallow areas, such as the bay where the cooling water intake is located, the flow tends to be well mixed with little stratification. However, in the deeper parts of the main river, distinct density gradients were noted.

The flow field around the Chalk Point CWIS is affected by tidal forces, inflowing water from the Patuxent River, density variations due to salinity differences, and thermal stratification. The optimal CFD system for modeling the flow field was MIKE3. The formulation of MIKE3 includes all of the driving forces acting on the water body at Chalk Point. The model domain extends from approximately 2 miles downstream of Chalk Point to approximately 4 kilometers upstream of Chalk Point.

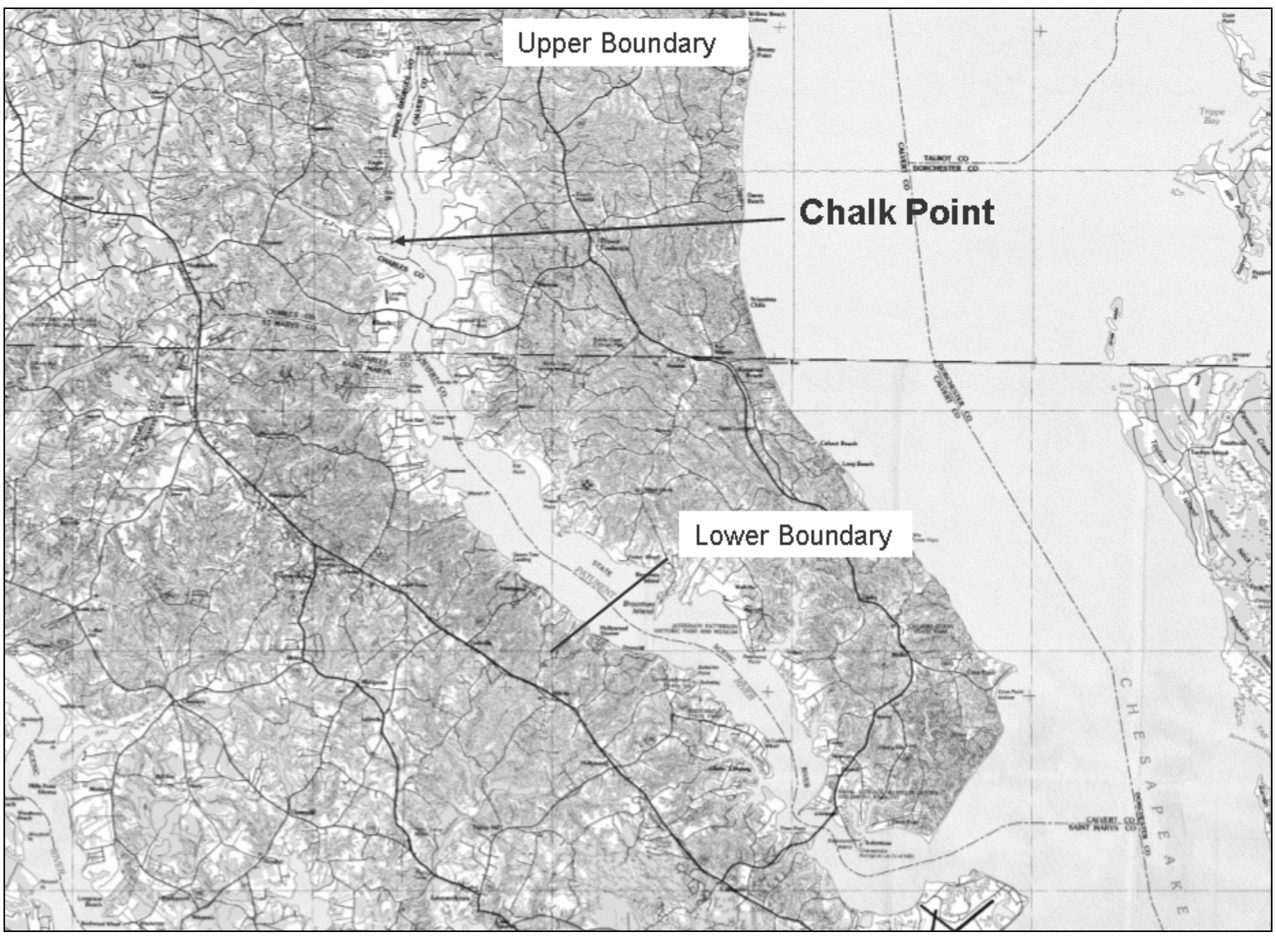


Figure 16-5
Site Map for Chalk Point Model

Given a working model which accurately reproduces the physical processes around the intake, particle tracking methods were used to determine the probability of entrainment for a passive particle at a specified location. Particles were released through out the flow field and then tracked to determine if they are entrained by the intake. A stochastic particle tracking model was used where the path followed by a particle is randomly determined. Particles are tracked for 24 hours at which time particles which were not entrained by the CWIS are typically still in the system and may be entrained later if the simulation were extended. Figure 16-6 shows the percentage of non-entrained particles released from discrete locations in the mesh.

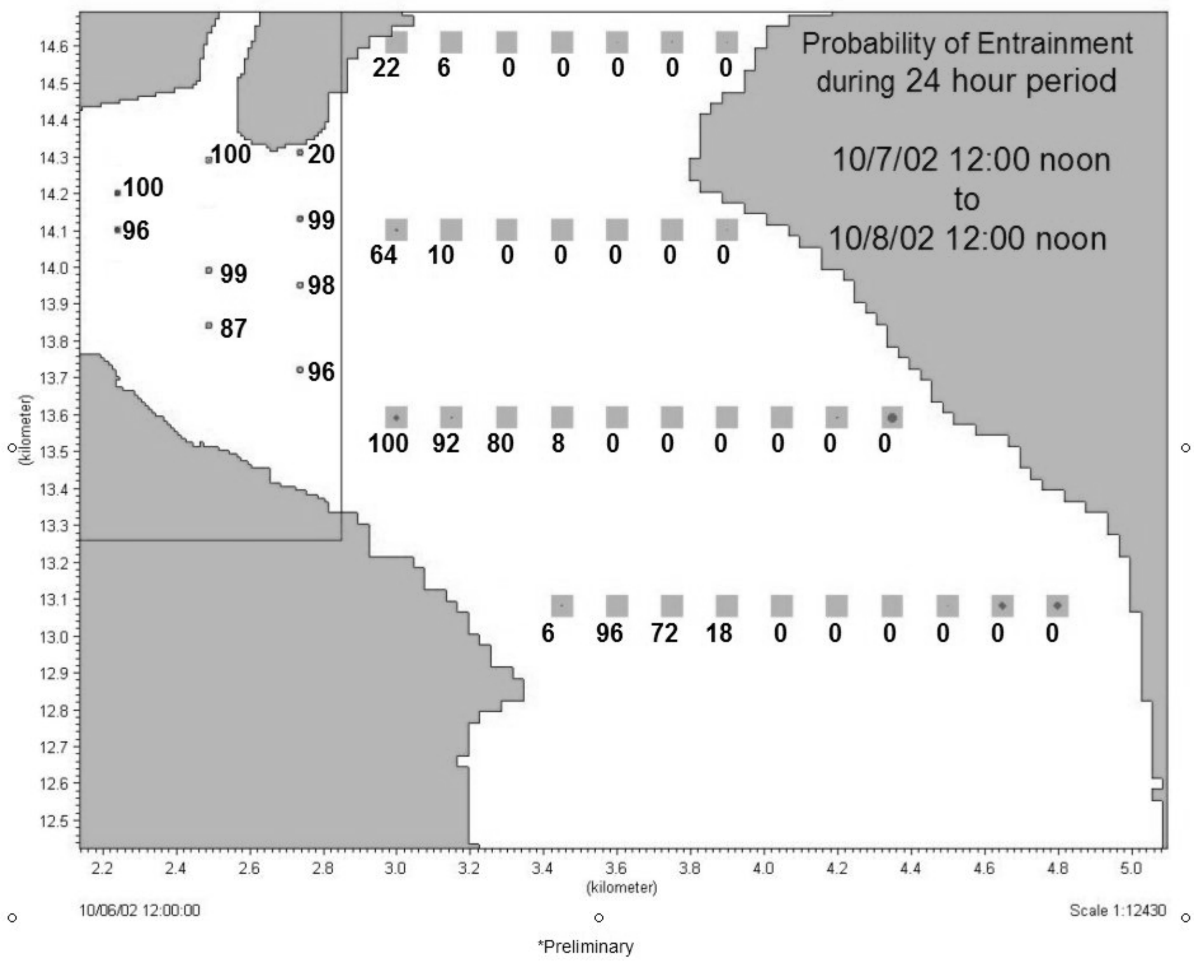


Figure 16-6
Example 24-Hour Entrainment Probability for Chalk Point

Similar case studies were performed at five other sites with varying water body types and modeling software. A more detailed description of the case study presented here and at the five other sites is given in the EPRI sponsored report Using Computational Fluid Dynamics Techniques to Define the Hydraulic Zone of Influence of Cooling Water Intake Structures (TR1005528).

The following general conclusions with regard to the abilities of CFD to identify the HZI of CWIS were reached as a result of this study:

- Commercially available CFD programs can be used to determine the HZI of CWIS located on lakes, reservoirs, rivers, tidally affected rivers, estuaries and open coastal locations. The software used in this study was robust, easy to use, and in all cases reliable.
- The commercially available CFD programs used in this study were developed around fully generalized solution algorithms that contain auxiliary physical models. As a result, the effects of physical and temporal factors that influence the HZI of CWIS can be included in simulations carried out with these products.
- A great strength of CFD relates to the degree by which study results can be visualized
- CFD computer programs are formulated in a way that enables near- and far-field flows to be modeled simultaneously. Therefore, a single simulation can be used to estimate the extent of the HZI of a CWIS. The same computer simulation can also be used to answer other hydraulic questions. For example, it can be used to estimate the extent of the thermal discharge plume.
- No single software package is applicable to all the flow conditions at various power plants. The HZI study used four commercial packages by three different vendors.

Fluent	Fluent Inc.	www.fluent.com
FLOW3D	Flow Science Inc.	www.flow3d.com
MIKE21	DHI Software	www.dhisoftware.com
MIKE3	DHI Software	www.dhisoftware.com

Upper Grande Ronde Watershed Thermal Modeling Application [32, 33]

In the Pacific Northwest, forest management activities have resulted in numerous environmental problems over the past decades, such as decline in anadromous fish populations. One reason cited for this decline is the elevated stream temperatures that result from removal of shade-producing riparian vegetation along fish-bearing streams. Current water quality standards for temperature in these Oregon streams state that the maximum temperature is not to exceed 17.8°C. However, observed water temperatures at many locations throughout the watershed range as high as 29.3°C. The water temperature data indicate that all stream reaches approach lethal or sublethal conditions for salmon populations. These high temperatures are thought to be caused primarily by lack of riparian vegetation.

To investigate how increased riparian shading might reduce stream temperatures in the watershed, HSFP was modified and applied to the watershed [32, 33]. Figure 16-7 shows the location of the watershed, along with the watershed segmentation and location of USGS stream gauging stations and meteorological stations in and around the watershed. In total, 51 subbasins were defined within the watershed.

A major modification required to HSPF for this application was the development of a method to dynamically calculate the incoming solar radiation that actually fell as the stream segments throughout the watershed. Such a calculation procedure was developed and included how riparian vegetation would influence the transmission of solar radiation to the surface of the stream. Simulations were performed continuously for two full years: 1991 and 1992. Stream temperature calibration was done for the summer of 1991, and validation for the summer of 1992. The calibrated temperatures tended to overestimate the summer maximum temperatures by about 2.6°C to 3.0°C. Similar results were obtained for the 1992 validation period. The model was then applied to multiple riparian vegetation buffer zone configurations. It was found that the riparian vegetation significantly reduced the number of reaches with lethal or sublethal temperatures. The results were especially dramatic when the canopy overhung stream segments. By adding riparian vegetation, the number of stream segments stressed by water temperatures was reduced from 51 to 7.

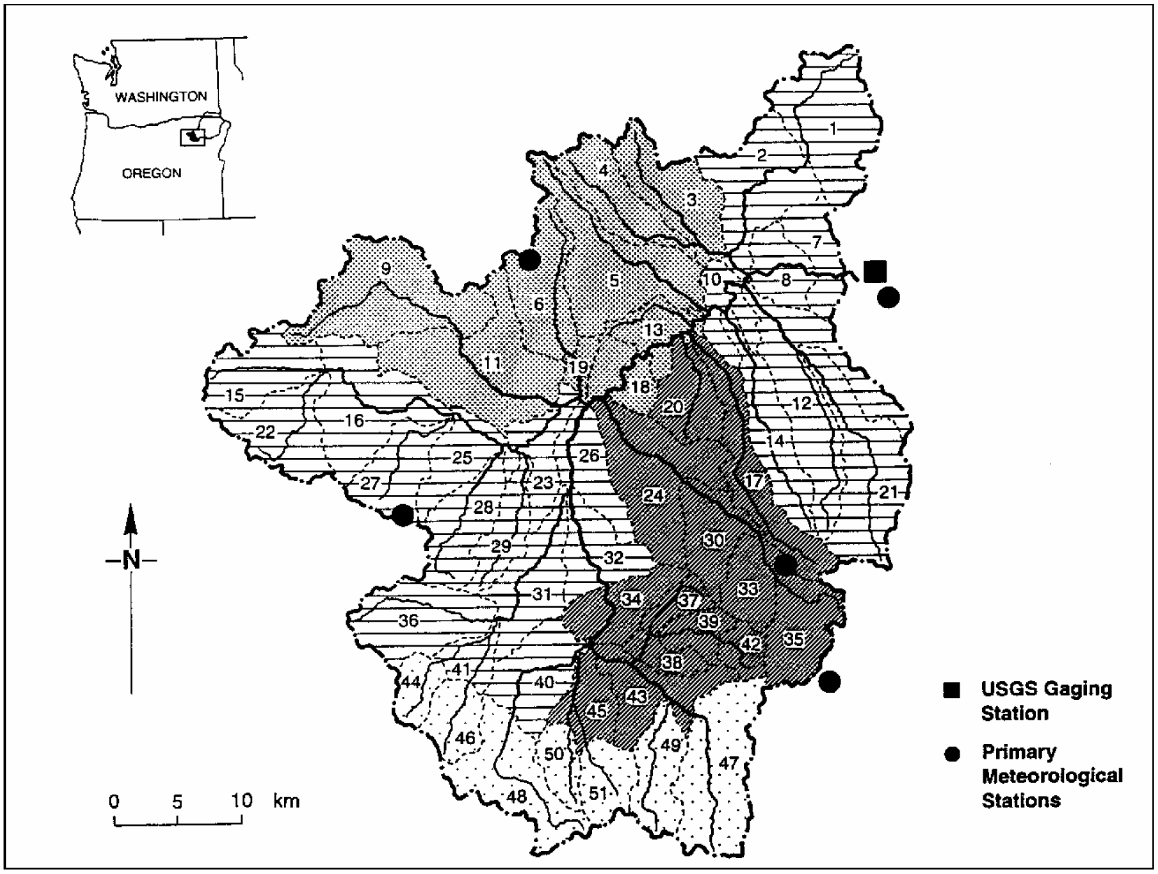


Figure 16-7
Watershed Segmentation and Locations of Meteorological and Stream Gauging Stations

Ongoing Thermal Modeling Advancements

Although thermal modeling of natural surface waters as we know it today began about 40 years ago there have been many advancements during that time and more advances are expected in the future. The increased computer resources (both speed and storage) have allowed modelers to develop more complex and realistic algorithms that can be solved on fine resolution computational grids. Gradually, more three-dimensional models have been developed and their use is spreading. Because of the complexity of the output generated by these models, a need grew to develop methods to display the results in ways that were easily understood. This helped to spawn the field of visualization, where three-dimensional modeling results can be displayed in a fashion that mimics the real world. This tool requires significant computational power, and is possible only because of today's faster computers. Table 16-4 provides information on visualization techniques appropriate for three-dimensional modeling.

Thermal modeling improvements in the past 10 years have focused on the more complex water bodies (coastal water, estuaries, large lakes). The newer models often use curvilinear boundary fitting coordinate systems that more realistically represent model boundaries. In streams and rivers, models that were developed two decades ago are still in use today, but often in a modified form to include important processes not initially included in the model.

More use is being made of remote sensing techniques than ever before to help validate thermal models. The technique called forward looking infrared radar (FLIR) can be used to accurately assess surface water temperature distribution over many kilometers daily. Such techniques can help to reveal fine structure in the temperature fields typically not predicted by models [34].

Potential Future Thermal Modeling Applications and Issues

More Rigorous Analysis of Overlapping Thermal Plumes

As electrical power requirements grow and if more thermal effluent is released into surface waters, the likelihood will increase that thermal plumes will reside in close proximity to each other, and may overlap. Accurate simulation of overlapping plumes will require more complex hydrodynamic models that can adequately deal with both far field and near field issues.
The near field around a thermal discharge is difficult to model accurately, due to the complex hydrodynamics that exist there. In the past, these problems have been addressed either by making overly simplistic assumptions about the behavior of the near field plume, or by using simplified near field plume models such as CORMIX, and linking model output to a far field model. While this may be a reasonable first-step, there is little data to show that such approaches accurately simulate near field plume behavior. One reason for the likely limited success of such an approach are the many simplifying assumptions made in these near field models, such as idealized channel configurations and simplified hydrodynamics, that may not realistically represent the prototype.

Table 16-4
Information on Visualization Techniques Appropriate for Three-dimensional Modeling

Organization	URL	Discussion
NOAA's Pacific Marine Environmental Laboratory	http://www.pmel.noaa.gov/visualization	Shows tsunamis, ocean vents, fish population dynamics, animations of water temperatures during El Nino and La Nina.
NOAA's Pacific Marine Environmental Laboratory for Virtual Reality	http://www.pmel.noaa.gov/vrml/3DViz.html	Shows visualizations similar to above.
Naval Research Laboratory Ocean Dynamics and Prediction and Visualization Lab	http://www7320.nrlssc.navy.mil/html/vislab-home.html	Examples include Sea of Japan and fly-by; site appears dated.
USGS Woods Hole Field Center	http://crusty.er.usgs.gov/omviz/	Shows tools used for POM, ECOM, and SCRUM.
USGS Woods Hole Field Center	http://crusty.er.usgs.gov/	Shows animations of Boston outfall plume; fly- bys.
Danish Hydraulic Institute	http://www.dhisoftware.com/	CD can be requested with software and animations.
Environmental Modeling and Research Lab, Brigham Young University	http://www.emrl.byu.edu/sms.htm	Tools used for visualization of many models, such as Corps of Engineers models.
Search engines on the internet	Check topic such as "advanced visualization software"	Many research organizations should be found.
University Corporation for Atmospheric Research	http://www.unidata.ucar.edu/packages/netcdf/	Most applications at present are for atmospheric research.
MATLAB (product of The Math Works)	http://www.mathworks.com	The URL is the home page for MATLAB. See the Woods Hole Field Center visualization URL for MATLAB applications.
Advanced Visualization Systems	http://www.avs.com/	AVS software is used as the cornerstone of many visualization systems.

Climate Change Applications

One new potential application of thermal models is to climate change issues. Over this century, the climate has been projected to continue warming, as it has during the past 100 years, due to such factors as increased carbon dioxide concentrations in the atmosphere (Figure 16-8). Not only will air and water temperatures increase, but surface water acidity may also increase since carbon dioxide is a weak acid. Also, surface water flow rates are expected to respond both to changing precipitation and warmer temperatures that should enhance evapotranspiration. All of these changes can potentially act as combined stressors on aquatic ecosystems, and affect balanced indigenous communities. A number of researchers have begun to examine projected stream temperature changes that result from global warming. One such effort was completed by Mohseni et al. [35] who predicted stream temperature responses across the United States. They chose a scenario where atmospheric carbon dioxide levels were doubled from present day values, and a new equilibrium climate condition was assumed to be attained. While it is not certain how atmospheric carbon dioxide concentrations will change over time, a doubling of the concentrations (2 x CO_2) is near a middle-of-the-road scenario. Weekly stream temperatures for the 2 x CO_2 scenario were predicted based on weekly air temperature data from 166 weather stations, where the air temperatures were incremented based on the climate change model results.

Of the 803 stream locations examined, temperatures were projected to increase at 764 of them. Across the United States, mean annual stream temperatures were predicted to increase by 2°C to 5°C, with higher increases near the central states, and lower increases on the West Coast. These temperature increases would have the most impact on aquatic organisms at locations where such species are now experiencing temperatures near the upper end of their thermal tolerance limits.

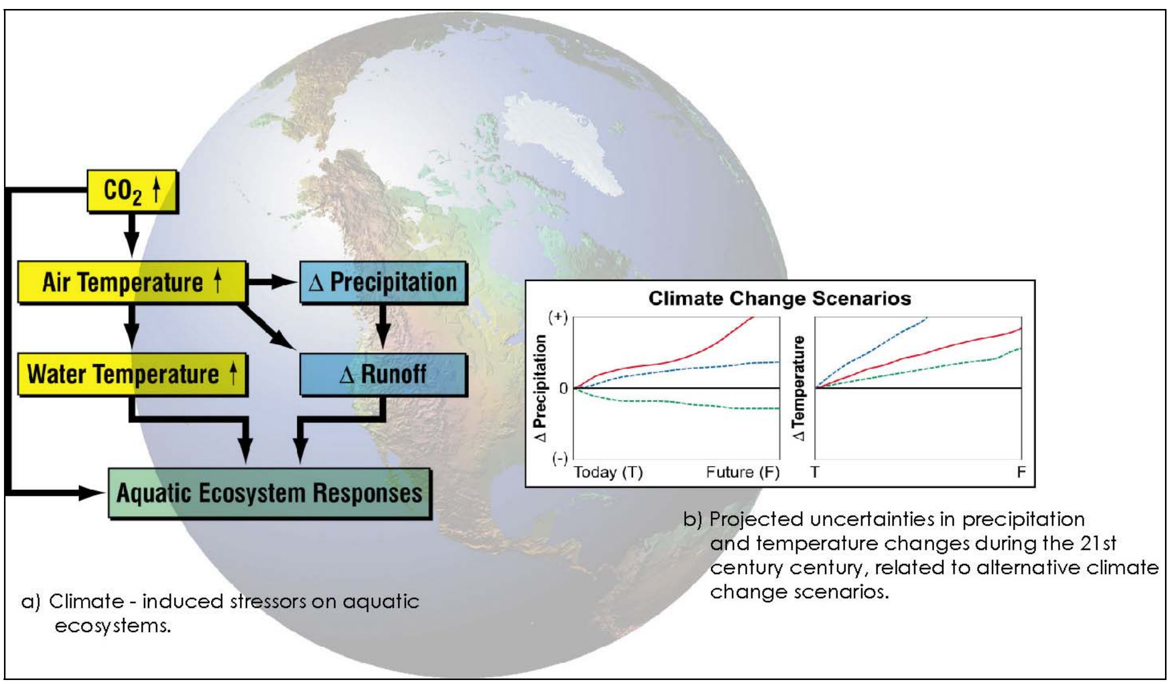


Figure 16-8
Expected Effects of Climate Change Over the 21st Century on Aquatic Ecosystems

Thermal Modeling Issues that Arise from Climate Change Studies

An indirect effect of climate change research over the past decade is the realization that there are large uncertainties in atmospheric and solar phenomena that influence the earth's energy balance, and therefore uncertainties in how to parameterize these processes. This issue is very relevant to thermal models that predict the transfer of short and long wave energy across the air-water interface. A global thermal energy budget is shown in Figure 16-9 to help illustrate several points that follow. One outcome of ongoing climate change research is that how clouds affect radiation balances is poorly understood. Effects of cloud elevation, thickness, and composition all have been shown to influence the radiation that passes through a cloud, is absorbed by a cloud, or is reflected. Yet today's thermal models do not consider any of these factors on surface radiation budgets. They only consider the cloud cover fraction [9].

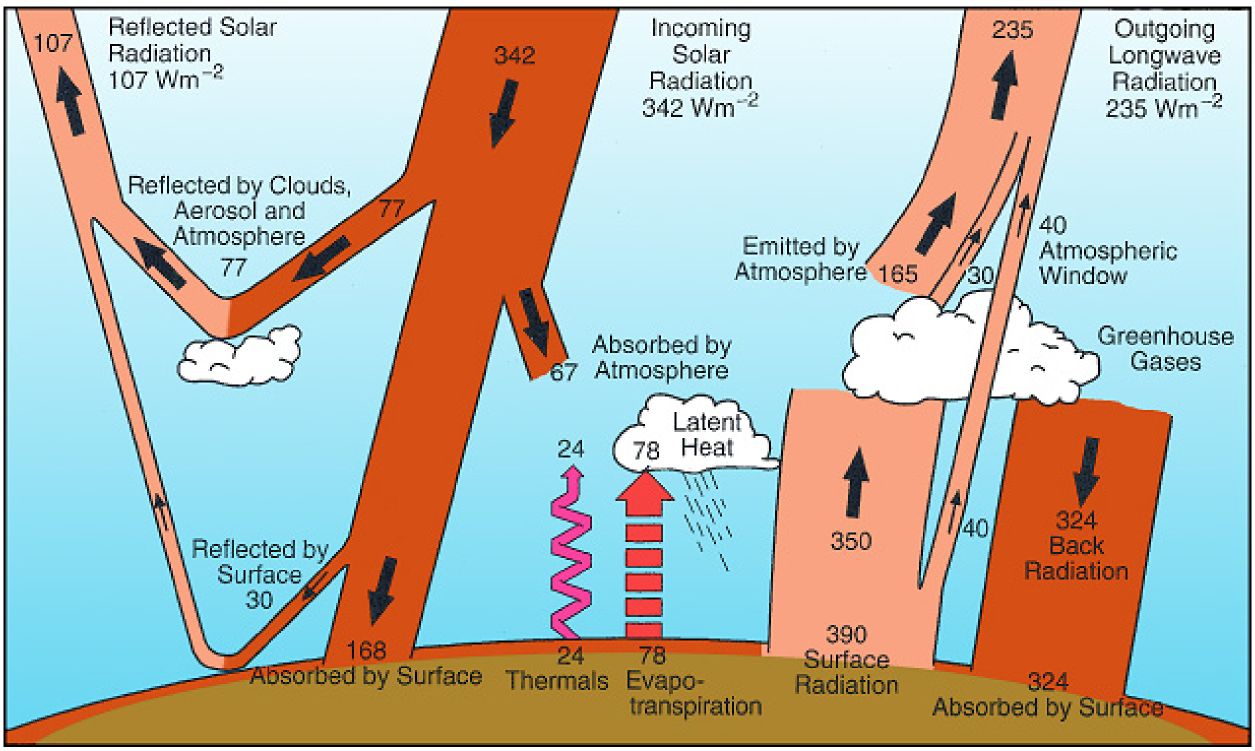


Figure 16-9
Earth's Global Balance [36]

A second aspect of the energy balance that is uncertain is that the total solar irradiance (TSI) of the sun itself is continuously changing, and there are uncertainties in both magnitude and direction of the change. Figure 16-10 shows the TSI variations over 20 years, based on satellite data and two reconstructions of that data for a 20 year period [37]. The two reconstructions use the same raw data, but analyze it differently, resulting in the two different time series. Both series show the 11-year sunspot cycle, where the TSI varies in a periodic fashion. However, the reconstruction by Frohlich and Lean [38] show that the two cycles are basically the same, where as the reconstruction of Willson and Mordinov [39] show the minimum irradiance has increased from the first cycle to the next. Since thermal models that simulate incoming solar radiation as part of the energy budget need TSI as input, the question arises as to what is the appropriate value to choose? Further, if simulations are to be made 10 to 100 years into the future, for example, what would be the appropriate irradiance to use then? The resolution of quantifying changing solar irradiance is likely to require additional satellite observations, rather than a re-analysis of the existing data.

Short Term Forecasting of Water Temperatures

In the future, models might be used in a forecasting mode to predict water temperatures a few days to a week in advance of the present. One reason for the interest in this is to optimize electrical power production, while at the same time not violating water quality standards. Figure 16-11 illustrates this scenario. During the three day period shown, the weather changes from a cool, rainy period to a warm, sunny period. Should the power plant operate tomorrow as it does today, the water temperature criterion would be violated since upstream temperatures would increase over this time period. Therefore the power produced would have to be reduced. If the power production is reduced more than necessary to just meet the temperature standard ("the optimum") there would be lost power that otherwise could have been generated. If the inlet temperatures could be predicted accurately in advance, the optimal operation of the power plant could be anticipated. To accomplish this however, may mean a considerable expansion in the modeling effort. First the weather would have to be predicted accurately over the period of interest. Weather predictions are not yet at the point where this can always be done reliably and accurately. Second, the watershed upstream of the plant may have to be modeled in its entirety or at least far upstream where the flow originates that reaches the power plant over the period of concern. Third, all this information needs to be efficiently and continuously assimilated into the model. Then an operational scheme could be imposed on the power plant to generate a pre-specified amount of power. At present, such a modeling system is well beyond the state of the art.

Evaporative Losses of Water From Power Plants That Use Once-Through Cooling Systems

In the past, evaporative losses from power plants that use wet cooling towers have been much more of a concern than evaporative losses from thermal effluent released to surface waters from a once-through cooling system. However, as water continues to become a more limited and vulnerable natural resource, the need to quantify and possibly reduce, this consumptive use is becoming a reality. Three-dimensional thermal models that employee detailed heat exchange algorithms are an appropriate way to make these estimates. As part of the energy balance calculations, latent heat exchange is calculated, and from the latent heat fluxes, evaporation losses can be calculated. The evaporation rates so calculated are absolute rates and include water that would have evaporated even if a thermal plume were not present. Consequently to estimate the excess evaporation from the plume, the thermal model would perform an identical simulation without the plume, calculate the evaporation, and subtract the background evaporation rate from the rate predicted with the plume. Such an approach produces the instantaneous evaporation rate. Typically, the amount of excess water evaporated over a period of one day is needed in order to get estimates in easily understandable units (such as MGD). Since evaporation rates depend on weather condition that are highly variable (such as wind speed and temperature) it would be important to estimate excess evaporation rates under a variety of weather conditions.

Expanded Use of Thermal Models for TMDL Analysis

In the United States today, nearly 2000 water bodies or water body segments are classified as thermally impaired, and will require thermal TMDL analyses to be performed. A detailed listing of these water bodies can be found on EPA's TMDL website. A number of issues relevant to electrical power companies are related to these TMDL analyses. One, power plants that discharge thermal effluents into the impaired reaches may be a large contributor to the thermal load. Such was the case for the Chattahoochee River example discussed above. On that river segment, the excess thermal load originated from two thermal power plants. Two, in the future background water temperatures are projected to increase and background river flows will likely increase in some parts of the United States while decreasing in other parts of the country, all due to climate change. As discussed above, water temperatures may increase by 2°C to 5°C by the end of the 21st century. A question that arises is how might the projected ambient temperature increase influence the 316(a) program? Given that in parts of the USA water temperatures are now near sublethal or lethal limits for indigenous species, the combination of increased ambient temperatures plus thermal loads may elevate water temperatures above these critical temperatures, even if thermal TMDLS were to be completed. Consequently, future planning for increased power projection, and thermal modeling of water temperature regimes in a warmer climate may be a focal point of thermal modeling.

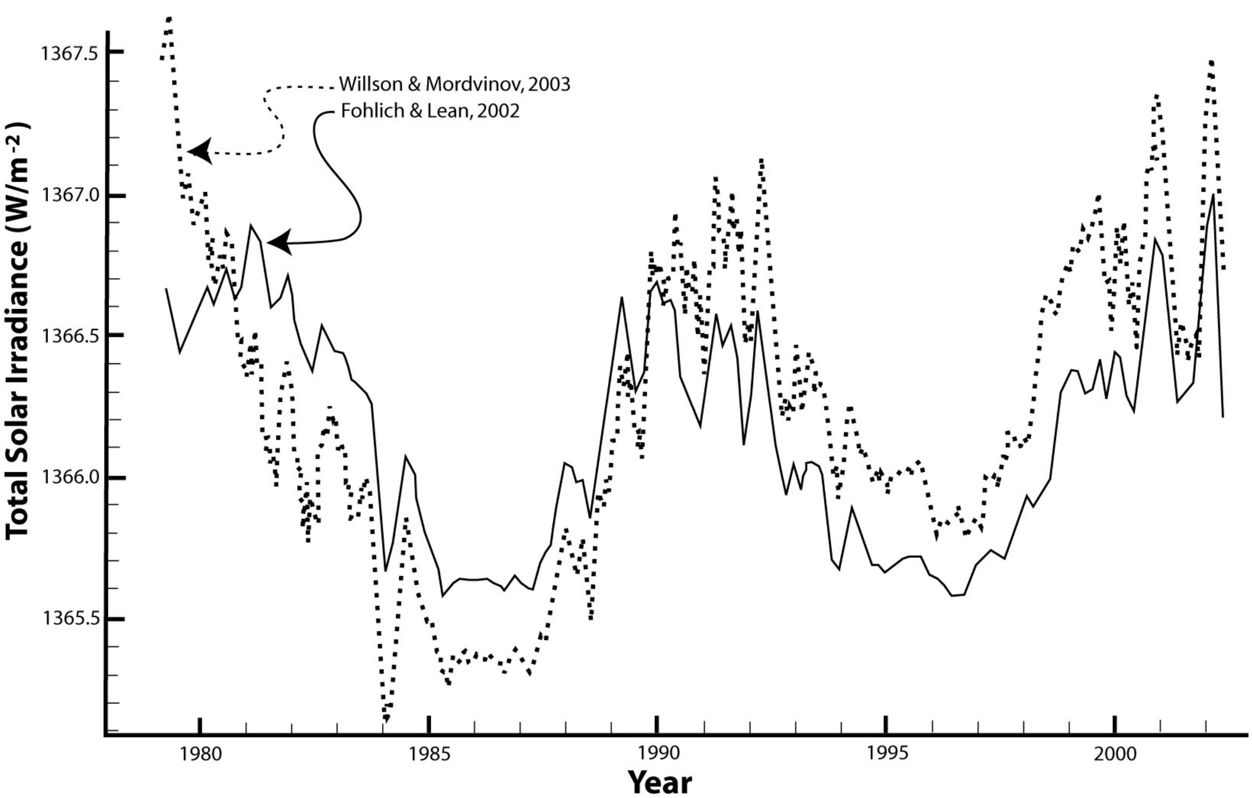


Figure 16-10
Comparative Results of Two Independent Studies to Determine Variability of the Sun's Total Solar Irradiance

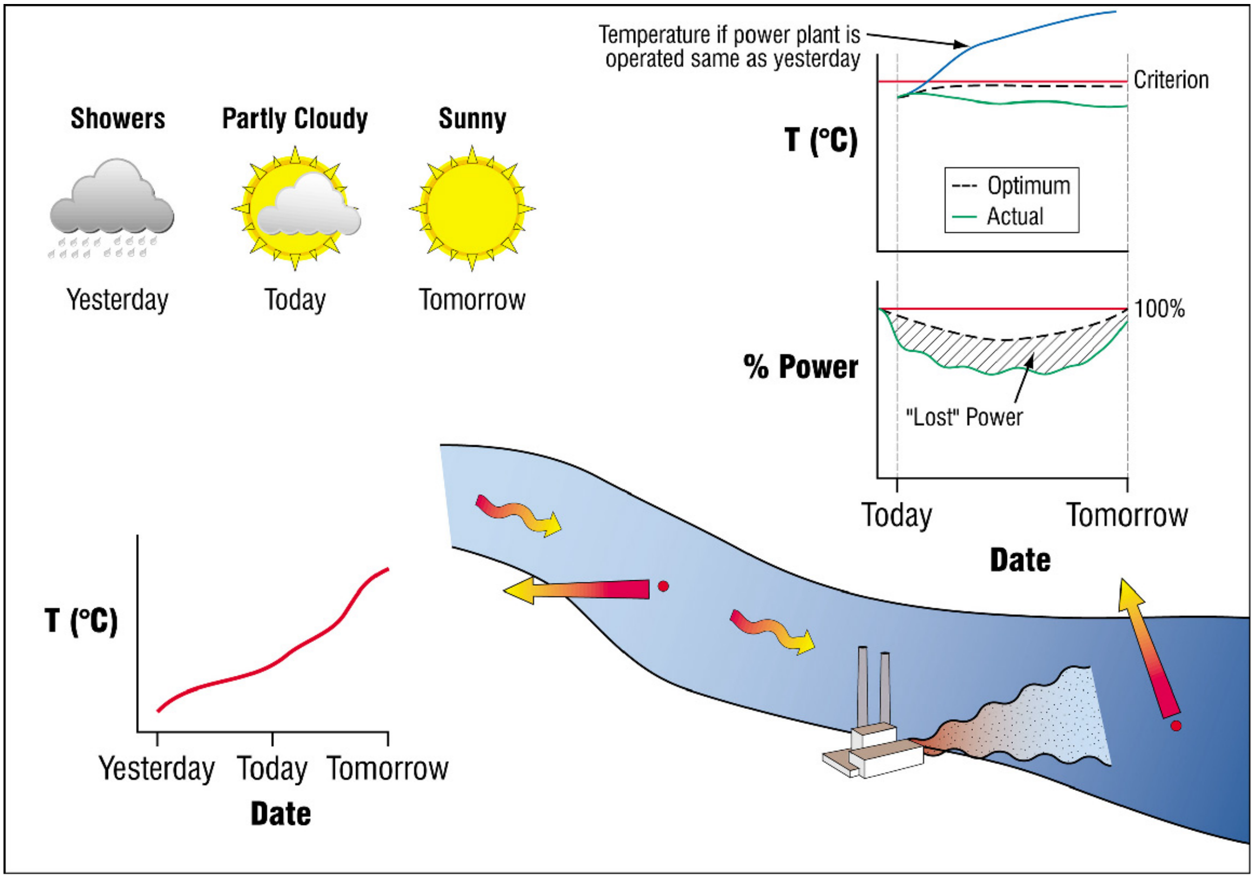


Figure 16-11
Conceptualization of Predicted Power Plant Operation a Day in Advance of Present

Conclusions

Thermal modeling of surface waters has progressed a long way since the days prior to the availability of computers. Faster computers with large storage capacity have contributed greatly to advances in thermal modeling. Advances in more sophisticated algorithms, three-dimensional modeling, and visualization techniques can all be traced back to advances in computer resources. However, issues do remain today in thermal modeling. For example, to make use of the more complex models requires more complex data sets, including waterbody bathymetry and boundary condition data that are complete and accurate. Often, historical data are insufficient to support the most complex models, and supplemental data collection efforts may be too limited to fill in all data gaps. New uses for models will inevitably be developed, whether to predict the effects of climate change on surface water temperature decades into the future, or to forecast water temperatures several days in advance of present time. The field of thermal modeling remains active and is expected to continue evolving during the 21st century.

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17
ENVIRONMENTAL ENHANCEMENT AND RESTORATION AS PART OF A THERMAL DISCHARGE STRATEGY

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Introduction

In May 2003, Argonne National Laboratory completed an extensive report for the Electric Power Research Institute (EPRI) describing the use of environmental enhancements (also known as mitigation or restoration measures) as part of a program for mitigating environmental impacts associated with cooling water intake structures [1]. The U.S. Environmental Protection Agency (EPA) is currently developing regulations under §316(b) of the Clean Water Act (CWA) to regulate cooling water intakes, and EPRI funded this report to provide background information and guidance to EPA and those companies that would need to comply with final §316(b) rules.

While planning an October 2003 workshop on CWA §316(a) issues (§316(a) sets requirements for thermal discharges), EPRI staff requested a paper that discussed how these same types of environmental enhancements could be made part of a thermal discharge compliance program. This paper gives some general thoughts on that subject, but readers are encouraged to review the full environmental enhancement report [1] or the many references cited in that report's reference list for additional information.

Background

Discharges of cooling water from power plants are regulated under National Pollutant Discharge Elimination System (NPDES) permits issued by state agencies or regional offices of the EPA. Thermal discharge requirements for these plants are generally based on state water quality standards for temperature. Most states allow the use of mixing zones such that discharge temperatures can exceed the in-stream temperature standards within mixing zones as long as the standards are met at the edges of the mixing zones. Each state determines the size and shape of allowable mixing zones differently. Thermal mixing zones often are very large in size because discharges of both cooling water and heat from once-through cooled plants are huge.

Section 316(a) of the CWA allows the states or EPA to establish alternative thermal limits if the discharger can demonstrate that the otherwise applicable thermal effluent limits are more stringent than necessary to protect the organisms in and on the receiving water body, and that other, less stringent effluent limitations would protect those organisms. Approximately 190,000 MW of U.S. generating capacity relied upon §316(a) variances in the early 1990s

[2]. §316(a) variances are renewed along with the facility's NPDES permit approximately every five years. Although some plants have dropped out of service and others have converted once-through cooling systems to closed-cycle cooling systems, most of that generating capacity still relies on §316(a) variances for compliance.

The most common approach for making a successful §316(a) demonstration is to conduct sufficient ecological studies to show that the plant's existing level of thermal discharge is protective. This report evaluates two other options for making a demonstration by modifying conditions in the ecosystem near the plant so that a balanced, indigenous population of organisms can be observed. These include environmental enhancement measures and trading. They are described below.

Environmental Enhancements

Environmental enhancements are activities that provide either expanded or improved habitat, thereby allowing aquatic populations to expand, or directly supplement fish populations through hatchery stocking programs. This paper looks at five types of enhancements, which are listed and discussed below.

Wetlands Creation, Restoration, and Banking. Wetlands are a rapidly vanishing ecological resource in North America. Wetland creation, restoration, and banking have been extensively used to protect and manage wetland resources and to enhance or increase fish and wildlife habitat. Wetland creation involves the construction of new wetlands at locations that previously had little or no natural wetlands present. It also is used to compensate for habitat impacts, associated with contaminant releases and subsequent cleanup at hazardous waste sites, and to restore habitat disturbed by mining operations, highway construction, housing developments, and other construction or excavation activities. Wetland restoration involves the rehabilitation of areas where previously supported wetland communities have been destroyed or degraded. Wetland banking refers to the restoration, creation, enhancement, or preservation of wetlands for purposes of providing compensatory mitigation in advance of authorized impacts to similar wetlands at other locations.

Creation of Submerged Aquatic Vegetation Beds. Submerged aquatic vegetation (SAV) beds play an important role in many freshwater and marine ecosystems. SAV beds provide nursery habitat for juvenile fish and foraging habitat for fish and wildlife, and they are essential habitat for many invertebrate organisms, such as brown shrimp. In some cases, SAV creation or restoration is used to increase recreational fisheries opportunities because of the fish-attracting aspects of SAV beds. Creation of SAV beds often involves planting native SAV in areas where historic SAV habitat has been destroyed.

Creation of Artificial Habitats. Artificial habitats (e.g., reefs) are widely used in freshwater and marine locations to create underwater structures that enhance fish reproduction, growth, and survival and promote increased production of invertebrate biota. Artificial habitats are used to increase spawning and nursery habitat for some fish species, as well as provide habitat for other aquatic biota. They are often used to enhance recreational fishing and diving opportunities as well. Creation of artificial habitats to produce relatively permanent fish habitat involves the placement of typically man-made materials such as cobble and boulders, engineered structures, or old ships.

Restoration of Fish Passage. In many river systems, dams, dikes, culverts, and water diversions have been widely used to provide flood control, generate hydropower, provide stream crossings, and create lakes and reservoirs for water supply purposes. Each type of structure impacts the ability of resident fishes to move between in-stream habitats. The restoration of fish movements has received significant attention throughout North America and elsewhere. Restoration of fish movement includes the use of such technologies as fish ladders, lift gates, and the capture and trucking of fishes around riverine obstacles.

Fish Stocking. Fish stocking is widely used in the management of recreational and commercial fisheries in both freshwater and marine systems. Fish stocking is also being used in the recovery of endangered fishes. Stocking involves the production of fish in hatcheries for subsequent release into the wild. Many fisheries in the United States are the result of historical stocking, and some are maintained solely through ongoing stocking. Depending on the species and stocking goals, fish can be stocked from within a few weeks or less of hatching all the way to catchable size.

Implementation of Environmental Enhancements

If enhancements are proposed for use as part of a §316(a) program, the company requesting the variance as well as the regulatory agency will need to evaluate certain questions:

- How do enhancements mitigate thermal discharge impacts?
- What resources do they benefit (specific target species or broad ecological benefit)?
- What are the expected ecological and environmental responses?
- What are the implementation costs?
- What are the operations and maintenance requirements and costs?
- How long does it take until the enhancement is effective?
- How long must the enhancement be monitored?

The answers to each of these questions may be different depending on the type of enhancement measures that are employed and the site-specific physical and environmental features of the plant's location. Table 17-1 compares implementation time, costs, and monitoring requirements for the five types of enhancements.

Reference 1 provides some general guidelines on how to undertake these evaluations. A four-step framework was identified to aid in the selection and scaling of enhancement projects. Steps in this framework include: (1) setting the baseline, which determines the type and amount of resources for which mitigation is needed; (2) considering technological and/or operational alternatives; (3) selecting the enhancement approach; and (4) scaling the enhancement project. Critical components of this framework include the need for early and frequent discussions with regulators, natural resource agencies, and appropriate stakeholders; a cost-benefit analysis of potential technological and/or operational mitigation alternatives; and the use of one or more metrics linking impingement and entrainment impacts to environmental enhancement, and then scaling the restoration project to meet the needed mitigation level.

Table 17-1
Comparison of Considerations for Implementation of Enhancements

Enhancement Measure	Implementation Time	Implementation Cost	Monitoring Requirements	Operation and Maintenance Costs
Wetlands construction/restoration	<1 year	Short-term cost moderate to high for initial ground work	Likely–up to 5 years	Moderate for monitoring and maintenance
Wetlands banking	<1 year	Moderate to high, but one time only for purchase	None (performed by bank owner)	None (performed by bank owner)
Submerged aquatic vegetation	<1 year	Short-term cost moderate to high for construction	Likely–up to 5 years	Moderate for monitoring
Artificial reefs	<1 year	Short-term cost moderate to high for construction	Likely–up to 5 years	Moderate for monitoring
Fish passage	<5 years	Short-term cost very high for construction	Annual	Low to moderate for monitoring and maintenance
Stocking	Annual, depending on stocking plan	Long-term cost moderate for annual stocking	None to annual	Moderate for annual monitoring

One of the most difficult issues surrounding the use of enhancement measures is how to equate the ecological value of a unit of enhancement effort. For example, how many young fish or adult fish can be expected to result from the restoration of one acre of wetland or removal of a river barrier that leads to use of 3 hectares of additional river bottom habitat? Following the oral presentation of this paper at the EPRI §316(a) workshop, there was extensive discussion of that issue among members of the audience. The science for making these predictions is in its formative stages and should improve over time. Many of the biological issues involved with making the valuations were discussed in a recent §316(b) conference presentation by a leading fisheries biologist who has studied power plant impacts [3].

Trading Strategies

Environmental trading occurs between two entities when one sells a credit for better-than-required environmental performance to another entity that uses the credit in lieu of directly meeting its own environmental performance requirements. For wastewater discharges from point and nonpoint sources, this type of trading is known as effluent trading. The electric power industry has taken advantage of various types of air emissions trading during the past decades, but has not participated in water trading efforts to date. Two examples of actual or potential trades involving heat as the pollutant are described here.

The first is a project that EPRI is co-funding. It evaluates the quality of the Cheat River watershed in West Virginia. The Albright power station discharges heated water to the Cheat River, which causes an adverse impact on the ecosystem. Researchers have evaluated many of the environmental stressors on the same stream segments. They concluded that acid mine drainage from abandoned coal mines in the area is having a far greater ecological impact on the river. The goal of this project is to establish a trading program for the Cheat River in which entities impacting the stream can attempt to mitigate those pollutant sources with the greatest impact (i.e., the acid mine drainage) in lieu of automatically controlling their own pollution contribution to a full extent [4]. In other words, they would try to "get the most bang for the buck." The program is still being formulated and no trades have yet occurred.

The second program takes place on the Tualatin River in Oregon. The river is currently too warm to support its designated use for coldwater fisheries, partially because of discharges from several municipal sewage treatment plants. A variety of options were evaluated, including some impractical solutions like banning water heaters in the local county and refrigerating the effluent. The selected option of having the municipalities that operate the sewage treatment facilities plant trees along the banks of the river to provide shade and indirectly cool the water was practical and affordable [5]. In addition, this approach offered other benefits like increased stability of stream banks, an additional barrier to runoff from the land adjacent to the river, and increased habitat.

Conclusions

The impacts and costs of complying with more restrictive thermal discharges standards can be far-reaching and quite significant [6]. Although thermal discharge issues have been relatively quiet for the past 10-20 years, there is increased potential that they may come to the forefront again as states assess the condition of streams through the total maximum daily load (TMDL) program. If elevated temperature is indicated as preventing a stream from meeting its designated uses, states will develop thermal TMDLs and allocate the quantity of heat that can be added to those streams. It is likely that some of the power plants that are currently operating under §316(a) variances will either lose the variances or be asked to conduct new, more rigorous demonstrations that variances are warranted.

Power companies should be prepared to conduct new §316(a) demonstrations. If existing thermal discharge levels are not found to protect a balanced indigenous population of aquatic organisms, the companies should consider offering environmental enhancements or trading to provide additional ecological value to the receiving waters and the local environment. The guidelines and techniques outlined in reference 1, although prepared for §316(b) issues, can be applied to §316(a) issues, too.

The legal language in §316(a) may not clearly allow the use of enhancements or trading. Consequently, some opponents of anything other than the strict command-and-control regulatory approach may object to including such measures for compliance. Although there are no guarantees, efforts to build cooperation among all interested parties through early and continued communication can lead to better acceptance of these innovative programs. Willingness to provide more enhancements than might minimally be expected can also improve stakeholder acceptance. Sometimes traditional opponents can be converted to allies for innovative programs if they can be convinced of the value of such programs early in the process.

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18
AN ECOLOGICAL INDEX TO ASSESS ALTERNATIVE REGULATION OF THERMAL IMPACT AT THE ALBRIGHT POWER STATION, WV

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Introduction

There is a critical need for restoration action and more effective watershed management approaches in the Mid-Atlantic Highlands (MAH) region of the eastern U.S. [1]. The MAH consists of the mountainous portions of Pennsylvania, Maryland, Virginia, and Kentucky, and the entire state of West Virginia. Ecologically this region is important because it contains a large percentage of the high elevation, coldwater stream ecosystems occurring in the eastern U.S. Furthermore, this area represents the headwaters of two vitally important regional watersheds: the Chesapeake Bay and the Mississippi River. Consequently, the overall quality of aquatic ecosystems in the MAH has implications for water resources that extend far outside the region.

A recent assessment by the USEPA of stream ecological condition in the MAH found that more than 70% of streams are severely or moderately impaired by human related stressors [2]. Impairment to aquatic communities in this region extends from a range of human related activities, including agriculture, forestry, and urban development, but mining related impacts are unquestionably the most severe. For example acid mine drainage (AMD) from abandoned underground mines has degraded hundreds of miles of streams in West Virginia alone. Furthermore, two highly controversial coal extraction methods, longwall mining and mountaintop removal/valley fill mining, continue to develop at an alarming rate, thereby placing watersheds throughout the region at considerable risk of deterioration.

Several recent scientific advances and policy directives have improved the likelihood of effectively managing mining impacted watersheds in this region. First, the West Virginia Division of Environmental Protection (WVDEP) has worked in cooperation with the USEPA to conduct watershed assessments and develop Total Maximum Daily Load (TMDL) programs for AMD impacted watersheds throughout the state [3, 4, 5]. The successful implementation of these programs would dramatically improve surface water chemistry and ecological integrity of aquatic ecosystems in the state. Second, the WV state legislature recently passed a stream Anti-Degradation policy, which theoretically will protect remaining high quality aquatic resources in the region. Third, West Virginia, with support from the USEPA, industry representatives, and local watershed organizations is exploring the feasibility of developing watershed specific and statewide water quality trading programs. If successful, the trading program could facilitate implementation of TMDL plans, produce significant improvements in water quality, and reduce the economic burden of meeting clean water goals in the region.

Effective use of these advances to restore and manage mined watersheds in the Mid-Atlantic Highlands will require a watershed scale approach. I draw this conclusion for several reasons. First, stream ecosystem stressors stem from multiple, diffuse sources scattered throughout the watershed. For example, many watersheds in the Central Appalachians are simultaneously impacted by multiple sources of sediments, nutrients, mine drainage, and stormwater runoff, and ecological impairment often is the result of the cumulative and interactive effects of these stressors. Consequently, effective restoration and management must consider multiple stressors and sources as a whole rather than in isolation. Such an approach requires a watershed scale perspective. Second, stream remediation in mined watersheds is extremely expensive. A typical point source of acid mine drainage can cost anywhere from $10,000 to $100,000 per year to treat depending on local geology, the extent of mining, and the size and chemical conditions of the receiving waterbody. Third, many sources of impairment in mined Appalachian watersheds predate federal regulatory statutes. Consequently, straightforward management of water quality through the NPDES permitting process cannot be used to recover stream water quality and designated uses. The cost of remediation combined with the fact that most pollution sources cannot be tied to a responsible party makes it absolutely necessary to prioritize remediation efforts, and this is possible only through a watershed scale approach.

Finally, water quality trading has the potential to generate economic incentives for restoring and protecting aquatic resources in mined watersheds. However, our analyses indicate that opportunities for same pollutant trades in the Cheat River watershed are scarce, and this probably is true in most other central Appalachian watersheds. One reason for this is that there are so few active or permitted mine sites in this region. Another is that most permitted mine sites are already at or below their allowed TMDL based effluent levels for iron, aluminum, and manganese. As a consequence, any future benefit of water quality trading in mined watersheds will necessarily involve a cross-pollutant trading program. I believe that a cross-pollutant trading program is a viable option for implementing the Cheat River TMDL. However, such a program will require a significant level of regulatory oversight and a detailed watershed management approach in order to be successful.

Objectives

In this report, I develop and demonstrate a method for assessing the ecological benefits of pollutant trade scenarios in the lower Cheat River basin. My method employs an ecological based index, which can be used as a common currency when calculating the environmental gains and losses of a specific trade. My method also allows for the calculation of pollution/ecological condition/dollar equivalencies. My specific objectives are as follows:

1. Identify ecological issues critical to a trading program.
2. Develop and justify an ecological index for use in the Cheat River watershed.
3. Demonstrate how the index can be used to calculate equivalencies between pollutants, ecological condition, and dollars.
4. Demonstrate how the index can be used to identify priorities for restoration.
5. Demonstrate how the index can be used to calculate the net ecological benefit of various trade scenarios.

I believe that this approach will prove invaluable in developing a far reaching trading program that can allow, under certain restrictive conditions, trades involving multiple stressor types and sources. I cannot express enough, however, the importance of embedding this process into a holistic watershed restoration and management program.

Ecological Issues Critical to a Water Quality Trading Program

The foundation of my approach to a holistic water quality trading program rests on three basic tenets of stream ecology, and ecosystem assessment and restoration. From this foundation, I develop an ecological index and demonstrate how the index can be used to guide trade decisions to facilitate recovery of watershed condition in the Cheat River watershed.

Environmental and Ecological Condition

The first tenet is that at any given time, the overall condition of a watershed is influenced simultaneously by a variety of human influences and environmental stressors (Figure 18-1). For example, active forestry practices that include road and skid trail development typically produce an increased load of sediments to waterbodies within the watershed. Typical agricultural practices also produce increased sediment loads, but also increased rates of nitrogen and phosphorus loading. Although it may be difficult to quantify, it also is generally agreed that biological communities within a watershed (e.g., algae, invertebrates, and fishes) are impacted by the cumulative and interactive effects of multiple environmental stressors. Consequently, changes in biological communities can be measured in a way such that they provide an indication of the level of impairment within the watershed [6, 7, 8, 9], and therefore, provide a measure of ecological condition. For example, ecologists and state regulatory agencies have spent considerable time and resources developing benthic invertebrate and/or fish community based indices of ecological condition. These indices are used regularly to guide aquatic ecosystem assessment and management throughout North America and the world [10, 11].

I define “Environmental Condition” as a measure of the overall condition of a watershed as defined by physical (e.g. habitat, sediment, temperature, streamflow) and chemical (e.g., pH, conductivity, metals, nutrients, dissolved oxygen) properties. Consequently, the environmental condition of a watershed is simply a measure of the suite of environmental variables that describe the watershed. Likewise, I define “Net Environmental Benefit” as a net improvement in the physical and chemical conditions of a watershed. In contrast, I define “Ecological Condition” as a measure of the overall condition of a watershed as defined by biological communities, especially benthic invertebrate and fish assemblages. Furthermore, “Net Ecological Benefit” is defined as a net improvement in the condition of biological communities in the watershed.

Quantifying “Net Environmental Benefit” requires a comparison of like environmental variables. It is impossible to calculate “Net Environmental Benefit” if two actions influence different environmental properties. For example, if Action 1 increases acidity by 1 ton/year, and Action 2 reduces acidity by 3 tons/year, then the “Net Environmental Benefit” is equal to 2 tons of acidity/year. However, if Action 1 increases phosphorus levels by 1 ton/year, and Action 2 reduces acidity by 3 tons/year, then “Net Environmental Benefit” cannot be calculated. As a consequence, “Net Environmental Benefit” can be applied to same-pollutant trades, but not to cross-pollutant trades. On the other hand, quantifying “Net Ecological Benefit” can potentially be used to reduce multiple stressors to a single common denominator, and as a consequence, provide a mechanism for assessing benefits associated with cross-pollutant trades. For example, if Action 1 increases phosphorus by 1 ton/year and reduces ecological condition by 100 units/year and Action 2 reduces acidity by 3 tons/year and increases ecological condition by 300 units/year, then "Net Ecological Benefit" of Action 2 relative to Action 1 is 200 units/year. In fact, the primary objective of developing a measure of "Net Ecological Benefit" is to provide a means for objectively determining the potential environmental gains and losses of a cross-pollutant trading program.

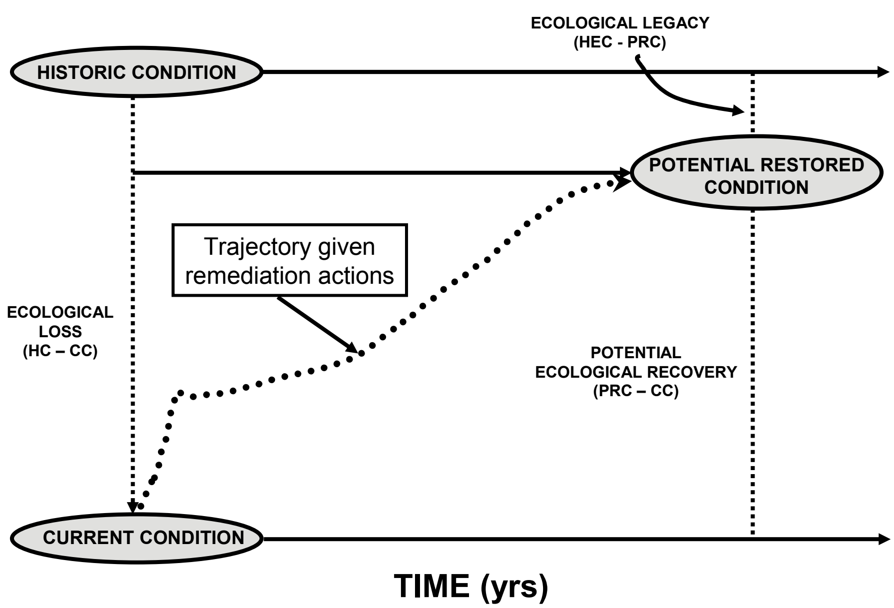

Figure 18-1
A General Model of Watershed Condition and Aquatic Ecosystem Restoration and Management

Hierarchical Structure of Watersheds

The second tenet is that aquatic ecosystems exist as hierarchically nested subsets, where each subset represents a different spatial scale [12]. For example, microhabitats (areas < 1 m^2) are nested within hydraulic channel units (e.g., pools and riffles), channel units are nested in stream reaches, reaches are nested in stream segments, segments are nested in local drainage networks, and drainage networks are nested within whole watersheds. My approach explicitly considers the hierarchical structure of watersheds and multiple spatial scales and attempts to quantify ecological condition across these scales. Specifically, I will focus on scales ranging from the stream segment scale (1-km) up to whole watersheds (100–200-km^2 basin area). Although our approach stops at the scale of the Cheat River watershed, I do not rule out the possibility of extending this approach to larger scales, such as the upper Monongahela River basin or even the upper Ohio River basin.

Watershed Degradation and Restoration

The final conceptual foundation of my approach is illustrated in Figure 18-1, which is a modified version of a general model of aquatic ecosystem restoration presented by the National Research Council [13]. Briefly, the model recognizes that watershed and instream attributes interact to determine present ecological condition. Stream condition is measured by the health of the stream biotic community and is a function of physical habitat, the stream hydrograph and water chemistry. Any changes in the stream's functional attributes, will likely affect its condition. In most cases, because of historic or active land use practices, the current ecological condition of a watershed is below its historic condition. An objective of paper is to describe a process that can be used to quantify each of the critical variables illustrated in Figure 18-1. In doing so, I will show how the ecological costs and benefits of alternative management scenarios can be assessed.

An Ecological Index for the Cheat Watershed

I developed an ecological index to provide a method for quantifying ecological condition at multiple spatial scales. It also was necessary for the index to provide a meaningful measure of the Net Ecological Benefit of alternative management scenarios in the Cheat watershed. To this end, I found that a useful index must include at least two components: 1) the area of stream affected by an action, and 2) the ecological condition of that area.

Stream Size

The simplest measure of affected area is stream miles. However, this measure does not consider variation in the width or flow volume of streams within a watershed. Other potential measures for the area of influence include: stream surface area (length x width), stream order (Strahler), and watershed area drained by the stream segment. Each of these alternatives, in some way, captures spatial variability in stream size. Of these alternatives, I prefer stream area, because it is simple, it gives greater weight to larger streams, and it is a continuous rather than ordinal measure.

I calculated the surface area of all stream segments recognized by the WVDEP in the lower Cheat River basin. This includes all perennial streams in the watershed that could potentially be placed on the state's 303(d) list of impaired waterbodies. A stream segment was defined as a stream reach bounded at the upstream and downstream ends by perennial tributary confluences. In other words, a stream segment is a reach with no significant surface water inputs. Surface area was calculated for each segment in one of two ways. Method 1 was used for smaller first to third order stream segments. In this method, ArcGIS was used to estimate the length of the stream segment. I then used a regression equation relating active channel width to basin area to estimate the average width of the stream segment. The regression equation used data from 50 locations distributed throughout the Cheat watershed (Petty et al., unpublished data) and was of the following form:

$$\text{Active Channel Width} = 0.31 \text{ (basin area in km}^2\text{)} + 3.97$$ **Equation 18-1**

Method 2 was used for larger stream segments, including the Big Sandy Creek mainstem and the Cheat River mainstem. This approach simply used ArcGIS to estimate surface area directly from digital photographs of the watershed.

Ecological Condition

Numerous ecological condition indices exist and may be of potential value in a trading program. The most commonly used measures for ecological condition are based on benthic macroinvertebrate and fish communities. Benthic invertebrate based examples include: the WV Stream Condition Index (WVSCI) [5], the Hilsenhoff Biotic Index (HBI) [9], and the Save-Our-Streams Index. Fish community based examples include: the Mid-Atlantic Highlands Index of Biotic Integrity [2], Fish Diversity, or Biomass of Targeted Species (trout and/or smallmouth bass).

I selected the family level WVSCI for use in developing an ecological condition index for the Cheat River basin. Family level means that it was developed based on identification of invertebrate taxa at the family level of taxonomic resolution. I selected the WVSCI score for two reasons. First, this index is used widely by the WVDEP as part of the statewide watershed assessment program. Second and more importantly, I have found it to be extremely powerful as a continuous measure of impairment in watersheds impacted by acid mine drainage (Figure 18-2). The WVSCI declines continuously as a function of several important chemical measures of AMD impairment, including pH, iron concentration, aluminum concentration, and manganese concentration (Figure 18-2). The ability to predict ecological condition as a function of environmental conditions is of critical importance in determining the utility of a particular ecological condition index [6].

The WVSCI is a multi-metric index of ecological condition that integrates numerous measures of the benthic invertebrate community into a single value [5]. The metrics included in the final index include: 1) total number of families (i.e., total family richness), 2) number of families in the orders Ephemeroptera, Plecoptera, and Trichoptera (i.e., EPT family richness), 3) percent of all families that are EPT taxa (i.e., % EPT), 4) percent of all families that are considered pollution tolerant (i.e., % tolerant taxa), and 5) percent of the total community of invertebrates that are in the top two dominant taxa (i.e., % dominant taxa). Metrics 1-3 of the WVSCI are expected to increase with decreasing levels of environmental impairment. In contrast, metrics 4 and 5 are expected to decrease with decreasing levels of environmental impairment. The final value is standardized, such that the highest quality stream segments in a watershed receive scores of 100. The lowest score possible is a WVSCI of 10 [5].

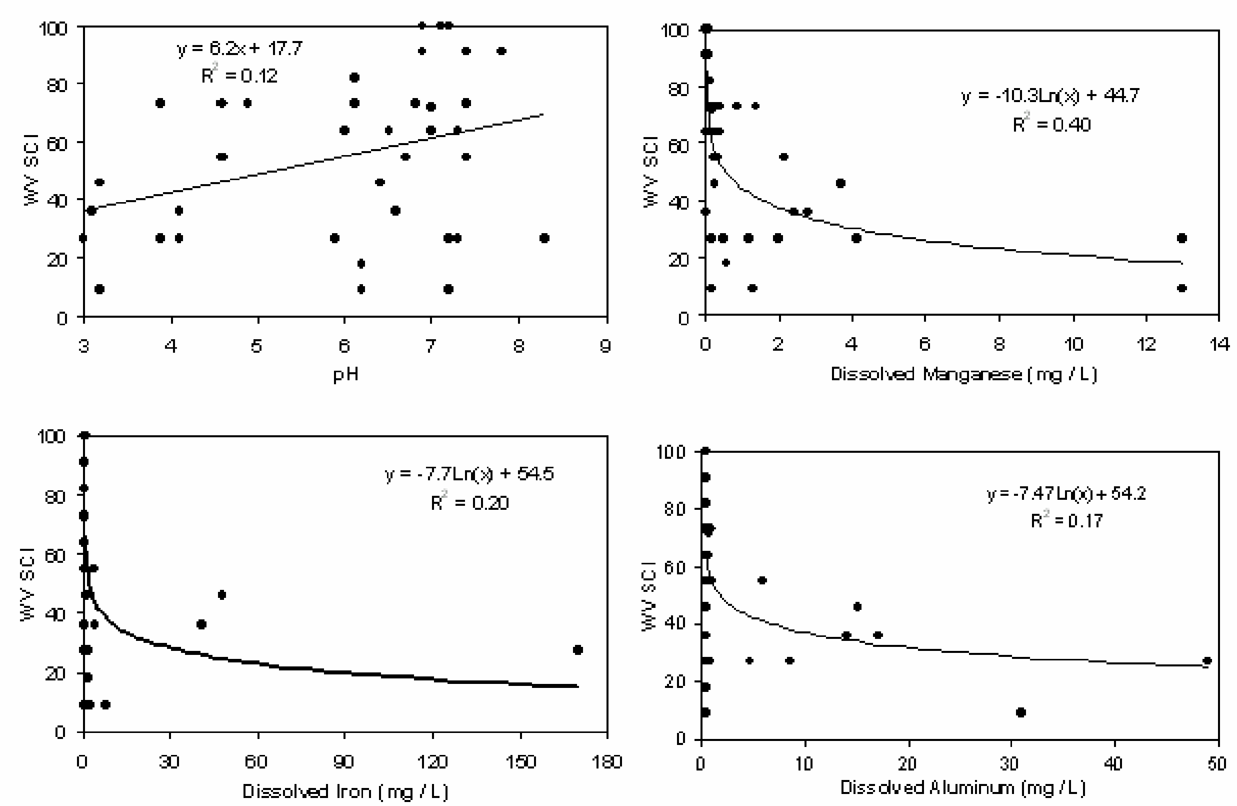

Figure 18-2
Relationship Between WVSCI Various Water Chemistry Parameters. Data used in this Analysis are from the WVDEP Watershed Assessment Program 1996 Assessment of the Cheat River Watershed [3]

EcoUnits Index

The index that I propose for calculating ecological condition and Net Ecological Benefit in the Cheat River basin incorporates a measure of stream surface area and ecological condition as the WVSCI. I refer to this index as the "EcoUnits Index," because each stream segment within the watershed and the watershed as a whole possess a measurable number of ecological units that are a function of the surface area and ecological condition of the segment.

The basic structure of the EcoUnit Index (EUI) is:

EUI = Stream Surface Area (acres) x Observed SCI/Max SCI **Equation 18-2**

The resulting index produces a value in units of acres. The value itself can be interpreted as the surface area of the stream segment weighted by the ecological condition of the segment. Consequently, the EUI for a stream segment will equal the surface area of the segment when ecological condition within the segment is equal to the condition of the highest quality segment in the watershed. EUI is calculated separately for each stream segment in the watershed. A score for an entire watershed or subwatershed can be calculated by summing segment level EUI across all segments within the watershed of interest.

Shortcomings and Alternatives to the EcoUnit Index

The utility of this index for managing trades will be developed in subsequent sections. However, I feel that several shortcomings of the EUI should be considered now. First, the index places a greater weight on larger streams (because they are wider) than smaller streams. Stakeholders in the Cheat River watershed agreed that larger streams should be valued more highly than smaller streams. On a pure ecological basis, this decision is difficult to justify because of the extreme importance of small streams to stream ecosystem processes [14]. Ultimately, if deemed necessary, the EUI can be modified to use stream length instead of surface area as a measure of the area of management influence. For the purposes of this report, however, I have used surface area in all subsequent analyses.

Second, the effects of ecological condition and surface area on the EUI are given equal weight. This is not particularly important when simply describing the current condition of the watershed. However, it is extremely important when comparing the ecological costs and benefits of alternative mitigation scenarios. For example, say Action 1 will reduce the modified WVSCI score from 1.0 to 0.5 over an area of 10 acres. The result is a change from 10 EUI's (1 x 10) to 5 EUI's (0.5 x 10) or a loss of 5 EcoUnits. Now, say that Action 2 will increase the WVSCI score from 0.5 to 0.6 over an area of 50 acres. The result is a change from 25 EUI's (0.5 x 50) to 30 EUI's (0.6 x 50) or a gain of 5 EcoUnits. Obviously these are two very different actions that produced significant changes to the ecological condition of the watershed but no net loss of EcoUnits in the watershed.

Finally, there are numerous measures of stream condition that I have not included in the EUI. For example, the Kentucky U.S. Army Corps of Engineers have adopted a similar procedure to assess stream fill permits. The index that they use combines the following elements: stream length, an invertebrate-based condition score, conductivity, and key components of the USEPA rapid visual habitat assessment score (riparian condition, substrate embeddedness). I have not included a variety of physical and chemical parameters within the EUI for the Cheat River basin for a couple of reasons. First, the purpose of the EUI is to describe the ecological condition and not the environmental condition of stream segments in the watershed. Adding measures of habitat and chemistry to the index would result in an admixture of environmental and ecological variables. Second, I know that there is a relationship between ecological condition and measures of physical and chemical condition. For example, WVSCI score tends to decrease in streams with high conductivity and low visual habitat assessment scores (Petty et al., unpublished data). Consequently, it would be inappropriate to include multiple, non-independent variables into the equation used to calculate the EUI.

Uses for the EUI

The EUI can be used in numerous ways to facilitate the management and restoration of the Cheat River watershed (Table 18-1). These uses can be split into three categories: those that are essential for the development and implementation of watershed management plans, those that are essential for the development and success of a cross-pollutant trading program, and those that are essential for both watershed management and cross-pollutant trading. The objective of the following section is to illustrate each of these uses and the potential of the EUI in guiding TMDL implementation and a water quality trading framework.

Table 18-1
Potential Uses for the EcoUnit Index

1. Estimate variables in the watershed restoration timeline as described in Figure 16-1. Variables include: current ecological condition, historic condition, potential restored condition, ecological loss, potential ecological gain, and ecological legacy.
2. Estimate measures of current condition, ecological loss and potential restored condition at multiple spatial scales.
3. Calculate the intensity and extent of ecological loss in a watershed.
4. Predict changes in ecological condition as a function of expected improvements or reductions in water quality.
5. Develop ecological loss equivalencies among different stressors (in isolation or combined).
6. Develop equivalencies between ecological condition and dollars.
7. Compare ecological gains and losses (i.e., Net Ecological Benefit) of alternative management or trade scenarios.

Application of the Index to the Cheat River Watershed

Description of the Cheat River Basin

The Cheat River is one of the larger tributaries to the Monongahela River, which, with the Allegheny River, forms the Ohio River in Pittsburgh, Pennsylvania. Its watershed—3,671 square kilometers (km) or 1,426 square miles—is located almost entirely in West Virginia, although 6.7% lies in Pennsylvania and 0.1% in Maryland. Two major branches meet in Parsons to form the Cheat River: Shavers Fork flows north-northwest from Pocahontas County, and the unlabeled Black Fork gathers several smaller tributaries (Blackwater River, Dry Fork, Laurel Fork, Glady Fork and Red Creek) from Tucker and Randolph Counties. The mainstem of the Cheat River flows north 135 km (84 miles) from Parsons to its confluence with the Monongahela River at Point Marion, Pennsylvania, just north of the border with West Virginia. The river is dammed a short distance upstream from its mouth to form Cheat Lake, also known as Lake Lynn (Figure 18-3). Upstream from Cheat Lake, the Cheat is the largest uncontrolled watershed in the eastern United States.

Because of large scale variation in coal geology, the Cheat River mainstem can be separated into an upper and a lower sub-basin (Figure 18-3). The upper sub-basin includes all tributaries draining to the Cheat River beginning at the confluence of the Shavers Fork and Black Fork in Parsons, downstream to Pringle Run. Mining is essentially absent from the upper sub-basin and provides an ideal, low impact reference against which to compare environmental and ecological conditions in the lower sub-basin. The lower sub-basin begins at Pringle Run and continues downstream to Cheat Lake. In this region, surface and deep mines are prevalent and there exists a high degree of variability in surface water quality.

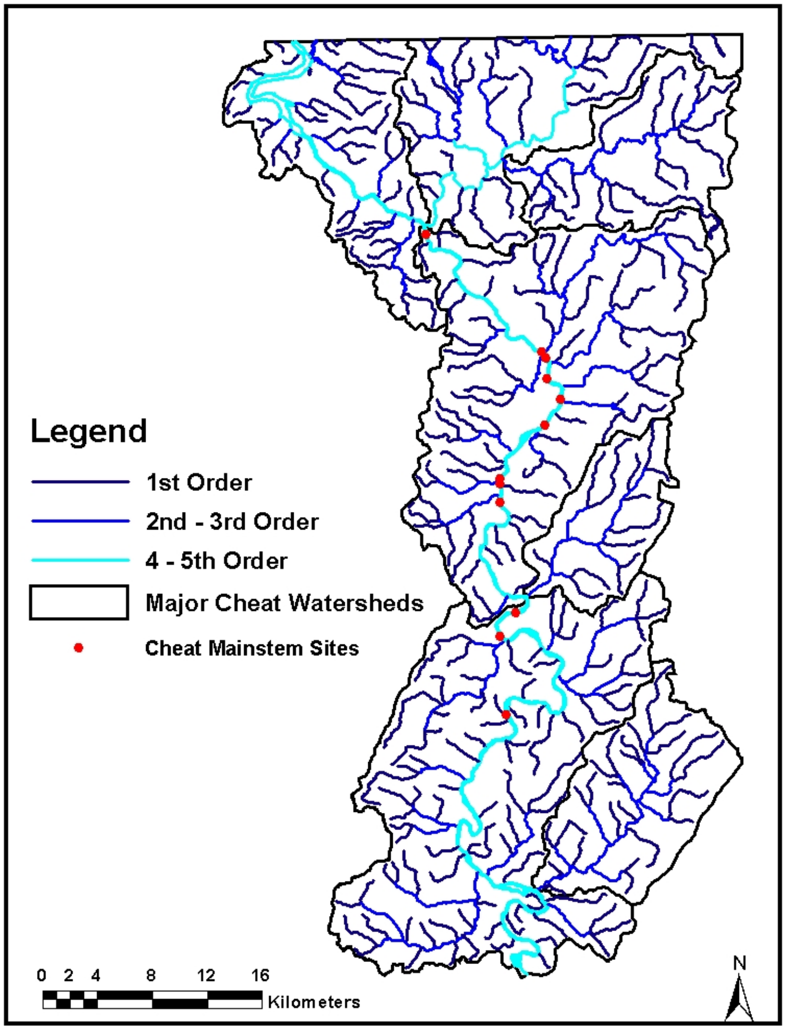

Figure 18-3
Map of Cheat River Watershed, WV

Sitting right in the middle of the most intense AMD impacts, is a coal-fired steam electric generating plant that operates along the Cheat River in Albright (Figure 18-3). This facility, built in the 1950s, has a generating capacity of 292 megawatts, burns approximately 600,000 tons of coal per year and still operates without cooling towers. Clean Water Act Section 316(a) thermal variances have been granted for these thermal discharges, which would otherwise violate the Clean Water Act. In 1977, when the plant received its first variance, the Cheat River at Albright was significantly polluted by AMD, and the thermal discharge arguably had little impact on stream life that was already severely impacted. Since then, DEP has required various fish and benthic surveys over the years to justify a continuance of the variance. When the NPDES permit most recently came up for renewal, however, the Cheat River had recovered to the extent that DEP required more detailed analysis to justify a continuation of the variance. In return for a continuation of its thermal variance, Allegheny Energy might agree to finance AMD treatment in significantly impaired tributaries upstream from Albright. A critical objective of this report is to develop the analytical tools needed to justify such a trade.

Variation in the WVSCI Scores for Stream Segments in the Cheat Watershed

I used data from the WV DEP Watershed Assessment Program [3] and my own lab to calculate WVSCI scores and water quality for stream segments of tributaries in the Cheat River basin. Benthic macroinvertebrate data for the Cheat River mainstem, however, currently is unavailable. Consequently, I used best professional judgment to assign WVSCI scores to mainstem reaches. Because values for ecological condition in the mainstem are unknown, the following analyses should be interpreted as hypothetical. Furthermore, it is important to realize that these analyses are for the purpose of demonstrating the analytical process of assessing cross-pollutant trades. My results should not be construed as verified results suitable for making regulatory decisions. I identified 455 perennial stream segments in the Cheat River basin downstream of Parsons. Of these, 209 segments are located in the upper Cheat River basin (i.e., upstream of the Pringle Run confluence) and the remaining 246 segments are located in the lower basin (i.e., from Pringle Run downstream to Cheat Lake). I observed significant differences in the SCI scores of stream segments located in the upper and lower basins (Figures 16-4 and 16-5). SCI scores for stream segments in the upper basin ranged from 65 to 93 with a mean score of 84. These values can be interpreted to represent the range of scores observed in a watershed with no mining impact. In contrast, SCI scores for lower basin streams ranged from 10 to 93 and possessed a mean score of 62 (Figure 18-4). These values represent a broad range of impact from acid mine drainage present in the lower Cheat River basin. Our analysis of stream SCI scores indicates that less than 10% of all stream segments in the upper basin can be classified as impaired, whereas nearly 80% of streams in the lower basin are impaired (Figure 18-5).

As a further summary of these data, Table 18-2 lists the poorest and highest quality stream segments in the lower basin. Again, it should be noted that values for the Cheat River mainstem are based on our best professional judgment at this time. This list illustrates the broad range of stream ecological condition present in the lower Cheat River basin. Almost all of the ecological impairment to streams in the lower basin is a direct result of AMD.

EcoUnits and Elements of the Restoration Timeline

I used estimates of SCI score and surface area (in acres) to calculate EcoUnits (i.e., EUI’s) for each stream segment in the watershed. Estimates of EUI’s were then used to calculate the following variables for each stream segment: Current Ecological Condition (CEC), Historic Ecological Condition (HEC), Potential Restored Condition (PRC), Total EcoUnit Loss (HEC–CEC), Potential EcoUnit Recovery (PRC–CEC), and Expected EcoUnit Legacy (HEC–PRC). Each variable was calculated separately for each stream segment and then summed across segments to obtain measures at a range of spatial scales.

HEC can be interpreted as the condition of stream segments prior to mining or thermal impact. I used the average condition of stream segments in the upper basin (WVSCI = 85) as an estimate of HEC. I defined PRC as the expected ecological condition of a stream segment following AMD remediation. Although very little data exist with regard to ecological recovery in AMD impaired watersheds, there is some indication that many waterbodies can recover up to 90% of the condition of reference streams (Petty et al., unpublished data). On average, however, observed recovery is somewhere around 80% of reference conditions. Consequently, I set PRC equal to 78, which is 80% of 85 or 80% of expected reference conditions. Observed conditions (CEC) were then used along with HEC and PRC to calculate measures of ecological loss, potential ecological recovery, and the expected ecological legacy following remediation.

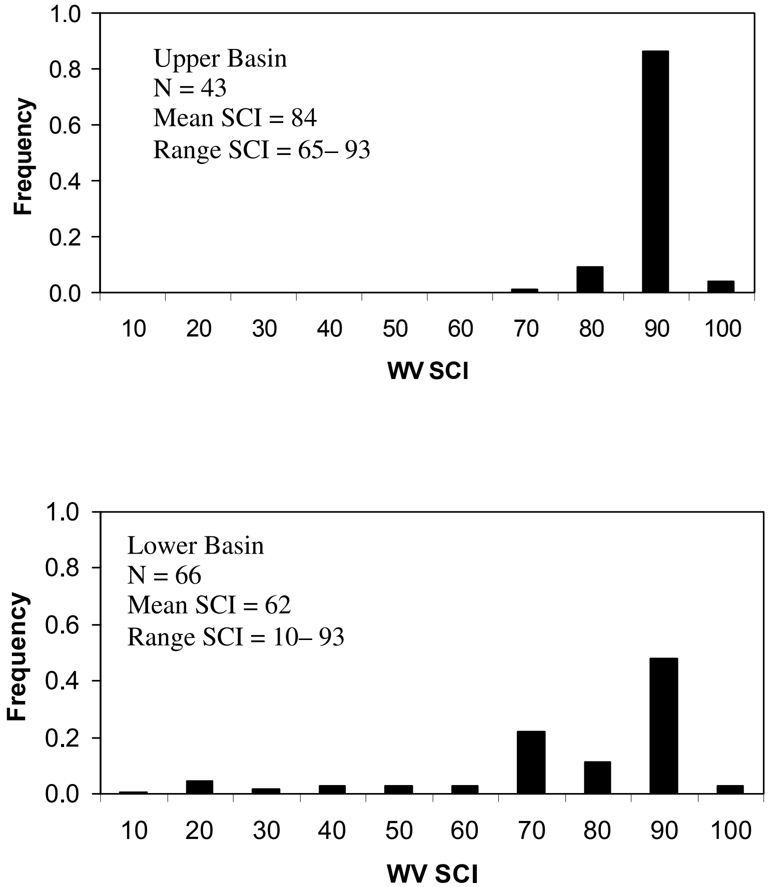

Figure 18-4
Frequency Distribution of WVSCI Scores from the Upper and Lower Basins

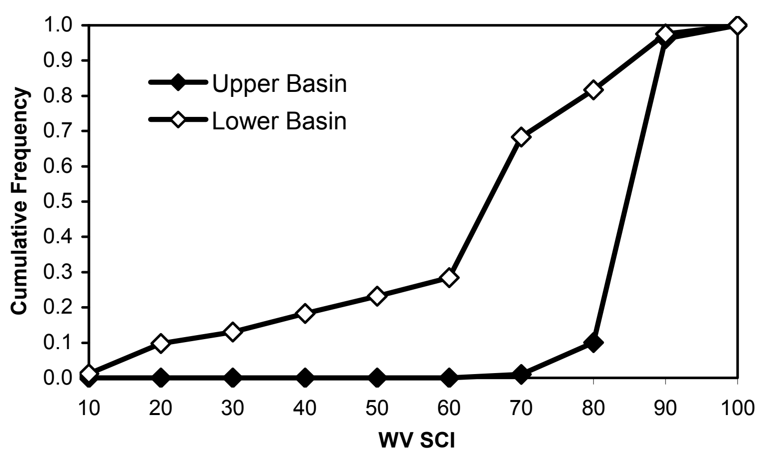

Figure 18-5
Cumulative Frequency Distribution of WVSCI Scores from the Upper and Lower Basins

Table 18-2
Poorest and Highest Quality Stream Segments in the Lower Basin as Determined by the WVSCI

Poorest Quality			Highest Quality		
Stream	**Rank**	**WVSCI**	**Stream**	**Rank**	**WVSCI**
Fickey Run	1	10	Upper Roaring Creek	1	93
Middle Fork Green's Run	1	10	Laurel Run	2	92
Cheat Riv (Muddy-Big Nasty)	1	10	Upper Muddy Creek	2	92
Cherry Run	4	13	Ryan Hollow	4	90
Martin Creek	5	14	Dority Run	4	90
Muddy Ck (below Martin)	5	14	Darnell Hollow	6	89
Lower Bull Run	5	14	Mill Run	7	88
SF Green's Run	8	15	Johnson Hollow	8	86
Glade Run	9	16	Long Hollow	9	85
Lick Run	10	17	UT Big Sandy Creek	9	85
Heather Run	11	19	Daugherty Run	10	84
Church Run	12	20	Elsey Run	10	84
Morgan Run	12	20	Gibson Run	10	84
Cheat Riv (Albright-Muddy)	12	20	South Laurel Run	10	84
5 others	12	30	Rubles Run	10	84

Source: WVSCI data are from [3] or Petty el al., unpublished data, Data from tributaries are real data, whereas Cheat River mainstem estimates are based on best professional judgment.

The entire Cheat River watershed possesses 4,699 acres of surface water (Table 18-3). My estimates indicate that 682 EUI's (in acres) have been lost from the watershed and of that 662 EUI's have been lost in the lower basin alone (Table 18-3). Full restoration of the watershed could expect to retrieve a little more than 500 EUI's. In addition to whole watershed calculations, Table 18-3 provides a summary of EUI values estimated for various watersheds at a variety of spatial scales. For example, Daugherty Run is a small, unimpaired watershed in the lower basin and accounts for 29 of 1427 EUI's currently present in the lower Cheat River basin. Furthermore, the lower Cheat River mainstem is a heavily impacted portion of the watershed and accounts for 320 of 662 EUI's that have been lost from the watershed as a result of mining related impact (Table 18-3).

Intensity and Extent of Ecological Loss at Multiple Spatial Scales

One of the most powerful uses of this approach is the ability to obtain relative measures of ecological loss within the watershed at different spatial scales. To facilitate this process, I calculated two separate measures of relative ecological loss for each stream segment in the watershed. I refer to the first measure as "Weighted EUI Loss" (WEL), which was calculated as:

WEL = 1–(Historic EUI/Current EUI) **Equation 18-3**

This measure can be interpreted as the proportion of historically available EUI's that have been lost from the stream segment. WEL is a measure of the intensity of ecological loss from a stream segment regardless of the size of the segment (i.e., the measure is standardized by surface area).

I refer to the second measure as "Relative EUI Loss" (REL), which was calculated as:

REL = Segment EUI Loss/Total Watershed EUI Loss **Equation 18-4**

Table 18-3
Current, Historic, and Potential Restored EUI Values at Various Spatial Scales

Sub-basin	Acres	WV SCI	Historic Eco Condition	Current Eco Condition	Restored Eco Condition	EcoUnit Loss	Pot. EcoUnit Gain	Expected EcoUnit Legacy
Daugherty Run	32	85	29	29	29	0	0	0
Beaver Creek	46	69	42	35	39	7	3	4
Roaring Creek	59	70	55	46	51	9	5	4
Green's Run	44	23	40	10	37	30	27	3
Muddy Creek	161	50	147	90	136	57	46	11
Lower Cheat River Mainstem	881	43	805	485	739	320	254	66
Lower Basin	2282	62	2089	1427	1931	662	505	168
Entire Watershed	4699	72	4306	3623	4131	682	508	174

Source: WVSCI data are from [3] or Petty el al., unpublished data, Data from tributaries are real data, whereas Cheat River mainstem estimates are based on best professional judgment. Note: All values, except "WV SCI" are in units of acres.

REL can be interpreted as the proportion of total ecological loss in the watershed that can be attributed to a given stream segment. REL summed across all segments is equal to 1. REL summed across all segments in a particular sub-watershed represents the proportion of total loss that can be attributed to that specific sub-watershed. Ultimately, the value is a measure of the extent of ecological loss associated with the focal stream segment or sub-watershed.

I believe that measures of "Weighted EUI Loss" and "Relative EUI Loss" will be invaluable in setting restoration priorities at both the watershed and stream segment scales. As an initial summary of these data, I used measures of WEL and REL to identify stream segments suffering the most intense ecological loss (i.e., greatest WEL) and the most extensive ecological loss (i.e., greatest REL) in the lower Cheat River basin (Table 18-4 and Table 18-5 respectively). Table 18-4 ranks stream segments on the basis of EUI loss intensity and illustrates that many of the most severely impacted streams have lost up to 90% of their historical ecological resources. Also, note that these segments represent a combination of small tributaries (e.g., Cherry Run) and larger water bodies (e.g., Cheat River mainstem and Bull Run).

Table 18-4
Stream Segments Suffering the Most Intense Loss of Ecological Resources (i.e., Greatest Weighted EUI Loss) in the Lower Cheat Watershed

Stream Name	Current EUI	Weighted EUI Loss	Relative EUI Loss
Cheat River (Muddy Creek–Big Nasty)	60	0.882	0.088
Middle Fork Green's Run	13	0.882	0.019
Fickey Run	5	0.882	0.007
Cherry Run	4	0.847	0.006
Muddy Creek (below Martin Creek)	25	0.835	0.040
Lower Bull Run	8	0.835	0.010
Martin Creek	6	0.835	0.008
South Fork Green's Run	6	0.824	0.008
Glade Run	5	0.812	0.004
Lick Run	7	0.800	0.010
Heather Run	5	0.776	0.008
Cheat River (Albright–Muddy Creek)	52	0.765	0.076
Morgan Run	88	0.765	0.007
Church Run	5	0.765	0.005
Green's Run	8	0.647	0.012

Source: [WVSCI data are from [3] or Petty el al., unpublished data, Data from tributaries are real data, whereas Cheat River mainstem estimates are based on best professional judgment.] "Weighted EUI Loss" is the proportion of historic EUI's that have been lost due to current environmental impairment. For example, 88% of the historic EUI's have been lost from the Cheat River mainstem below Muddy Creek. Values of "Relative EUI Loss," which is the proportion of total watershed scale EUI loss that can be attributed to a specific segment, are presented for comparison.

In contrast, Table 18-5 ranks stream segments on the basis of the extent of EUI loss. This table illustrates that the most extensive loss of ecological resources has occurred in the Cheat River mainstem and the larger tributaries. For example, by summing the Relative EUI Loss for the first four rows in Table 18-5, it can be calculated that the Cheat River mainstem from Pringle Run downstream to Big Sandy Creek is responsible for more than 27% of the total loss of ecological resources in the entire Cheat River watershed.

Bivariate plots with Total EUI Loss on the x-axis and Weighted EUI Loss on the y-axis provide another way to visualize the range of ecological impairment in the watershed (Figure 18-6). These plots can be used to identify stream segments characterized by both intensive local impairment and extensive EUI loss (upper right-hand quadrant of Figure 18-6). They also can be used to identify priority areas for restoration. For example, if the goal of a restoration program is to maximize the recovery of ecological resources, then restoration should focus initially on stream segments that are responsible for a large proportion of the total ecological loss in the watershed (i.e., stream segments in the right-hand quadrants of Figure 18-6). However, because I know that it may be difficult to fully restore streams suffering intense ecological losses, moderately impaired streams may take precedence over severely impaired streams (i.e., stream segments in the lower quadrants of Figure 18-6).

The process described above is useful for identifying priorities at the stream segment scale. A similar process can be used to prioritize waterbodies at the watershed scale (Figure 18-7). Measures of WEL and total EUI loss can be summed across all stream segments in a watershed to give watershed scale values. I have done this for several watersheds in the lower Cheat River basin and created a bivariate plot using these values (Figure 18-7). From this plot, I can see that the lower Cheat River mainstem is undoubtedly the highest restoration priority in the watershed. It possesses moderate levels of local impairment and extensive loss of ecological resources (Figure 18-7). Muddy Creek and Greens Run are two large, severely impaired tributaries of the Cheat River. Total EUI loss is more extensive in Muddy Creek than Green's Run, because it is a larger waterbody. Green's Run has suffered more intense ecological loss, because it is impacted by AMD essentially from its headwaters down to its confluence with the Cheat River. In contrast, the headwaters of Muddy Creek are not impaired and represent one of the highest quality regions in the lower Cheat River basin. Data for the Daugherty Run watershed, which possesses no mining impact, is presented for comparison.

Table 18-5
Stream Segments Suffering the Most Extensive Loss of Ecological Resources (i.e., Greatest Relative EUI Loss) in the Lower Cheat Watershed

Stream Name	Current EUI	Weighted EUI Loss	Relative EUI Loss
Cheat River (Muddy Creek–Big Nasty)	60	0.882	0.088
Cheat River (Albright–Muddy Creek)	32	0.765	0.076
Cheat River (Big Nasty–Big Sandy)	42	0.529	0.061
Cheat River (Pringle–Albright)	40	0.412	0.050
Muddy Creek (below Martin)	25	0.835	0.040
Little Sandy Creek	25	0.290	0.040
MF Green's Run	13	0.882	0.019
Green's Run	8	0.835	0.012

Source: WVSCI data are from [3] or Petty el al., unpublished data, Data from tributaries are real data, whereas Cheat River mainstem estimates are based on best professional judgment. "Relative EUI Loss" is the proportion of total EUI loss in the watershed that can be attributed to that specific stream segment. For example, 8.8% of the total ecological loss in the Cheat River watershed can be attributed to the Cheat River mainstem from Muddy Creek to the "Big Nasty" rapid. "Weighted EUI Loss" for each segment is presented for comparison.

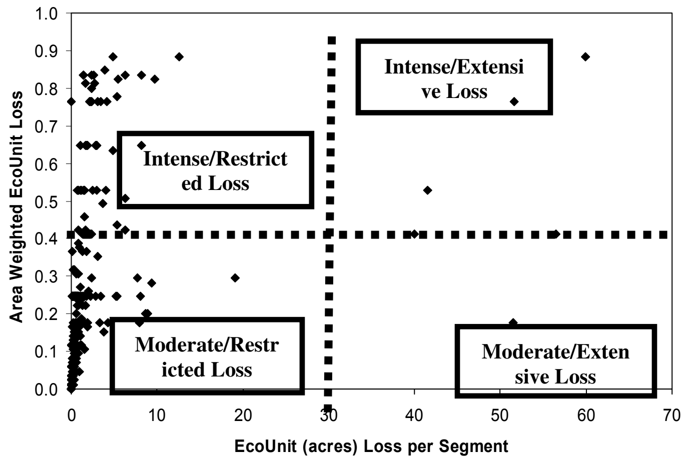

Figure 18-6
Relationship Between Total EcoUnit Loss in a Stream Segment and the Intensity of Ecological Impairment (i.e. Weighted EcoUnit Loss)

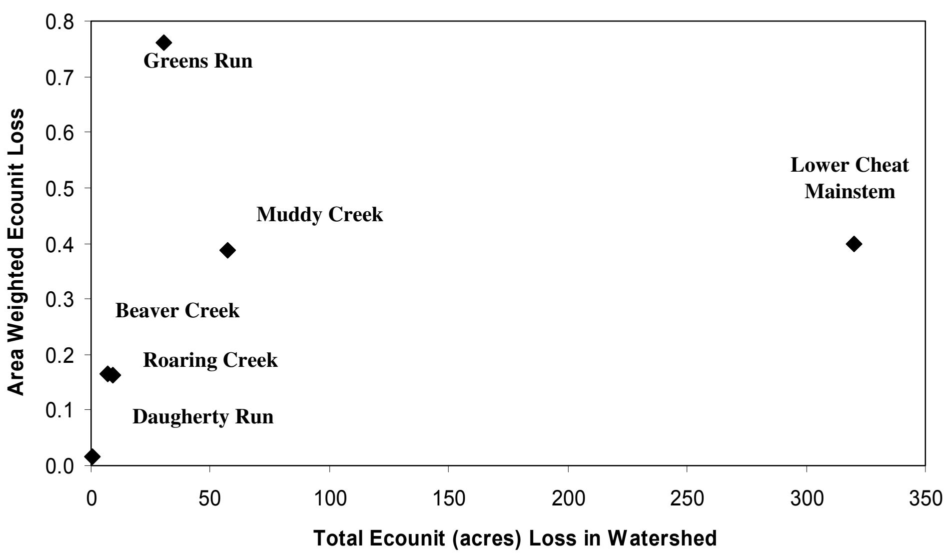

Figure 18-7
Relationship Between Total EcoUnit Loss in a Watershed and the Intensity of Ecological Impairment (i.e. Weighted EcoUnit Loss)

Equivalencies between EcoUnits, Environmental Stressors, and Dollars

Our ability to develop effective cross-pollutant trading programs is wholly dependent on our ability to calculate equivalencies between ecological condition and different environmental stressors. In this section, I demonstrate how the EcoUnit currency can be used to calculate equivalencies between ecological condition, environmental stressors and dollars. Such equivalencies can then be used to determine how much a particular trader must pay to ensure Net Ecological Benefit in the watershed.

To develop equivalencies, I have focused on AMD from active and abandoned mines and thermal effluent from an electrical power plant. I focus on AMD and heat for several reasons. First, AMD is responsible for most of the ecological impairment in the Cheat River watershed. Second, the existing TMDL for the Cheat River focuses on AMD. Third, thermal effluent likely has a measurable effect on the ecological condition of the Cheat River. Fourth, the most likely trade scenario in the near future would involve trading thermal mitigation requirements for AMD remediation.

Consequently, I conducted our analyses to calculate the following equivalencies:

1. EUI equivalent of 1 ton per year (tpy) Acidity added to the watershed
2. EUI equivalent of 1 tpy Acidity removed from the watershed
3. Dollar equivalent of 1 tpy Acidity removed from the watershed
4. Dollar equivalent of 1 EUI recovered through AMD remediation
5. EUI equivalent of the Thermal Effluent
6. Acidity equivalent of the Thermal Effluent
7. Dollar equivalency between Acidity and Thermal Effluent

The results of these analyses are presented in Table 18-6. I again need to emphasize that the values presented are for demonstration purposes only because stream condition scores for the mainstem are not based on real data but rather on best professional judgment. Given this caveat, I found that 0.036 EUI's are lost, on average, for every 1 tpy Acidity added to the watershed. Because of the imperfect nature of AMD remediation, however, I expect only 0.029 EUI's to be gained for every 1 tpy Acidity removed through AMD remediation actions (Table 18-6). Based on WVSCI values used for the Cheat River mainstem, 32 EUI's were attributed to thermal effluent. Given this impact, the thermal effluent has an ecological effect that is equivalent to 1,110 tpy Acidity. Ultimately, given the cost of AMD remediation ($300/ton Acidity/year), I calculated the dollar equivalency between thermal effluent and AMD to be $333,087/year.[1]

It is critical to realize that the equivalencies I calculated represent values that are averaged across the entire Cheat River watershed. For example, every single ton of Acidity probably does not produce an EUI loss of 0.036. Some acid sources may produce no detectable effect on biological resources. Other acid sources may produce complete ecological losses. The ultimate effect of 1 tpy Acidity is influenced by numerous factors, but the most important factor is the chemical properties of the receiving body. For example, 1 tpy Acidity that enters a stream with a high alkaline load may have little or no ecological effect on the stream. However, that same load of Acidity entering a stream with no alkaline load may have a devastating effect. This same issue applies to our calculation of the EUI equivalent of 1 tpy Acidity removed, as well as our calculation of the cost equivalent of Acidity removal. In all cases, our estimates are based on watershed scale averages and can be interpreted as our expectation of ecological gains and costs of a full, watershed scale restoration program.

[1] Such an equivalency does not take into account trading ratios, which, if greater than 1:1, would require a greater investment. Also, these calculations should be viewed as preliminary and should not be considered as justification for or against a trading program. These values are intended for demonstrative purposes only.

Table 18-6
Summary of Equivalency Calculations

Current Ecological Condition (EUI's)	485 acres
Historic Ecological Condition (EUI's)	805 acres
Ecological Losses (EUI's)	
Total	320 acres
Heat	32 acres
AMD	288 acres
Recoverable EUI's	
Total	263 acres
Heat	32 acres
AMD	231 acres
Acid Load (tpy Acidity)	8,000 tpy
EUI's Lost per Acid tpy Added	0.036 acres
EUI's Gained per Acid tpy Removed	0.029 acres
AMD–Heat Equivalency (tpy Acidity)	1,110 tpy
Cost to Treat Acidity ($/tpy Acidity)	$ 300/year
Cost to Treat Total Acid Load ($/year)	$ 2,400,000/year
Cost to Recover 1 EUI via AMD Remediation ($/year)	$ 10,409/year
Cost Equivalency between Heat and AMD	$ 333,087/year

Note: Equivalencies were calculated for the following contrasts: Ecological Loss from AMD vs. Heat; Ecological Gain from AMD remediation vs. Dollars; Cost Equivalence between Heat Remediation and AMD (i.e., Cost for AMD remediation needed to offset ecological loss from heat). These equivalencies were calculated for the lower Cheat River mainstem only. TPY refers to tons per year of acidity. Please note that all values are preliminary and should be viewed simply as conceptually representative of the value of the ecological condition accounting approach. Also, the value of $300/year to treat 1 tpy Acidity is simply an estimate of the average cost. The true average cost to treat AMD in the watershed is currently unknown.

A second critical issue is that all equivalencies calculated are specific to the current conditions of the watershed, both in terms of acid loads and EUI losses. This is especially true for equivalencies relating thermal effluent to acidity and thermal effluent to dollars to treat acidity. In the event that acid loads and/or the ecological condition landscape change, equivalencies among EUI's, stressors, and dollars will also change.

These issues lead us to two important conclusions. First, trading programs will necessarily be watershed scale specific. The EUI equivalent of 1 tpy Acidity probably is not constant across all AMD impacted watersheds in West Virginia. Second, successful trading programs will require regular sampling of water chemistry and ecological condition at the watershed scale. This sampling will focus on providing information needed to update equivalencies among EUI's, stressors, and dollars.

Calculating Net Ecological Benefit of a Thermal for AMD Trade

The EcoUnit Index (EUI) was designed explicitly for use in quantifying ecological costs and benefits of alternative management actions. In this section, I present the results of analyses comparing the relative ecological benefits of a cross-pollutant trade scenario involving the trade of thermal effluent impacts for reductions in AMD within the Cheat River watershed.

To conduct this analysis, I calculated the total number of EUI's that have been lost from the lower Cheat River mainstem. I then calculated the number of lost EUI's that can be attributed to AMD pollutants and thermal effluent separately. I then looked at the number of EUI's that could be regained through a series of management options, which included: no action, heat remediation only, AMD remediation only, and heat and AMD remediation together.

Comparison of the expected ecological gains associated alternative mitigation actions in the lower Cheat River mainstem are presented in Figure 18-8. These analyses were conducted at the stream segment scale and the scale of the entire lower mainstem. The 3-km reach immediately downstream of the Albright Power Station was used as the focal reach for reach scale analyses. Because thermal effluent is the dominant factor limiting ecological condition in this segment, significant improvements at the reach scale require mitigation of the thermal effluent. For example, AMD remediation is expected to produce negligible gains in ecological condition at the reach scale, as compared to an increase in 32 EUI's following heat remediation (Figure 18-8). In contrast, because AMD is the dominant limiting factor at a watershed scale, AMD remediation will produce the most pronounced improvements at the larger spatial scale (i.e., at the scale of the entire lower mainstem) (Figure 18-8). Specifically, AMD remediation is expected to increase EUI's by more than 200 at the mainstem scale as opposed to an increase of only 32 EUI's following heat remediation only. Consequently, at the mainstem scale, AMD remediation is expected to produce an increase in EUI's that is more than seven times the recovery expected from heat mitigation alone.

As a further analysis, I compared the intensity and extent of ecological loss associated with alternative mitigation actions in the lower Cheat River mainstem (Figure 18-9). These analyses support the conclusion that AMD remediation is needed to produce significant levels of ecological recovery at larger spatial scales, whereas mitigation of thermal effluent is needed to produce significant recovery at the reach scale.

Conclusions

Cross-pollutant trading provides an opportunity to produce significant improvements in the overall ecological condition of severely impaired watersheds like the Cheat River basin. Cross-pollutant trading can allow managers to focus restoration efforts and money on environmental stressors that represent the dominant factors limiting watershed health. In the Cheat River basin, AMD related pollutants are clearly the most important factors limiting the ecological condition of the watershed. In fact, nearly half of the historic condition of the lower Cheat basin has been lost, and 90% of that loss can be directly attributed to AMD. In this report, I present an analytical approach that makes it possible to place secondary stressors in a watershed into context with the dominant stressors. In this case, I compared the ecological effects of thermal effluent to those of AMD. I also developed a procedure for calculating ecological and cost equivalencies between thermal effluent and acidity. Through these calculations, it is possible to quantify the ecological costs and benefits of following strict regulation of thermal effluent and compare this to an alternative mitigation scenario, which would allow AMD remediation in lieu of heat reductions. In doing so, various stakeholders in the Cheat River watershed can make more informed decisions about the expected costs and benefits of a cross-pollutant trading program.

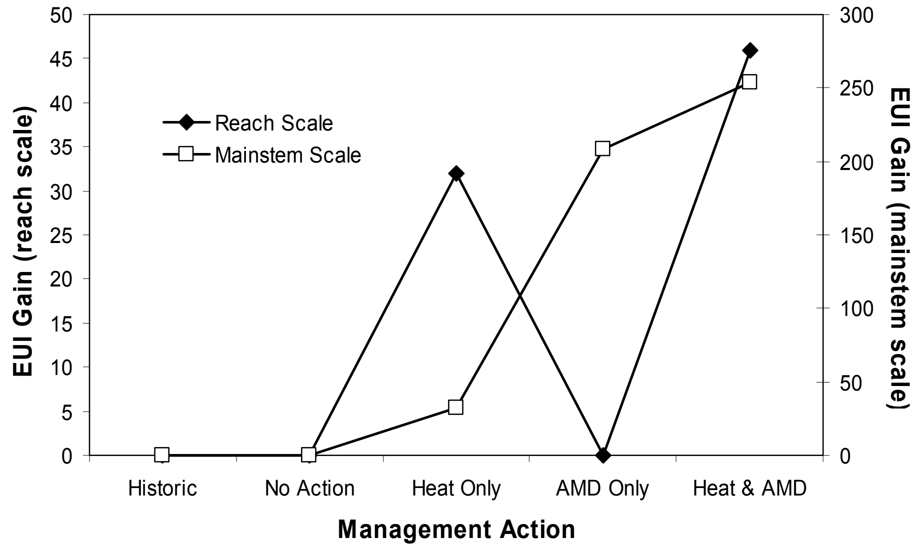

Figure 18-8
Variation in Expected EUI Gains as the Result of Various Remediation Options

Several additional conclusions extend from the analyses presented in this report:

- A cross-pollutant trading program must target dominant factors limiting ecological condition in a watershed (AMD pollutants in the case of the Cheat River).
- The trading program must be managed at a watershed scale to produce meaningful benefits, and therefore should be part of a holistic watershed management plan.
- Quantifying ecological equivalencies between dominant and subdominant stressors is critical to a cross-pollutant trading program. Subdominant stressors for which equivalencies cannot be estimated should not be included in such a program. Without equivalencies, it is impossible to predict ecological gains and losses expected from alternative mitigation options, and therefore it is impossible to justify a cross-pollutant trade.
- Variables used to calculate EUI's probably will vary from watershed-to-watershed.
- The procedure presented here requires large amounts of information on the physical, chemical, and biological conditions of the target watershed. In order to facilitate cross-pollutant trading, industry, watershed groups, and regulatory agencies will need to invest in data collection that can be used to quantify the current condition landscape and calculate ecological and cost equivalencies among primary and secondary stressors in watersheds. Without this information, trading programs cannot be implemented.

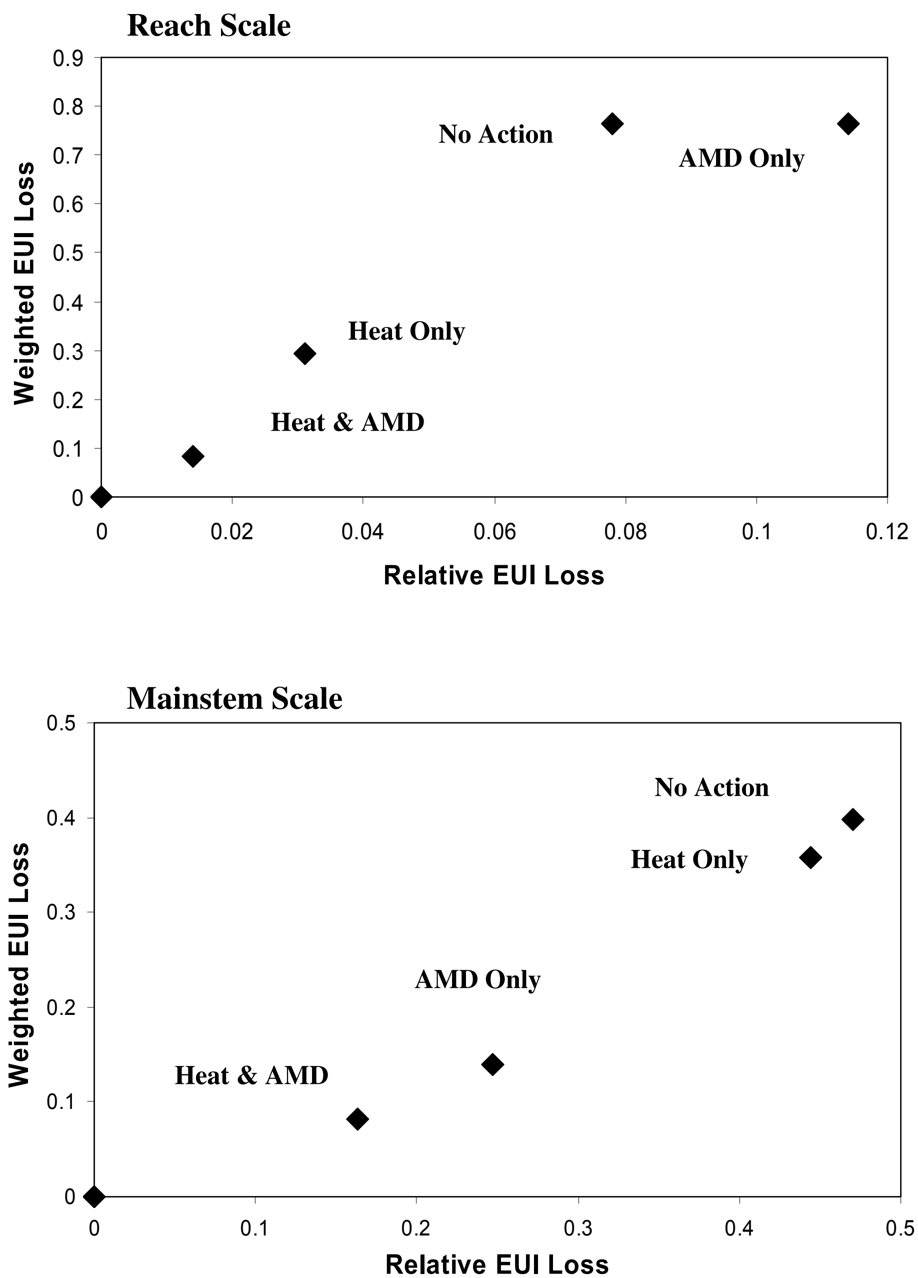

Figure 18-9
Variation in the Intensity of Ecological Loss (i.e., Weighted EUI Loss) and Extent of Ecological Loss (i.e., Relative EUI Loss) as the Result of Various Remediation Options

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A

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B

WORKSHOP AGENDA

EPRI Workshop on 316(a) Issues: Technical and
Regulatory Considerations
October 16 – 17, 2003
AEP Headquarters
Columbus, Ohio

Purpose of workshop: Provide participants with an overview of technical and regulatory issues and considerations concerning Section 316(a): Effects database; water quality criteria issues; strategies for 316(a) variances; instream assessment of thermal effects; permitting; cooling technologies; trading; and TMDL considerations. The workshop is intended to provide a forum for information exchange among participants, presentation of case studies, and an update on the current knowledge of thermal physiology and thermal effects.

Participants: Open to EPRI members, non-EPRI members, regulatory agency representatives, other interested parties. In case of seating limitations, EPRI members will be accommodated first.

Thursday, October 16

Continental Breakfast	7:00-8:00
Moderator for Sessions I, II, & III: Rob Reash (AEP)	
I. Introductions and workshop overview - Bob Goldstein (EPRI) and Rob Reash (AEP)	8:00-8:15
II. Been there, done that: a legal perspective on 316(a) implementation since passage of the FWPCA of 1972 - Kristy Bulleit (Hunton & Williams)	8:15-8:45
III. Regulatory perspectives on 316(a)	
Development of Thermal Water Quality Standards and Point Source Implementation Rules in Wisconsin - Michael Wenholz (Wisconsin DNR)	8:45-9:15
Updating the methodology used to establish temperature criteria for Ohio's rivers and streams and the Ohio River mainstem - Chris Yoder (Midwest Biodiversity Institute) and Erich Emery (ORSANCO)	9:15-9:45

Morning Break	9:45-10:15
Overview of CWA Section 316(a) Evaluations of Power Plants with Thermal Discharges in Maryland- Steve Schreiner (Versar) and Rich McLean (Maryland DNR)	10:15-10:50

Moderator for Sessions IV & V: Randy Lewis (Cinergy)

IV. Thermal physiology and effects: An update on the science

Determining the Relationship between Temperature and Biological Effect: An Update on the Science - Charles Coutant and Mark Bevelhimer (Oak Ridge National Laboratory)	10:50-11:25

V. Measuring community balance and thermal effects

Considerations of study design, sampling protocols, and data interpretation in measuring community balance in the field - Greg Seegert (EA Engineering Science & Technology)	11:25-12:00
Lunch	12:00-1:00

Afternoon Session Moderator: Doug Dixon (EPRI)

VI. Site-specific cases studies

In a nutshell: 15 years of biomonitoring and 316(a) permitting at two AEP facilities on the Muskingum River - Rob Reash (AEP)	1:00-1:25
Fisheries studies near Cinergy generating stations on the Wabash River - Randy Lewis (Cinergy)	1:25-1:50
Peach Bottom Atomic Power Station: Show and Tell (or Temperature Shock and Awe) - Dilip Mathur (Normandeau Associates)	1:50-2:15
RFAI: We almost made a silk purse out of that sow's ear- Larry Olmstead (Duke Power, retired)	2:15-2:40
Afternoon Break	2:40-3:00
Application of Multimetric Bioassessment Techniques in a 316(a) Demonstration at Georgia Power Company's Plant Branch, Lake Sinclair, Georgia - Terry Cheek (GeoSyntec Consultants) and Bill Evans (Georgia Power)	3:00-3:25
Cooling lake issues - John Humes (Hoosier Energy)	3:25-3:50
Managing cooling lake fisheries in partnership with Illinois Department of Natural Resources - John Petro (Exelon)	3:50-4:15

PSEG 316(a) Study experience at Salem and Hudson generating stations - John Balletto (PSE&G)	4:15-4:40
Probability-based impact assessment for a 316(a) demonstration: an example from Entergy Nuclear Vermont Yankee – Mark Mattson (Normandeau Associates), Paul Harmon (Normandeau Associates), Craig Swanson (Applied Science Associates), Mark Hutchins (Hutchins Consulting), and Lynn DeWald (Entergy)	4:40-5:05
Recent Experiences Renewing 316(a) Variances in West Virginia, Maryland and Pennsylvania – Richard Sands (URS) and Joe Lapcevic (Allegheny Energy)	5:05-5:30
Moderated Panel Discussion: Doug Dixon and afternoon speakers	5:30-6:00
Reception and Buffet Dinner at AEP Cafeteria	6:00-8:00

Friday, October 17

Continental Breakfast	7:00-8:00
Friday Morning Session Moderator: Bob Goldstein	
VII. Thermal modeling and cooling technologies	
Modular cooling towers and other technologies for cooling – Billy Childers (AGGREKO, Inc.)	8:00-8:30
Economic and environmental impacts of thermal discharge requirements - John Veil (Argonne National Laboratory)	8:30-9:00
Thermal modeling and applications: from the 1960s to now - Bill Mills (Tetra Tech)	9:00-9:30
VIII. Thermal trading and watershed enhancement/mitigation approaches	
Environmental enhancement and restoration as part of a thermal discharge mitigation strategy - John Veil (Argonne National Laboratory)	9:30-10:00
Morning Break	10:00-10:30
Thermal trading at Allegheny Albright Station - Todd Petty (WVU)	10:30-11:00
IX. Discussion and Summary - Bill Mills (Tetra Tech)	11:00-12:00
Summary/open questions & answer forum: Identify technical and regulatory issues that need further resolution or development	

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Section 316(a) and 316(b) Fish Protection Issues

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