CHAPTER 3

Biological Assessment and Criteria: Building on the Past

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1.0 INTRODUCTION

Use of ambient biological communities, assemblages, and populations to protect, manage, and even exploit water resources have been developing for the past 150 years. Although precise analytical tools available to the water resource scientist have become more sophisticated, direct measurements of plants, invertebrates, fish, and microbial life are still used as indicators for sanitation, potable water supplies, protection of fisheries, and recreation (McKenzie et al. 1993). With increasing natural resource demands and ecological protection needs anticipated for the twenty-first century, water resource scientists, engineers, managers, and planners must use ambient biological assessments and criteria to protect water resources in an economically and environmentally sound manner (USGAO 1991; ITFM 1992; Adler et al. 1993; NRC 1993).

Since the 1960s, many natural resource, land management, and regulatory agencies have recognized the importance of ambient biological assessments for managing and protecting water resource quality (Burgess 1980; Weber 1980). Unfortunately, without a widely acceptable technical framework for using biological assemblage data, the majority of water resource management decisions relied solely upon surrogate measures of the aquatic community. Such measures include toxicity testing, tissue chemistry and comparisons with chemical criteria through direct measurements and modeled predictions. Despite recent demonstrations of the successful uses of ambient biological criteria (USEPA 1991a; Southerland and Stribling, Chapter 7), there is still a great deal of misunderstanding and concern regarding the application of biological assessments at many levels of management (Suter 1993; Karr 1993a,b). Much of this skepticism comes from personal and professional experience with using, or trying to use, biological assemblage data to assess water resource quality and the great difficulty biologists had (which many still do) with expressing the results in a meaningful, objective, and consistent manner. In other cases little attempt was made to use biological assessments and the potential value of the data were not recognized.

This chapter is intended to provide a better understanding and appreciation of how far numeric biological assessment and criteria development has progressed since the early days of stream pollution biology. This chapter also introduces four tools developed in the past decade that now allow us to transform biological assemblage data into numeric criteria and standards: (1) a functional definition of biological integrity to serve as an understandable water resource goal; (2) minimizing the problems with interpreting the natural geographic and temporal variability of data by aggregating within regions of ecological similarity (Omernik, Chapter 5); (3) using multiple reference sites within ecological or faunal regions to obtain assemblage expectations, or reference condition, for specific geographic areas (Hughes, Chapter 4); and (4) combining several assemblage attributes (or metrics) to produce a single numeric

Table 1. Milestones in Biocriteria Development

Year	Event
1894	First biological stream monitoring station established to study pollution effects (Illinois River)
1901	Concept of biological classification systems proposed
1908	Indicator organisms for water pollution established
1913	Report linking river pollution with effects on aquatic life
1949	Histogram approach for numerically displaying aquatic community response to pollution
1954–5	Numeric biotic indices for benthic macroinvertebrate assemblages (Beck's biotic index and saprobic index of Pantle and Buck)
1966	Shannon–Wiener diversity index applied to pollution biology
1976	Index of Well-Being for fish assemblages (combined the Shannon-Wiener diversity index with the numbers and weight of the fish)
1981	Index of Biotic Integrity for fish assemblages (multiple metric approach using attributes of the fish community)
1984	Maximum species richness lines to set metric criteria for IBI
1987	Publication of ecoregions (although used as early as 1982)
	Ten metric Invertebrate Community Index developed for Ohio
	Regional reference conditions for biocriteria established in Ohio
1989	Rapid Bioassessment Protocols for benthos and fish
1990	Numeric biological criteria adopted into Ohio water quality standards
1993	EPA Science Advisory Board Review of Draft technical guidance document for biocriteria development in wadable rivers and streams

measure of biological integrity (Barbour et al., Chapter 6). Table 1 presents some of the key conceptual or technical advances that have occurred in this century and shows how recent many of our most important advances have been.

2.0 EARLY INDICATORS OF WATER POLLUTION

Water pollution is not a phenomenon of the twentieth century. Detailed observations of the effects of pollution upon aquatic life and human health have been made for over 150 years. This section highlights some early works of scientists who tried to make others aware of the effects of civilization's rapid progress upon not only natural systems, but also on human health.

2.1 Recognizing Pollution

The classic studies of Chadwick (1842; Flinn 1965), Hassall (1850) and Cohn (1853) have often been credited as the first to use aquatic organisms as indicators of environmental pollution. These researchers documented the relationship among human illness and poor sanitary and drinking water conditions in the mid-1800s. Their efforts led to the development of the first set of national legislation addressing water pollution in Great Britain (Alexander 1876). These early efforts were the basis for the bacteriological tests still used today for protecting against human illness due to contaminated potable and recreational waters (Wilson and Miles 1946; APHA et al. 1993).

Severe reduction of river and stream fisheries were also recorded in England. Fisheries in the rivers Mersey and Irwell declined until the early 1800s when all aquatic life virtually disappeared (Klein 1957). Previously abundant salmon were sufficiently depleted in the River Thames by 1833 due to river pollution causing all commercial fishing to cease by 1850 (Fitter 1945). Recognition of cause and effect between sources of pollution and impacts upon aquatic life led to early attempts to remedy the problems. One of the first documented cases of stream recovery based on ambient biological communities occurred in the river Soar at Leicester in England. The River Soar was reported to be a "common sewer for the drainage of this town" in the late 1700s and in the 1830s "the Soar became so corrupt that fish could not live in them and consequently disappeared entirely" (Chesbrough 1858). After altering the drainage patterns of the town by installing a new sewer system, which discharged wastes further downstream, the river started its recovery and "such a purification of the river as to restore the water to its original clearness, and cause the reappearance, in the summer of 1856, of fish which had not been seen in it for twenty-years before" (Chesbrough 1858).

2.2 Contributions of Early Naturalists

Along with the exploration and westward expansion of the United States during the late 1700s through the middle 1800s, detailed studies and faunistic surveys provided the groundwork for some of the first intentional studies using aquatic life to gauge existing and encroaching pollution. Without documenting the distribution and abundance of the living resources it would have been difficult to determine the effect of natural or man-made influences. Naturalists tediously recorded the presence and distribution of existing and new species of terrestrial, aquatic, and semiaquatic life, most notably through the exploration of the Ohio River valley by the original "boatload of knowledge" (Frost and Mitsch 1989; Pitzer 1989). One of the first ichthyologists to visit the Ohio River basin was C.A. LeSueur in 1817 and 1818 (Trautman 1981). LeSueur (1827) later published the classic *American Ichthyology*. Rafinesque (1820) described over 100 "new" species, although many of them were later shown to be the same as other known species. Kirtland (1838) began his studies on the fishes of Ohio. Jordan surveyed the eastern states and published several reviews of midwestern fish between 1876 and 1891 (Trautman 1981) and Forbes began surveying the fishes of Illinois in the 1870s (Forbes and Richardson 1908).

While adverse effects upon the fish, wildlife, vegetation, and other aquatic life were documented, their use as environmental indicators was purely qualitative. For example, Hildreth (1848) noted the destruction of riparian vegetation in the midwestern United States due to habitat alteration from farming and logging as well as the influence of agricultural water use on Ohio rivers and streams. Kirtland (1850) documented the westward advancement of "progress" in northeastern Ohio between 1797 and 1850 stating that "the whole face of nature has been changed." He described the reduction in the great sturgeon and muskellunge populations already occurring by 1850 and commented that "many smaller species have increased in all our waters...the slaughter houses about the river, afford them large supplies of food and contribute to their increase." The relationship between pollution and undesirable effects upon fisheries was firmly understood by American naturalists by the mid-1800s.

Charles Darwin's (1859) account of his travels on H.M.S. *Beagle* facilitated the movement of natural biologists away from simply enumerating the distribution of populations in faunistic surveys to a fascination with natural selection and how populations respond to natural and anthropogenic changes. It was Stephen Forbes* who initiated some of the most important and fascinating work in aquatic biology and pollution effects. Forbes (1887) built upon Darwin's concept of adaptation and natural selection to develop ecological principles based upon the interrelationships of aquatic populations in his classic presentation entitled "The lake as a Microcosm."

2.3 Illinois River Biological Station

Forbes' insight and application of the principle of natural selection led to the establishment of a biological station on the shores of the Illinois River in 1894 (Bennett 1958). This was the first laboratory designed to assess the effects of pollution on aquatic life. "The general objects of our Station are to provide additional facilities and resources for the natural history of the state...especially with reference to the improvement of fish culture and to the prevention of a progressive pollution of our streams and lakes" (Forbes 1895). In retrospect, after nearly 25 years of operating the Havana Biological Station, Forbes (1928) specifically defined the stations two main objectives: "...one the effect on the plant and animal life of a region produced by the periodic overflow and gradual recession of the waters of great rivers...and the other the collection of materials for a comparison of chemical and biological conditions of the water of the Illinois River at the time then present and after the opening of the sewage canal of the Sanitary District of Chicago which occurred five years later" (in 1900). The biological station would later provide the data and information to document the damage to the health of the Illinois River due to the opening of the Chicago Drainage Canal (Kofoid 1903, 1908; Purdy 1930) despite "authoritative" claims to the contrary (Leighton 1907; Randolph 1921).

* Director of the Illinois State Laboratory of Natural History from 1872 to 1917, later renamed the Illinois Natural History Survey, which he directed from 1917 until his death in 1930.

3.0 BIOLOGICAL STREAM CLASSIFICATION

While the Illinois State Laboratory of Natural History began its systematic collection of aquatic life in 1894, the use of indicator organisms to help classify trophic status of rivers and streams was also developing in Europe. Development and application of the first stream classification system was based upon the responses of aquatic life to organic enrichment forming distinct longitudinal "zones" in streams. This "saprobien system" eventually resulted in the development of numerical indices for describing community and assemblage structure (and sometimes function) based upon tolerance or sensitivity to types of pollution.

3.1 Saprobien System

Robert Lauterborn (1901) is credited with originating "the conception of the sapropelic* world of life" by defining the saprobic zone in running waters as the "zone of processes of decomposition." The following year, Kolkwitz and Marsson (1902) presented a historical account of a "new" saprobien system based on indicator organisms (in this case plankton) to classify streams. These German biologists identified three specific stages, which they referred to as saprobic zones, of progressive decomposition to classify slow-moving waters affected by sewage: polysaprobien, mesosaprobien, and oligosaprobien. A fourth zone, the katharobien, was defined as unaffected by the decomposition and essentially "pristine." Kolkwitz and Marsson (1908, 1909) published the first extensive lists of indicator organisms associated with the individual zones of contamination. They first reported the associated saprobien zones (i.e., tolerances) of about 300 species, predominantly benthic and planktonic plants. They later added to the list over 500 planktonic and benthic animals (mostly zooplankton and bacteria) to the list.

Many studies have confirmed the value of this type of a classification system. Strong supporters of the saprobien system in the United States included Stephen Forbes and Robert Richardson (1928) who began publishing reports on the conditions of the Illinois River. They defined the river's degradation via pollutional zones (septic, polluted, contaminate, and clean water) similar to those of Kolkwitz and Marsson. Forbes and Richardson's zones were based upon "integrated" studies of water chemistry, plankton, benthic macroinvertebrate, and fish populations. Their pollutional surveys conducted prior to 1911 documented that 107 miles of river were polluted below the mouth of the Chicago Drainage Canal (Forbes and Richardson 1913).

When the United States Public Health Service Act of 1912 required the Public Health Service to conduct studies on the sanitary condition of interstate waters, Cumming (1916) initiated this effort in 1913. He used three "stages of impurity" and "clean water" based on the saprobien system to describe the aquatic health of the Potomac River watershed. Demonstration of the severity and sources of pollution in the river led to wastewater treatment for the Washington, D.C. area. The Public Health Service studied the Ohio River in 1914 to 1915 and used a classification system based on plankton abundance resulting in four possible ratings of pollution for each sample site: pollution abundant, moderate pollution, slight pollution, and pollution absent (Purdy 1922).

The Public Health Service later studied the Illinois River in 1921 and 1922 but relied upon Richardson's studies to fully document the effects of pollution on the aquatic life. Richardson (1921a, b) defined pollution zones and tolerances for the biota, focusing primarily on the benthic macroinvertebrate community. He documented 146 miles of polluted river between 1913 and 1915, and 226 miles of polluted conditions in 1920, including 146 miles of near anoxic conditions. The last report on the pollution biology of the Illinois River conducted by Richardson was published in 1928, which also marked the change of responsibility for the river surveys within Illinois to the State Water Survey (Bennett 1958). In this classic study, Richardson (1928) detected shifts in water quality based on observations of the benthos alone, although he also used chemical data to better define the pollutional zones. He further refined the pollutional, or saprobic, zones (called septic, pollutional, subpollutional, and clean water) based on "index values" of the benthos using specific taxa as indicators.

Many early authors used indices similar to the saprobien system to classify running waters according to biological effects from pollution (Ingram et al. 1966; Mackenthun and Ingram 1967). Some authors

* From the Greek *sapros* meaning rotten or dead. *Webster's New World Dictionary* defines saprobic as "of or pertaining to organisms living in highly polluted water."

recommended using communities of microorganisms (Liebmann 1951; Fjerdingstad 1965) or adopting more levels to reflect additional pollution ratings such as toxic and radioactive conditions in addition to organic enrichment and decomposition (Fjerdingstad 1960; Slàdeček 1965). For example, Kolkwitz (1950) published a revised saprobien system resulting in a total of seven saprobic zones. [For a critical review of the saprobien system, see Bick (1963), Fjerdingstad (1964), Friedrich (1990), Friedrich et al. (1992), and Ghetti and Ravera (1994).]

3.2 Critical Issues with Indicator Organisms

Although widely used, the saprobien system and the general concept of indicator organisms met with a great deal of criticism. Doudoroff and Warren (1957) stated their doubts about the saprobien system and the indicator organisms proposed because: (1) they reflected only pollution from sewage wastes, (2) the relationship among organisms and pollution tolerances were not well studied, and most importantly (3) the indicator organisms used were generally not reflective of economic value. The authors were concerned about the lack of attention biologists paid to economically valuable species: "[t]hey [some biologists] seem to have curiously attached at least as much importance to the elimination of any species of diatom, protozoan, rotifer, or insect as to the disappearance of the most valuable food or game fish" (Doudoroff and Warren 1957). They strongly advocated the study of economically valuable fish species and the use of toxicity testing as better indicators of water quality.

There were also doubts about assigning a single saprobic zone, or indicator value, to a species that could also be found outside of that zone. Brinkhurst (1969) stated that "I can see no way in which different saprobity values could be given to each species to account for its reactions to the many different forms of polluting materials...I find the systems...less efficient than the opinion of a qualified biologist expressed in plain language." Fjerdingstad (1964), Hynes (1965), and Cairns (1974) were also critical of using indicator species and they all advocated a community-based approach instead. Hynes favored using benthos while Fjerdingstad was convinced that attached algae (diatoms) were the superior group since they were not subject to stream drift.

Hawkes (1957) summarized concerns regarding the saprobien system and the manner in which specific biological indicators were perceived to be used as follows:

> [i]n using this system it must be borne in mind that factors other than pollution affect the nature of stream communities. The absence of organisms may be a more important indication than the presence of other species. The community of organisms should be taken into consideration rather than the presence or absence of one or few 'indicator organisms'. In some cases specific identification is essential, in others the genera or the whole family may be indicative. Knowledge of stream ecology is continually advancing and no doubt the list will be modified and extended in light of this knowledge.

Even as we have progressed in our understanding of the limitations of using indicator organisms to measure water resource quality, indicator species continue to be used as "integrators" (Ryder and Edwards 1985) and as components of modern ecological health indices (Cairns 1993; Karr 1993a).

3.3 Importance of the Saprobien System

Although Bartsch and Ingram (1966) felt that the saprobien system was purely a European tool that was not used "to determine the existence and magnitude of pollution, but to obtain colorful biological data for an anti-pollution campaign," the saprobien system was very important in focusing biologists on measuring water resource quality by the presence or absence of a wide range of indicator biota. This approach became more sophisticated by eventually relying upon communities or specific assemblages (e.g., benthos) and attention to their relative abundance and distributions. In fact, the saprobien system is widely used outside the United States and is no longer dependent (if it ever was) upon only a few indicator species (Friedrich et al. 1992; Ghetti and Ravera 1994). Much of the disagreement and criticism of using indicator species, and even communities, was largely based on the presentation and interpretation of the results and what was defined as pollution. The saprobien system and the other similar stream classification systems led to the investigation of the "pollutional status" and eventually to the definition of pollution as applied to the aquatic resources. Perhaps the most remarkable scientific contribution of the saprobien system was the direct development of numeric biotic (or saprobic) indices, which are discussed in the following section.

4.0 NUMERIC BIOLOGICAL INDICES

> One of the biologist's challenges is to present information that is understandable, meaningful, and helpful to associated disciplines, to administrators, and to the general public who are the financial supporters as well as the benefactors of a pollution abatement program. (Ingram et al. 1966)

This challenge resulted in a search for numerical expressions in a form simpler to understand than long species lists and well-thought but lengthy technical explanations of the data.

4.1 Early Indices of Pollution

The work of Wright and Tidd (1933) was considered by some to be the "original numeric index" (Myslinksi and Ginsburg 1977). Abundance of oligochaetes was used to assess the degree of pollution as follows: values of less than 1000 m^{-2} indicated negligible pollution, between 1000 to 5000 m^{-2} indicated mild pollution, and over 5000 m^{-2} indicated severe pollution. However, Richardson (1928) found that numerical abundances of each index group was not as significant as their relative abundances and overall occurrences. He reported the number of pollution tolerant Tubificid worms in the Illinois River to range from under 1000 to over 350,000 per square yard in pollutional zones, and chironomid midge larvae to range from zero to over 1000 per square yard. Richardson concluded that seasonal and habitat changes were responsible for much of the variability of species abundance at a given site, supporting the use of relative abundance as the better index measure.

Several attempts were made in the next two decades to numerically characterize the biological data in a meaningful and understandable manner. A variety of schemes were used including indices based on trophic function, structural ratios of taxa, feeding requirements, and other inventions (Washington 1984). Not satisfied with how aquatic biological field data was presented, Ruth Patrick (1950) developed a "histogram" approach based upon seven taxonomic groups. She used (I) blue-green algae, (II) oligochaetes, leeches, and pulmonate snails, (III) protozoa, (IV) diatoms, red algae, and most green algae, (V) other rotifers, clams, prosobranchia snails and tricladid worms, (VI) all insects and crustacea, and (VII) fish. Patrick's work portrayed the results in a graphical and numeric format, but also recognized the importance of community composition rather than a population-based analyses. Stream classes of healthy, semihealthy, polluted, very polluted, and atypical were assigned based upon comparing the predominance of cleaner water groups IV, VI, and VII compared with the other four groups. Cairns (1974) felt that

> The particular importance of this paper was that it showed that biological data used to assess pollution could be presented numerically and that one need not depend upon the usual unwieldy (and for nonbiologists incomprehensible) list of species to make an estimate of the degree of pollution...Thus, the method had both scientific merit and, perhaps more importantly, results could be easily communicated to people to whom species lists were meaningless.

Aquatic community assessments using the seven taxonomic groups was an ecological strength of Patrick's method but it was feared that state agencies would have difficulty with routinely collecting this wide array of data (Beck 1954). In addition, Cairns (1974) acknowledged one drawback of Patrick's method, similar to the saprobien system, in which "it required a highly skilled professional to make the species determinations necessary to properly categorize the system and its response to pollution stress." Although ecologically significant, Patrick's approach was too burdensome on most field biologists who also preferred the comfort of dealing with the one assemblage that they knew best. Thus the need for a cost-effective "rapid" biological assessment method was established. Biologists and other water resource scientists continued to struggle to derive a suitable index, or numerical expression, of the aquatic indicator organisms response to assess pollution effects. One method that many thought would solve this dilemma measured structural diversity of the community or assemblage rather than the functional or indicator role of the populations.

4.2 Diversity Indices

Species diversity, or the evenness of the distribution of individuals in a community assemblage, has been widely used since the 1960s as a measure of stream community response to pollution (Norris and Georges 1993). Cairns (1977) saw great potential for diversity indices and felt that it was "probably the best single means of assessing biological integrity in freshwater streams and rivers." Diversity indices gained favor with sanitary engineers and biologists as easy numeric indices that, as Hawkes feared, would be "entered into neat columns alongside analytical results" of chemical and physical parameters. One of the most popular diversity indices used for water resource quality assessment, H' was published by C.E. Shannon (Shannon and Weaver 1949). It is correctly termed the Shannon–Wiener index because Wiener (1948) independently published a similar measure at approximately the same time (Washington 1984). Possibly the first use of the Shannon–Wiener diversity index for assessing water quality was by Wilhm and Dorris (1966) who used diversity to describe longitudinal variation in the benthic community structure of an Oklahoma creek affected by municipal and industrial wastes. They found a severe decrease in the diversity index immediately downstream from a pollution source and an increase in the diversity index as recovery occurred. Wilhm and Dorris (1970) described the ranges of H' (calculated as d) associated with clean, moderately polluted, and substantially polluted streams.

Although the Shannon–Wiener diversity index has been used most often with benthic macroinvertebrates, it has also been used for many other assemblages. An example of this is the Index of Well-Being (Iwb) for fish. James Gammon (1976) developed the Iwb based upon both abundance and biomass measures as follows:

$$\text{Iwb} = 0.5 \ln N + 0.5 \ln B + H'_N + H'_B$$

where: N = number of individuals caught per kilometer
B = biomass of individuals caught per kilometer
H' = Shannon-Wiener diversity index calculated based on individuals per kilometer (H'_N) and biomass per kilometer (H'_B)

This combination of diversity and biomass has been highly successful for assessing fish assemblages in large rivers (Hughes and Gammon 1987; Plafkin et al. 1989; Yoder and Rankin, Chapter 9).

Despite the popularity and apparent success of the Shannon–Wiener index, it has also been severely criticized in the United States (Bilyard and Brooks-McAuliffe 1987; Fausch et al. 1990) and Europe (Metcalfe 1989; Friedrich et al. 1992; Ghetti and Ravera 1994) resulting in diminished use. The greatest criticisms of diversity indices include: (1) their inability to reflect ecological significance, (2) total reliance upon structural (abundance) measures that vary greatly depending upon the time of year sampled, the collecting gear used, and the level of taxonomic resolution, and (3) the loss of community composition information by using a single index value (Washington 1984; Metcalfe 1989; Fausch et al. 1990). Cairns et al. (1993) explained that since "the identity of the species is ignored in the calculation of diversity indices, these measures are not sensitive to compensatory changes in the community...which alter the taxonomic composition of the community but have little effect on community diversity." Hilsenhoff (1977) found little ecological significance when using diversity indices and concluded that "the diversity index does not accurately assess the water quality of streams, ranking some of the cleanest undisturbed wilderness streams with moderately enriched or polluted streams."

Despite these limitations, the popularity of the Shannon–Wiener diversity index, among others, is quite high (Norris and Georges 1993). The diversity index is currently used to characterize a variety of aquatic assemblages in different aquatic resource types throughout the world (Friedrich et al. 1992; Ghetti and Ravera 1994).

4.3 Beck's Biotic Index

Beck (1954) developed a biotic index that produced a numeric end point that could be easily interpreted by sanitary engineers and other water resource managers. Although he conceded the popularity of Patrick's method, Beck criticized Patrick's histograms because it was "hardly within economic reach of the average state regulatory agency or industry, and the information obtained is not available to the general research worker." Beck's index was originally based upon three classes of benthos — Class I (intolerant), Class II (facultative), and Class III (pollution tolerant) — but he decided not to use the tolerant organisms since they were sometimes found in cleaner waters although at a much lower abundance (Beck 1955). Beck's index, which ranged from 0 to 40, did not rely upon organism abundance but assigned numeric values (weights) of 2 and 1 for the different Class I and Class II taxa, respectively. The final index values were calculated by the formula with S representing the number of taxa within each group:

$$\text{Biotic index} = 2(S \times \text{Class I}) + (S \times \text{Class II})$$

Although this index did not achieve prominence and widespread use as an assessment tool among state biologists, it was considered a successful advance in the field of aquatic biology in the United States and was credited with popularizing the term "biotic index." This index is currently used by the Soil Conservation Service (Terrell and Perfetti 1989) as one of several water quality indicators. Perhaps the most widespread use of this index is by citizen volunteer monitoring organizations, which uses three classes of indicator organisms as originally proposed (Kopec 1989; Lathrop and Markowitz, Chapter 19).

4.4 Saprobic Index

There was another "biotic" that which was developed in central Europe at the same time Beck's biotic index was presented in the United States. Pantle and Buck's (1955) biotic index was based directly upon the saprobien system. Its simplicity and numeric relationship to the original four zones of stream pollution lead to the development of a widely used biotic index in the United States (Hilsenhoff 1982a) and could be considered the true predecessor of today's biotic indices (Friedrich et al. 1992; Ghetti and Ravera 1994). The authors directly used the saprobien system and assigned each zone a number from 1 to 4: 1 was oligosaprobic, 2 was beta-mesosaprobic, 3 was alpha-mesosaprobic, and 4 was polysaprobic. Each organism associated with the various zones based upon Liebmann's (1962) revised list of indicator organisms were assigned the respective indicator value (s) multiplied by a relative abundance weight (h) of either 1 (species only found by chance), 3 (species occurring frequently), or 5 (species occurring in abundance). These weighted values were then averaged to derive the saprobic rating, S, as follows:

$$S = \frac{\Sigma s \times h}{\Sigma h}$$

Saprobic ratings of 1 to 1.5 indicated very slight impurity, 1.5 to 2.5 was moderate impurity, 2.5 to 3.5 revealed heavy impurity, and 3.5 to 4.0 showed very heavy impurity.

Tümpling (1962) established regression lines for S with biochemical oxygen demand (BOD), oxygen deficit in percent saturation, and concentration of ammonium ion. There was a great concern that any valid index should be correlated with BOD loadings and instream dissolved oxygen. Tümpling (1969) also showed the index (S) could be used to determine saprobity with a 95% confidence interval to a level of 0.2 to 0.3 units of S. He cited Liebmann's (1962) support of this index and his own 1962 work as verification (Tümpling 1962). Guhl (1986) also found that surface waters could be defined as biologically different if the saprobic index varied by more than 0.2 units if sampled by the same investigator and 0.5 units if sampled by different investigators.

The saprobic index has been modified in many ways since it was first proposed by Pantle and Buck. However, the conceptual modifications made by Zelinka and Marvan (1961) were the most substantial and forever changed the use of the saprobic index. They addressed many of the criticisms of the original index, as well as the general use of indicator organisms by adding a saprobic valency and indicator weight to the original index. The saprobic valency expressed the relative frequency of the species in different degrees of saprobity on a scale that totaled 10. By establishing the saprobic valency, Zelinka and Marvan satisfied a major criticism of the saprobic index — it's dependance upon arbitrarily assigning a single saprobic zone to a species which is likely to be in more than one zone. They also felt that some species were more useful as indicator organisms and assigned an indicative weight from 1 (poor indicator) to 5 (very good indicator). The best indicators were those species that had been assigned a saprobic valency of 8, 9, or 10 within any given zone (Slàdečék 1991). This showed that the best indicators were representative of a single saprobic zone. The lowest indicator weights were given to species found throughout many or most of the saprobic zone. These conceptual modifications of the saprobic index have been quite popular and ensured widespread use not only throughout central Europe, but also other parts of the world. Bick (1963), Fjerdingstad (1964), Friedrich (1990), Slàdeček (1965, 1985, 1988, 1991), Friedrich et al. (1992), and Ghetti and Ravera (1994) provide a great deal of insight into the specific changes and uses of the Saprobic Index.

4.5 Selected Macroinvertebrate Indices

Woodiwiss (1964) presented the biological system of stream classification that was used by the Trent River Board in England. The Trent biotic index varied from 0 to 10, with 10 representing clean water, based upon the relative abundance of representative benthos groups. This index greatly influenced the development of the Chandler biotic index, Belgian biotic index, Extended biotic index, and Indice Biologique (Metcalfe 1989).

Metcalfe (1989) also categorized European indices into saprobic, diversity, and biotic but did not view modern biotic indices as an extension or modification of the original saprobic indices, contrary to Friedrich et al. (1992) and Ghetti and Ravera (1994). This was because saprobic indices had an early focus on plankton and periphyton whereas biotic indices were based on benthic macroinvertebrates. Friedrich et al. (1992) differentiated the indices by labeling biotic indices as those that utilize only some taxa from an assemblage and the saprobic indices as using as many species of the community as possible.

In the United States, Hilsenhoff (1977, 1982a) also recognized the direct relationship among the early saprobic indices and biotic indices. He credits Pantle and Buck (1955) and Chutter's (1972) index as predecessors of the Hilsenhoff biotic index. Chutter (1972) developed a biotic index for South African streams and assigned specific tolerance values for various taxa ranging from 0 to 10. His index accounted for both the number of individuals and the number of taxa, but contained only limited quality (i.e., tolerance) values.

Hilsenhoff's (1977, 1982a) biotic index (see below), originally scaled from 0 (clean) to 5 (polluted) was based on a wide array of aquatic insect taxa from Wisconsin identified to genus or species.

$$\text{Biotic index} = \frac{\Sigma n_i \times a_i}{N}$$

where n_i = number of individuals of each taxon
a_i = tolerance value assigned to that taxon
N = total number of individuals in the sample

He compared the biotic index with Shannon's diversity index based upon several physical and chemical parameters using rank correlation analysis. Hilsenhoff (1977) found that the biotic index correlated much better than the diversity index in distinguishing among pollution gradient in streams. Hilsenhoff revised the index in 1982 and in 1987 to reflect new index values, expanded the biotic index scale to 0 to 10, and included several new taxa. He then developed a popular family-level biotic index that has also been widely used for screening water resource quality (Hilsenhoff 1988). A Hilsenhoff biotic index, with tolerance values modified for specific geographic regions, is used for water quality assessment in many states (Lenat 1993; Bode and Novak, Chapter 8; Southerland and Stribling, Chapter 7) and has become incorporated into the new generation of numeric multimetric indices (Plafkin et al. 1989; Barbour et al., Chapter 6; Resh, Chapter 12). Resh and Jackson (1993) provide a comprehensive review of rapid bioassessment techniques using benthic macroinvertebrates.

4.6 Multiple Metric Indices

During this time, debate continued regarding the use of numerical biological indices based upon indicator organisms. Brinkhurst (1969) stated that "the value of biological methods of pollution detection is now widely accepted, but there is still considerable debate about the means of providing inexperienced biologists with simple standard procedures and of reporting biological data to non-biologists." The advantage of both diversity and biotic indices is that they reduced complex interactions and pollution responses of an aquatic community into a single number for water quality management purposes. However, neither of these indices were successful in describing the overall "health" or condition of the aquatic ecosystem under a variety of conditions. It was clear that a better tool was need to more consistently and accurately characterize the aquatic communities.

Karr (1981) published the Index of Biotic Integrity (IBI) to provide a more accurate and consistent approach towards measuring the societal goal of "biological integrity" (see Section 5.2 in this chapter). The IBI includes discrete measurements of 12 fish assemblage attributes, or metrics, based on species composition, trophic composition, abundance, and condition. Each metric was assigned a score (5, 3, or 1) based upon specific ecological expectations. The 12 metric scores were summed to provide a cumulative site assessment. The scores result in "integrity classes" for streams of excellent, good, fair, poor, very poor, or no fish (Karr et al. 1986). The IBI is called a composite or multiple metric index because it combines several community attributes into a single index value without losing the information from the original measurements. A number of natural resource and regulatory agencies have demonstrated this to be a very successful tool for water resource quality evaluations (Simon and Lyons, Chapter 16; USEPA 1991a; Abe et al. 1992). [Please refer to Simon and Lyons (Chapter 16) and Yoder and Rankin (Chapter 9) for more information on the application and regional and local modification of the IBI metrics.]

It did not take long before multiple metric indices were developed for benthic macroinvertebrates and periphyton. The Ohio Environmental Protection Agency developed an Invertebrate Community Index (ICI) in 1986 (DeShon, Chapter 15) that is based on ten structural and functional metrics that quantify subjective judgements that had been used for a number of years. Shackelford (1988) developed a multiple metric benthic index for Arkansas that combined seven measures of community diversity, indicator organism, and functional groups. USEPA then published a set of composite indices called Rapid Bioassessment Protocols (RBPs) for benthic macroinvertebrate and fish communities (Plafkin et al. 1989). The RBP benthic community metrics are based on very general structural and trophic relationships that could be applied nationally. The primary fish assessment methods were Karr's IBI and Gammon's Iwb. Hayslip (1993) recently compared the benthic metrics used by states in the Pacific Northwest and found a great deal of metric modification of the original RBPs.

Periphyton assemblages were also described using multiple metrics. A periphyton biotic index (Kentucky DEP 1992) was developed for use in the State of Kentucky to complement fish and macroinvertebrate water resource assessments. The metrics used included taxa richness, relative abundance of sensitive and tolerant taxa, percent community similarity compared with reference sites, and biomass. Bahls (1993) developed a periphyton index for Montana streams using three metrics for soft-bodied taxa (dominant phylum, indicator taxa, and number of genera) and four metrics for diatoms (Shannon–Wiener diversity index, pollution index, siltation index, and a similarity index compared with a reference condition). Rosen (Chapter 14) further discusses the periphyton indices and metrics used for developing biocriteria.

Indices of biotic integrity have not been without criticism. Suter (1993) outlined the following exhaustive list of potential faults of what he called "indexes of heterogenous variables" such as the IBI: (1) ambiguity, (2) eclipsing, (3) arbitrary combining functions, (4) arbitrary variances, (5) unreality, (6) post-hoc justification, (7) unitary response scale, (8) no diagnostic results, (9) disconnected from testing and modeling, (10) nonsense results, and (11) improper analogy to other indices. Suter (1993) explained that indices like the IBI "are justified on the basis of field studies rather than any theory of ecosystem health or any societal or ecological value of the index or its components." I am not going to present a response to each of these items, but allow these perceived issues to be addressed by the many qualified chapter authors in this book. However, I do agree that more research and testing is needed before the concepts of these indices can be expanded to develop truly ecosystem "health" indices. Karr (1993a), Simon and Lyons (Chapter 16), and Yoder (Chapter 21) present detailed responses to Suter's criticisms.

5.0 FRAMEWORK FOR CRITERIA DEVELOPMENT

There have been several key areas in which the scientific thought and application of biological tools significantly advanced water resource assessment and criteria development. Those already mentioned include the use of indicator organisms originated in the saprobien system, numeric biological indices, and the aggregation of several numeric biological attributes into multiple metric indices for measuring biological integrity. The remaining substantial developments are a combination of technical achievement and conceptual implementation for describing societal and ecological goals from which progress towards meeting those goals could be measured. They include defining pollution through beneficial use assessments for aquatic life support based on measures of biological integrity and using multiple reference sites to define attainable (reference) conditions within a regional framework (i.e., ecoregions).

5.1 Debating Ecological and Societal Goals for Water Resources

The legal authority for water rights and ensuring the water resource is fit to serve private and public uses has long been recognized (Warren 1971). However, the uses or combination of uses, and their priorities were quite different depending upon the needs of the user. In the 1800s and early 1900s, the focus on water uses were primarily as conveyances of municipal and industrial wastes and for drinking water, a very distasteful combination! When the water resource was not required for a potable water supply, the common "standard" was one that avoided a public nuisance. An example of the concern for economic use of the water resource without regard for how the aquatic life or downstream communities were affected was found in the city of Chicago during the early 1900s. Consider the following excerpts from a report delivered to the Board of Trustees of the Sanitary District of Chicago by its Commissioner (Wisner 1911):

> The question has always been considered from the standpoint of nuisance, and not as to whether or not the water was so polluted as to destroy fish life.... Our investigations lead to the conclusion that a nuisance may not occur, even though all the fish be dead through the lack of sufficient dissolved oxygen necessary to fish life.... From an inspection of the available data on the condition in the Illinois River, and in the Des Plaines River, prior to the opening of the Drainage Canal in 1900, it is evident that a marked improvement took place. The foul conditions had been tolerated for years. Fish life has decreased in the main river. Since the opening of the canal the fish catch is said to have improved, although no definite data are available. Owing to the great extent of the fish industry in the Illinois River it is essential that the condition of the river, insofar as the Sanitary District is concerned be kept as good as possible.... It is necessary that immediate steps be taken to ascertain the conditions along the river...and that the continued examination be made year after year by the Sanitary District in order to have the data in hand to refute possible law suits for damages to fishing, and the possible reopening of the St. Louis case. This is not only a matter of sanitation, but a question of self-defense, protecting the root and purpose of the Sanitary District.

Additional information about the early history of Chicago's water quality experiences can be found in Cain (1978) and Davis (1990).

Potential beneficial uses were clarified in the 1948 Federal Water Pollution Control Act (Public Law 845) which stated, "[i]n the development of such comprehensive program due regard shall be given to the improvements which are necessary to conserve such waters for public water supply, propagation of fish and aquatic life, recreational purposes, and agriculture, industrial, and other legitimate uses" (Mackenthun and Ingram 1967). Defining an independent use solely for the propagation of fish and aquatic life was a very important advance. However, it was difficult to determine whether this use was being attained, or was even attainable. The problem with using biological assessments and indices for water resource quality assessment was not just with the numeric interpretation of the data, but also with understanding what the measurements meant with respect to the desired condition of the resource. This problem was reflected by Doudoroff and Warren's (1957) comment that "[a]lthough most authors evidently have recognized the economic significance of pollution, it appears that when devising their biological indices and measures of water pollution and its severity some biologists have completely disregarded all economic considerations."

Many water resource quality specialists disagreed as to whether to emphasize economically important populations such as gamefish, coastal invertebrates, and freshwater mussels or all aquatic life equally. For example, the Ohio River Valley Sanitation Commission (ORSANCO) Compact of 1948 called for waters that are "capable of maintaining fish and other aquatic life" (Cleary 1955). However, in 1954 the Aquatic Life Advisory Committee to ORSANCO concluded that their mission was concerned with only the "production of fish crops" measured by bioassays (Cleary 1955). Although there was a great deal of information available on benthic macroinvertebrates, phytoplankton, attached algae, and fish assemblages, ORSANCO's focus turned only to using toxicity test results for criteria development.

Confusion and disagreement with defining societal goals for clean water and the control of pollution was reduced (or at least redirected) with the 1972 Amendments to the Federal Water Pollution Control Act, commonly referred to as the Clean Water Act (USGPO 1989). The general objective of the Act is to "restore and maintain the chemical, physical, and biological integrity of the Nation's waters" [Section 101(a)]. The second national goal of the Act [Section 101(a)(2)] was "wherever attainable, an interim goal of water quality which provides for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water." (Section 101(a)(2) has commonly become known as the "fishable and swimmable goal".) USEPA's written opposition to the controversial objective of the act firmly established the relationship and relevance of water quality to support the beneficial uses:

> The pursuit of natural integrity of water for its own sake without regard to the various beneficial uses of water is unnecessary, uneconomical, and undesirable from a social, economic, or environmental point of view. We believe the purpose of water pollution control is the achievement and protection of water quality for beneficial uses. (USGPO 1972)

5.2 Defining Biological Integrity

Legislation to protect aquatic life first appeared in 1876 and has continued to develop in its scope and intent (Table 2). Most of the early legislation was geared toward the protection of waters for human use (beneficial uses). The 1972 Clean Water Act represented a change in that its objective and new interim goal went far beyond the application of mere beneficial uses. It was viewed as having an "ecological" beneficial use for the sake of the environment, independent of any readily available economic benefits. This language was not trivial and caused a great deal of concern regarding how to define "integrity" (especially biological integrity) and the measurements to be applied. The 1972 House Committee on Public Works (USGPO 1972a) defined integrity as a "concept that refers to a condition in which the natural structure and function of ecosystems is maintained." Continuing, they stated "[o]n that basis we could describe that ecosystem whose structure and function is 'natural' as one whose systems are capable of preserving themselves at levels believed to have existed before irreversible perturbations caused by man's activities. Such systems can be identified with substantial confidence by scientists." It is rewarding that Congress had such a high a opinion of our discipline. The 1972 Senate Public Works Committee (USGPO 1972b) stated that "The 'natural...integrity' of the waters may be determined partially by consultation of historical records or comparable habitats; partially from ecological studies of the area or comparable habitats; partially from modelling studies which make estimations of the balanced natural ecosystems on the information available".

The National Commission on Water Quality (USGPO 1976), which was appointed to make a full and complete investigation and study of all aspects of the Clean Water Act requirements (Section 315), had difficulty in setting the course of its study because of the ambiguity of the term "biological integrity" (USGPO 1975). Their difficulty was obvious when they concluded that "[t]he most quantifiable indicators of biological health in an aquatic system are the physical and chemical parameters assessed at each of the study sites. Individually and together, they provide a broad picture of existing water quality and projected progress toward a quality that will support the purposes listed in the interim goal" (USGPO 1976). Based on this more comfortable and traditional position, they declared that the objective of the act really meant to focus on a combination of all three integrity measures as a single concept of ecosystem integrity (USGPO 1976).

Still not satisfied with the answers provided by Congress or the National Commission on Water Quality, EPA hosted a national forum on the Integrity of Water in 1975. Both qualitative and quantitative concepts of chemical, physical, and biological integrity were reviewed. Two definitions of biological integrity were informally proposed at the forum. The first was by Cairns (1977) who felt that "biological integrity may be defined as the maintenance of community structure and function characteristic of a particular locale or deemed satisfactory to society." The second definition was proposed by Frey (1977) who defined the integrity of water as "the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a composition and diversity comparable to that of the natural

Table 2. Important Legislation and Agreements Facilitating Biological Criteria

Legislation	Year	Key Elements for Biocriteria
River Pollution Prevention Act (England)	1876	First legislation intended to provide protection to fisheries as well as prevention of nuisance
Public Health Service Act	1912	First national investigations of pollution and aquatic life in major U.S. river systems
Federal Water Pollution Control Act (PL 80–845)	1948	Established federal authority for interstate water pollution control. Recognized propagation of fish and aquatic life as a legitimate beneficial use of waters
Ohio River Valley Sanitation Commission Compact	1948	Established mechanism for developing water quality criteria to protect aquatic life via toxicity testing
FWPCA Amendments (PL 84–660)	1956	Fish and aquatic life protection formalized as a beneficial use. Began national water quality monitoring network requiring systematic collection of aquatic life
FWPCA Amendments (PL 89–234)	1965	Federal authority to review state water quality standards
National Environmental Policy Act (PL 91–190)	1969	Submission of Environmental Assessment and Impact Statements on proposed federal actions
Clean Water Act (PL 92–500)	1972	Objective of the Act focused on maintaining and restoring biological integrity of surface waters. Started large movement towards defining and measuring biological integrity
Endangered Species Act (PL 93–205)	1973	Provides protection for special status species
Water Quality Act (PL 100–4)	1987	Shift from technology-based to water quality-based approach
Great Lakes Water Quality Agreement	1987	Adopted a biological integrity objective moving towards ecological integrity and measuring indicators of ecosystem health
Water Pollution Prevention and Control Act (Draft S 1114)	1993	Supports "biological discharge criteria" based upon establishing the biological conditions of the waterbody
Nonpoint Source Water Pollution Act (Draft HR 2543)	1993	Expands ecosystem integrity protection approach and supports biological criteria
Biological Survey Act	1993	Requires a national biological survey to assess the status and trends of the biological resources in the U.S. and establishes an new agency to carry out the survey

habitats of the region." Apparently, neither of these definitions were widely accepted at that time and it was evident that defining biological integrity, and hence the methods to measure it, was a complex task that required a more focused effort.

USEPA's Water Office asked the Corvallis Environmental Research Laboratory to review the definition of biological integrity and to suggest ways it might be monitored (Hurley 1981). As a result of that request, the USEPA Corvallis Laboratory assembled a team of experts from within USEPA, academia, and the Fish and Wildlife Service to tackle the problem (Hughes et al. 1982). This team provided the breakthrough to assemble a functional definition and framework for describing biological integrity. At a national workshop in 1981, they presented: (1) a definition of biological integrity establishing base (reference) conditions within faunal regions (ecoregions), (2) methods comparing baseline conditions with impacted conditions to determine relative well-being, (3) the use of multiple sites to establish reference condition, and (4) the foundation of Karr's Index of Biotic Integrity (Hughes et al. 1982). They concluded that

> a definition of biological integrity has been adopted that established base biological conditions as those found in the least-disturbed typical reaches of large, relatively homogeneous faunal regions. Once these base biological conditions have been established, data gathered at other locations within faunal regions will be compared with the base to determine the relative well-being of each non-base location. They suggested the use of fish assemblages as the indicator of biointegrity. (Hughes et al. 1982)

Karr and Dudley (1981) further defined biological integrity with the ecosystem perspective of Frey (1977) by adding functional organization to the desirable characteristics of the aquatic community. This differed from the narrower "fishable and swimmable" Clean Water Act goal and also met the intent of the House Committee (i.e., natural structure and function of an ecosystem) as well as the Senate Committee (i.e., comparable habitat). Karr and Dudley's (1981) definition has become widely accepted within the regulatory and scientific community (Schneider 1992). Karr (1981) recommended that biological integrity be used to "assess the degree to which waters provide for beneficial uses," especially aquatic life support. It did not take long before these concepts were tested for state programs (Southerland and Stribling, Chapter 7). Currently, the term biological integrity has been used synonymously with attaining the beneficial use for aquatic life protection (Yoder and Rankin, Chapter 9). [Please see Karr (1991; Chapter 2) and Adler (Chapter 22) for additional perspectives of the legislative and technical activities relating to ultimate focus on biological integrity.]

5.3 Reference Condition and Regionalization

When Hughes et al. (1982) recommended a definition of biological integrity and ways to measure it, they had in mind developing sets of least impacted reference (or attainable) conditions within faunal regions (i.e., ecoregions) to compare with each test location. The rationale for using ecological regions was to establish reference conditions based upon patterns in community attributes that had previously been found to vary naturally among geographic regions (Hughes and Larsen 1988). Without accounting for natural geographic variability it would be difficult to establish numerical indices that were comparable from one part of a State to another, much less nationally. Therefore, using ecoregional reference conditions allow an unbiased estimate of the surface water's attainable (least impacted) conditions (Hughes et al. 1986). These concepts were the turning point in finally developing defensible biological (and some chemical) criteria for state water quality standards and other programs. Hughes (Chapter 4) and Omernik (Chapter 5) discuss reference condition and ecoregions in detail.

The first complete application of the framework for biocriteria development occurred as a result of the Ohio Stream Regionalization Project (SRP). This cooperative effort among Ohio EPA, USEPA's Environmental Research Laboratory in Corvallis, Oregon, and USEPA's Region 5 office in Chicago was conducted in 1983 and 1984 (Whittier et al. 1987). The SRP identified and delineated five ecoregions in Ohio and then focused on selecting least-impacted reference watersheds and sites to determine the best attainable condition in those waters. Field sampling was conducted for over a year and included physical habitat, fish and macroinvertebrate assemblages, and chemical water quality in 109 streams. The fish assemblage was measured by several means including the Index of Biotic Integrity and the Index of Well-Being. The results were displayed in box plots and the attainable conditions were based upon the 50th percentiles of each of the attributes. Ohio EPA later refined the attainable conditions for aquatic life (warmwater biocriteria) based on a 25th percentile of the ecoregional reference site conditions of each measurable attribute related to drainage area. The success of this demonstration project led the State of Ohio to adopt numeric biological criteria in 1990 based on results of over 236 reference sites throughout the state (Yoder and Rankin, Chapter 9).

6.0 FUTURE PROSPECTS

Many natural resource, land management and regulatory agencies are beginning to implement biological assessments and even criteria development as essential tools to protect water resource quality and biodiversity (ITFM 1992, 1994; CEQ 1993; NRC 1993; USEPA 1993b, c,d,e,f) and to reduce the uncertainty in applying the traditional chemical criteria to protect those resources (USEPA 1990a, 1991c). Although these efforts have been sustained by state agencies for the past decade, federal agencies are also beginning to actively participate in biological assessments which will likely lead to criteria development (NRC 1993, Table 3). USEPA has issued guidance on developing biological criteria programs (USEPA 1990, 1993c), developing narrative biocriteria (USEPA 1992c), and is finalizing a technical guidance document for streams (Gibson 1994). EPA has hosted several biocriteria workshops and national meetings in cooperation with state agencies (e.g., USEPA 1987b; Davis 1990a; Hayslip 1993; see Southerland and Stribling, Chapter 7). However, there is no guarantee that these efforts will be successful. We must continually educate ourselves and our colleagues regarding the benefits of biological assessments and criteria in environmental restoration and protection. We must improve our existing methods and applications and maintain the necessary research on new and promising techniques. We also must not forget the dedication and philosophy of those scientists who brought us to the present.

Table 3. U.S. Federal Agencies Involved in National Biological Assessments

Agency	Program
Department of Interior	
Fish and Wildlife Service	National Contaminant Biomonitoring Program
	Biomonitoring of Environmental Status and Trends
	Waterfowl Breeding Population and Habitat Survey
	National Survey of Fish, Hunting and Wildlife
Geological Survey	National Water Quality Assessment
National Park Service	Watershed Protection Program: Park-Based Water Quality Data Management Program
	National Wild and Scenic Rivers System
Bureau of Land Management	Federal Land Policy and Management Act Assessments
	BLM Initiatives
Bureau of Reclamation	National Irrigation Water Quality Program
National Biological Survey	National Biological Survey
Department of Commerce	
National Oceanic and Atmospheric Administration	National Status and Trends Program
	Fisheries Statistics Program
	Living Marine Resources Program
	Classified Shellfishing Waters
Environmental Protection Agency	Great Lakes Fish Monitoring Program
	National Water Quality Monitoring Program
	Environmental Monitoring and Assessment Program
Tennessee Valley Authority	Water Resources and Ecological Monitoring Program
Department of Agriculture	
Forest Servce	Resource Planning Act Assessments
	Forest Service Water Quality Program
	Watershed improvement program
	Nonpoint source pollution management program
Soil Conservation Service	President's Water Quality Initiative

Source: USEPA 1993d.

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