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Fishes as indicators of environmental and ecological changes within estuaries: a review of progress and some suggestions for the future

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Estuarine habitats, and the fish assemblages associated with them, are potentially impacted upon by many anthropogenic influences which can have a direct influence on the food resources, distribution, diversity, breeding, abundance, growth, survival and behaviour of both resident and migrant fish species. The direct and indirect coupling between ichthyofaunal communities and human impacts on estuaries reinforces the choice of this taxonomic group as a biological indicator that can assist in the formulation of environmental and ecological quality objectives, and in the setting of environmental and ecological quality standards for these systems. This review examines the rationale and value of selecting fishes as bio-indicators of human induced changes within estuaries, using examples from both the northern and southern hemispheres. The monitoring of estuarine ' health ' using fish studies at the individual and community level is discussed, with an emphasis on the potential use of estuarine fishes and their monitoring and surveillance in national and international management programmes. In illustrating the above concept, examples are presented of the way in which fishes are threatened by anthropogenic impacts and of the way in which teleosts can contribute to a monitoring of estuarine ecosystem health.

Key words: fish conservation; biological indicators; estuaries; environmental health; anthropogenic impacts.

INTRODUCTION

There is a growing interest in the use of biological communities to assess the status of water resources (Deegan *et al*., 1997; Bain *et al*., 2000; Simon, 2000). While many investigations aimed at detecting environmental and ecological changes within estuaries have focused primarily on water quality (e.g. faecal coliforms, nitrate levels or BOD) and the associated biota (e.g. aquatic plants, invertebrates and birds), there are relatively few studies based solely on fishes (Costa *et al*., 1992; Dennison *et al*., 1993). In addition, monitoring programmes focusing on ichthyofauna (Paller *et al*., 1996) seldom address changes over more than one level of biological organization, for example cellular, individual, population and community.

What is the current status of knowledge concerning the use of fishes as indicators of ecological change and estuarine health, and how is this information being used? This paper reviews a number of case studies, and examines how scientists can incorporate ichthyological data into decision support systems for the wise management of estuaries. It also highlights the difficulties in the

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detection of environmental and ecological change due to anthropogenic factors, due primarily to a high background variability that is naturally found within estuaries. For example, estuaries are amongst the most fluctuating aquatic environments on earth, with the boundaries of natural variability, even for individual systems, seldom defined or recorded.

WHY USE FISHES AS ENVIRONMENTAL INDICATORS?

Environmental indicators have been defined as ' physical, chemical, biological or socio-economic measures that best represent the key elements of a complex ecosystem or environmental issue. An indicator is embedded in a well-developed interpretative framework and has meaning beyond the measure it represents ' (Ward *et al*., 1998). They can be qualitative or quantitative although the latter are perhaps more useful if they are to lead to management actions. Using indicators, it is possible to evaluate the fundamental condition of the environment without having to capture the full complexity of the system. Indicators are based on the best scientific understanding currently available so that changes in these simple measures can be related to more complex environmental trends. When time-series data for an indicator show a trend, then there is a need to provide some interpretation of the trend and its implications. Environmental indicators, however, not only help track changes in an ecosystem, they also simplify the state of the environment reporting in two ways. Firstly, indicators have a well-understood meaning and can be measured regularly, thus yielding valuable information about important aspects of the environment. Secondly, environmental indicators can be an aid to communication in that they allow information about the environment to be communicated effectively, especially as users of this information become more familiar with the agreed indicators (Australian and New Zealand Environment and Conservation Council, 2000).

Recent developments (Elliott, 2002) have used the so-called DPSIR (drivers, pressures, status, impact, response) approach to illustrating environmental change and the human responses to that change. This approach concentrates on a knowledge of drivers, i.e. the underlying cause such as climate change, industrialization, water abstraction and building of infrastructure. These developments will create pressures, such as loss of habitat, input of pollutants and interference with water patterns. In turn, the status of any component such as water chemistry, substratum type, community structure, biological populations and individual characteristics such as growth and condition, can be assessed. Following this, the magnitude of any impact, either spatially (as an extent of the impact) or duration (as the temporal basis of the impact) can be determined. Finally, the response component needs to be identified, i.e. man's actions to minimizing, mitigating or controlling the impacts. Each of these elements, but especially the pressures, status and impacts require indicators that define the cause and effects of changes to the system.

It is possible to generate a generic framework for the use of fishes as estuarine environmental indicators with regard to definition, classification, monitoring, assessment, reporting and management (Fig. 1), and within this to denote the parts of the framework dependent on the development and use of indicators for environmental health and indicators of response to changes in health. Whereas

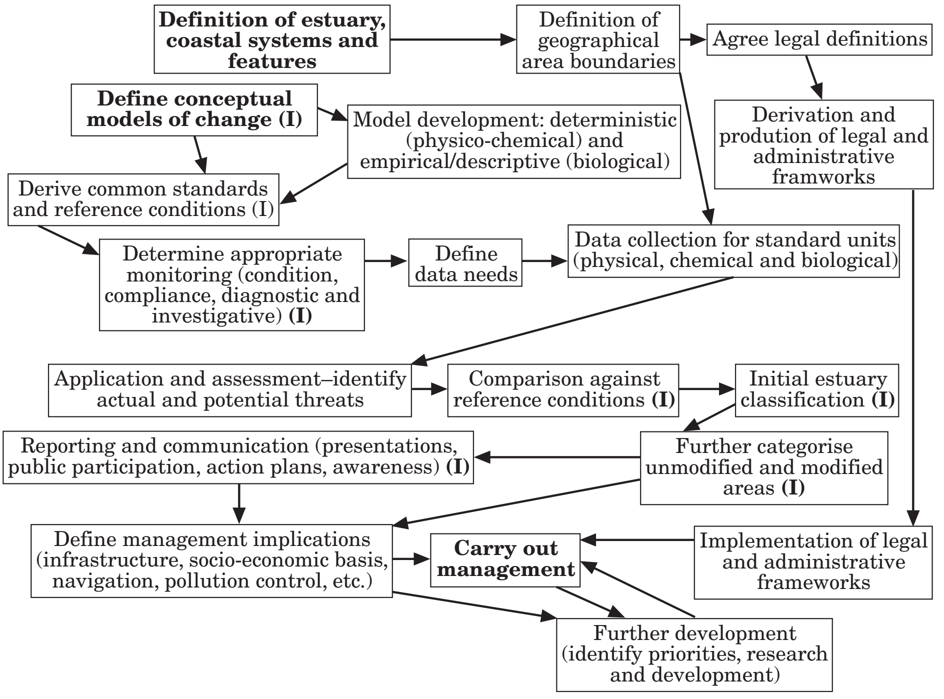

FIG. 1. Generic framework for fish indicator use (I, need for indicators) (modified from Elliott & McLusky, in press).

some of those indicators will then be used to explain to the public and policy-makers the nature of changes as a result of human activities, others may be used as diagnostic tools to quantify the consequences of any change. For example, a change in the fish community may be the result of anthropogenic activity such as a polluting discharge, but also such a change may be quantified in order to determine whether man is having a significant effect on the system or not.

Many groups of organisms have been proposed and used as indicators of environmental and ecological change (Karr *et al.*, 1986). Although no single group is favoured by all biologists, it appears that fishes, macroinvertebrates, birds and plants have received the most attention (Schaeffer *et al.*, 1985; Morrison, 1986; Fausch *et al.*, 1990; Dennison *et al.*, 1993). Fishes have been successfully used as indicators of environmental quality changes in a wide variety of aquatic habitats (Whitfield, 1996; Soto-Galera *et al.*, 1998) and have numerous advantages as indicator organisms for environmental monitoring programmes, including: (1) they are typically present in all aquatic systems, with the exception of highly polluted waters; (2) there is extensive life-history and environmental response information available for most species; (3) in comparison to many invertebrates, fishes are relatively easy to identify and most samples can be processed in the field, with the fishes being returned to the water (non-destructive sampling); (4) fish communities usually include a range of species that represent a variety of trophic levels and include foods of both aquatic and terrestrial origin; (5) fishes are comparatively long-lived and therefore provide a long-term record of environmental stress; (6) they contain many life forms and functional guilds and thus are likely to cover all components of aquatic ecosystems affected by anthropogenic disturbance; (7) they are both sedentary and mobile and thus will reflect stressors within one area as well as providing groups to give a broader assessment of effects; (8) acute toxicity and stress effects can be evaluated in the laboratory using selected species, some of which may be missing from the study system; (9) they have a high public awareness value such that the general public are more likely to relate to information about the condition of the fish community than data on invertebrates or aquatic plants; (10) societal costs of environmental degradation, including cost-benefit analyses, are more readily evaluated because of the economic, aesthetic and conservation values attached to fishes. The use of fishes as indicators of biological integrity, however, does have difficulties and problems, including (a) the selective nature of sampling gear for certain habitats and sizes and species of fishes; (b) the mobility of fishes on seasonal and diel time scales can lead to sampling bias; (c) fishes may be relatively tolerant to substances chemically harmful to other life forms; (d) fishes can swim away from an anthropogenic disturbance, thus avoiding localized exposure to pollutants or adverse environmental conditions; (e) estuarine environments that have been physically altered by humans may still contain diverse fish assemblages.

Many of the disadvantages described above are out-weighed by the widespread advantages. In addition, it should be emphasized that a number of the negative aspects would also apply to other taxonomic groups (e.g. invertebrates) that may be used in biological monitoring of the aquatic environment.

ESTUARINE FORCING VARIABLES AND FISH RESPONSE

The major physical drivers in terms of the biological or ichthyological functioning of estuaries can be found under geographical and hydrographical categories (Fig. 2). As a general framework, fish community structure can be considered as being created by a set of environmental variables. These variables create the conditions available to the fishes but, depending on their environmental and physiological tolerances, this basic community becomes influenced by other biological variables such as predator-prey interactions and inter- and intra-specific competition (Elliott & Hemingway 2002). It should be noted that, although the geographical variables tend to have a direct impact on the estuarine hydrography, this latter component has a ‘ feedback ’ influence on the physiography through sedimentary and erosive processes.

Both the physical and biological variables contribute to niche production (or elimination) within an estuary but it is primarily the environmental variables that are driving the response of the biota, including the ichthyofauna (Green, 1968; Blaber, 1997). Therefore the measurement of any ecological response by the fish community, or individual species within that community, must take cognisance of the key role played by the physico-chemical environment in influencing the structure and functioning of that estuary.

Anthropogenic impacts can target both the biotic (e.g. fish abundance and biomass) and abiotic (e.g. river flow and turbidity) components of the ecosystem,

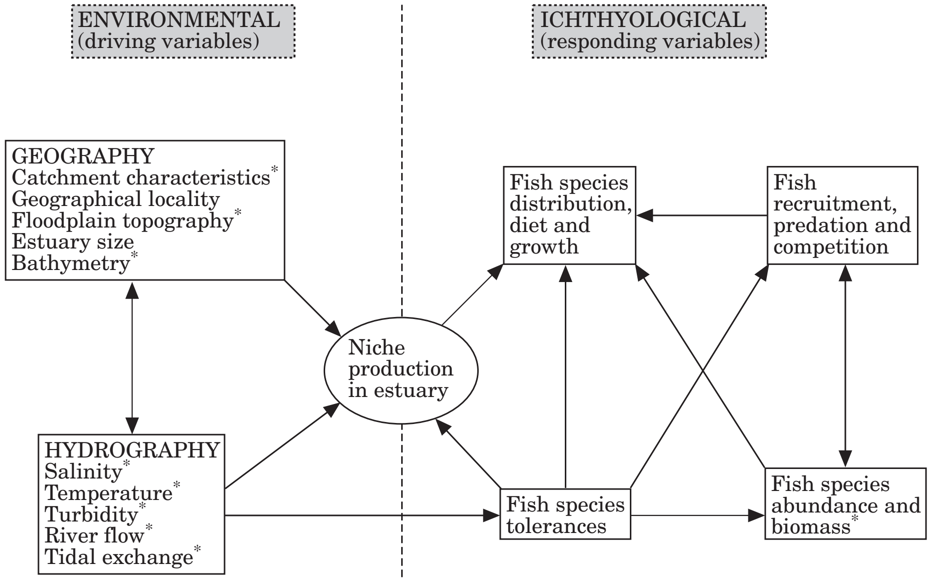

FIG. 2. Interactions between selected environmental and ichthyological variables in estuaries (* variables often influenced by anthropogenic activities).

thus influencing both the driving and responding variables within an estuary (Fig. 2). The listed environmental variables differ in their significance to fish communities inhabiting estuaries of a particular region, country or continent, e.g. estuarine hypersalinity is an important factor in the arid regions of southern Africa and western Australia but not in the wet regions of northern Europe or eastern North America. Conversely, other variables would be of equal significance in all systems, e.g. global warming and its role in driving estuarine fish community changes across all continents.

The key ' responding ' variable that is directly and frequently influenced by humans is ' fish species abundance and biomass ' (Fig. 2) and the prime effect of fishing in estuaries can be seen to have a large number of ecosystem effects (Blaber *et al.*, 2000; Elliott & Hemingway, 2002). The over-riding influences on estuarine fishes, however, occurs not just within an estuary but also both upstream (in the case of diadromous species) and at sea (in the case of marine species whose juveniles use the estuary). There are numerous examples of the effects of overfishing on estuarine fish stocks, including declines in the populations of certain over-exploited marine species being reflected in declining estuarine abundance. For example, Elliott *et al.* (1990) have shown that declining juvenile cod *Gadus morhua* L. populations in the Forth Estuary during the early 1980s reflected more widespread changes (due to overfishing) in the stocks of this species. A similar situation was recorded in the Swartkops Estuary where the white steenbras *Lithognathus lithognathus* (Cuvier) formed an important component of anglers' catches in the early 1900s but had virtually disappeared from the system by the end of the century, primarily due to overfishing in both the marine and estuarine environments (Baird *et al.*, 1996).

TABLE I. Examples of health and stress measurements of relevance to estuaries and their associated ichthyofauna

Level	Healthy fish fauna	Stressed fish fauna
Cellular	Stable lysosomes, genetic integrity.	Presence of detoxifying mechanisms, genetic damage.
Individual	No morphological anomalies, good body condition, low parasitism, natural behaviour.	Lesions, fin rot, development of tumours, poor body condition, abnormal behaviour.
Population	Larval recruitment suitable for maintenance, age and class structure as expected, distribution as expected.	Poor larval recruitment, lower numbers of juveniles, adult stages absent or poorly represented, altered distribution.
Community	Diversity as expected, normal combination and interaction of guilds, expected seasonal cycles.	Reduced prey availability, diversity lower than expected, loss of rare or sensitive species.
Ecosystem	No physico-chemical impairment, carrying capacity maintained, predator-prey interactions as expected.	Fewer niches, reduced habitat integrity, changes to the food web, reduction in higher predators.

FISH HEALTH AND STRESS

The term ' fish health ' has been used extensively in the literature (Albert & Washuta, 1992; Leamon *et al*., 2000) but little attention has been given to the implications of different interpretations of ' health ' at the various levels of biological organization. For the purposes of this paper, the following definitions have been used: (1) cellular health, this describes the structural integrity of cellular organelles and the maintenance, at a biochemical level, of cellular processes; (2) individual health, this describes structural and morphological health and functioning in terms of the physiology of the entire organism; (3) population health, this describes the sustainability and maintenance of the population of a particular species; (4) community health, this describes an appropriate assemblage of organisms and the relationships between species in that assemblage.

The methods for the detection of fish health at each of the above levels are varied and often not clearly defined, primarily due to the inherent variability of fish response within each component. The change being measured as the result of human activities (i.e. a ' signal '), for which an indicator is required, should be measured against a background of variability (i.e. ' noise '). Because of this, the ability to detect measurable change at the different levels is sometimes difficult, especially at the higher levels of biological organization (Table I). In addition, the relevance of any change to the health at one level to that of another is often difficult to interpret. Elliott & Hemingway (2002) examined the responses by fishes to change at all levels of organization and concluded that there are few examples in which a change at lower levels (e.g. cellular, individual) are translated to higher level responses (e.g. community or fishery level). Instead, the inherent variability is suggested as having the capacity to absorb change, i.e. ' environmental homeostasis '. This does not imply that the lack of information or response from one or more levels justifies any lack of action on the part of management; indeed the precautionary principle implies an assumption that a change at a lower level will eventually be manifest at a higher level unless it is checked. Thus, an appropriate management action may be to initiate intensive studies to elucidate the impacts of health changes on higher or lower levels.

Another issue related to fish health is ' stress ', i.e. the cumulative and quantifiable effect of a factor or combination of factors operating on an individual, population, community or ecosystem that renders it less fit for survival (Table I). Stress is regarded here as the quantifiable effect of anthropogenic activities which reduce the fitness for survival of any of a set of biological levels of organization. The aim in environmental assessment is to determine stress in the system, but organisms can continue to survive and function although their capacity is impaired. In the long-term, this will reduce their fitness for survival but the problem in environmental management is how to determine stress and to show that a system has been, or will be, altered beyond its ' normal range ' by a particular activity. This problem is compounded in estuaries because all anthropogenic impacts operate against a high natural background variability that is often not fully understood.

Depending on the tolerances of the taxa comprising the ichthyofaunal community, both fish abundance and species diversity can provide managers with a good indication of the ' stress ' to which a particular system is being subjected. This stress can take the form of environmental and ecological impacts from all sources including the way in which a fishery is being conducted. For example, Guastella (1994) found that the catch rate of anglers in Durban Bay declined between 1976 and 1991, and attributed this decrease to factors such as loss of habitat, poor water quality, disturbance by harbour traffic and possible over-exploitation of fish stocks. An example of estuarine ' stress ' and fish population response from Europe is provided by Hamerlynck & Hostens (1994) who found that the construction of a storm-surge barrier at the mouth of the Oosterschelde Estuary resulted in a decrease in the number of anadromous fishes in that system.

Although it is contended here that fish ' health ' relates to all aspects of the biological system, at its simplest level fish health deals primarily with the causes, processes and effects of disease (Roux *et al.*, 1993). Pathological studies may include procedures such as necropsies, histological examinations, liver enzyme assays and parasitological examinations (Elliott *et al.*, 1988; Albert & Washuta, 1992). Fish health is also influenced by any adverse effects following the accumulation of heavy metals and other toxins (e.g. pesticide residues) from polluted environments. The presence of any contaminant within a fish does not indicate pollution *per se*, as the latter relies on the determination of biological damage. If the fish cannot cope with a contaminant, leading to the induction of detoxification mechanisms and tissue damage (Brown *et al.*, 1987), then this constitutes pollution. The determination of detoxification mechanisms in fishes after field or experimental exposure to contaminants is showing considerable promise as a technique for identifying contamination (Davies *et al.*, 1984).

Several studies have been conducted on the effect of various pollutants on the contaminant load carried by estuarine fishes in South Africa (Hemens *et al.*, 1975; de Kock & Lord, 1988). An example of how fishes can be used as indicators of estuarine environmental abuse can be found in the study by Blaber *et al.* (1984), who determined that juvenile Mugilidae from the Mdloti Estuary had average dieldrin levels of 49 mg kg^{-1} at a time when this pesticide was a banned substance in South Africa. The use of another pesticide, DDT, for agricultural purposes ceased in 1976 but *c.* 121 t were still being used annually for malaria control in 1985. Fish samples collected from the Kosi estuarine system in 1976 all had DDT in both the muscle and liver, with the flathead mullet *Mugil cephalus* L. having DDT concentrations of 400 mg kg^{-1} in the muscle and 860 mg kg^{-1} in the liver (Butler *et al.*, 1983). The regional health authority which was responsible for the anti-malarial spraying operations in the area subsequently instituted stricter control of the procedures used, and samples of the same fish species collected in 1981 revealed DDT values <0·05 mg kg^{-1} in every case. Although the documentation of contaminant loading within estuarine fish populations is not necessarily a direct reflection of the measurable ' health ' of individuals within that population, the overall ' fitness ' of species is likely to be adversely affected by prolonged or elevated contamination levels.

Based on the above evidence it is perhaps surprising that fish contaminant load studies have generally not been incorporated as part of routine water quality monitoring programmes in all estuarine areas. In some regions, fish bioaccumulation studies are now standard practise (e.g. in north-west Europe as the result of the Oslo and Paris Commissions' Joint Monitoring Programme) and the presence of detoxification responses, as indicated by the EROD technique detecting the induction of ' mixed function oxygenases ', has been incorporated into an integrated monitoring programme for the North Sea. The high costs and expertise required to undertake biochemical and bioaccumulation assays, however, limit their routine use. Nevertheless, when performed as an adjunct to other water quality measurements, such studies can provide a more complete picture of the ecological status and environmental condition in a particular system.

FISHES AS INDICATORS OF ENVIRONMENTAL AND ECOLOGICAL CHANGE

An innovative way of illustrating how fishes can reveal the magnitude of changes in aquatic environments over time can be gleaned from fossil research in the Grahamstown area of South Africa (Gess & Hiller, 1995). This research has provided evidence of a fish assemblage that occupied an estuarine lagoon *c.* 360 million years ago. Palaeozoic fishes found in these deposits are represented by both juveniles and adults (Anderson *et al.*, 1994), with the preponderance of juveniles indicating that southern African estuaries in the Devonian Period also acted as important nursery areas for small fishes. In other words, the fish fossil record provides strong evidence to suggest that despite major taxa extinctions, the primary ecological functioning of estuarine ichthyofaunal nursery areas has changed little over the last 400 million years (Whitfield, 1998).

The use of fishes as indicators of environmental or ecological changes to aquatic systems is based on the tenet that fish species and fish communities are sensitive indicators of change within these systems (Karr, 1981; Shamsudin, 1988). Biological monitoring is preferred to chemical monitoring because the latter often misses many of the anthropogenic-induced perturbations of aquatic ecosystems, e.g. habitat degradation. According to Karr & Dudley (1981), physical and chemical attributes of water are unsuccessful as surrogates for measuring biotic integrity. This view is supported by Oberdorff & Hughes (1992), who used a fish assemblage based index of biotic integrity (IBI) to assess water quality in the Seine River catchment. They found that comparisons between the IBI and an independent water quality index (based on water chemistry) indicated that the former was a more sensitive and robust measure of water body quality.

Hocutt (1981) suggested that structurally and functionally diverse fish communities provide evidence of water quality in that they incorporate all the local environmental perturbations into the stability of the communities. He concluded that fish communities present a viable option for assessing human-related impacts on freshwater ecosystems. Conversely, monitoring programmes measuring water quality as a surrogate for protecting fishes can also be used, provided the link between water quality parameters and fish health has been clearly established.

Despite the above findings and recommendations, it is of interest to note that fish often have a low priority as a ' tool ' in the management of estuaries. For example, ' fish and fisheries ' ranked last overall (out of 12 ' main issues ') in terms of the management of the Tagus and Humber Estuaries (Fernandes *et al.*, 1995). Similarly, until recently, fishes have not featured prominently in European Directives relating to research and management of estuarine ecosystems except those relating to accumulation of pollutants in potential food species (with the Dangerous Substances Directive) (Elliott *et al.*, 1999). As a recent departure from this, the recently adopted Water Framework Directive (European Union, 2000) requires fish communities in transitional waters (i.e. estuaries and other similar bodies) to be monitored against a reference value. That reference condition could be determined by a physical comparison with a control area (which is difficult to locate), hindcasting (which requires good previous data), predictive modelling (which requires adequate empirical or stochastic models) or expert judgement (subjective and difficult to quantify). A further difficulty, in addition to deciding how the reference condition is derived, is to decide what is a normal area. Elliott & Dewailly (1995) attempted to determine the usual estuarine fish community for European estuaries based on both a taxonomic and functional approach, illustrating that such comparisons are possible.

In contrast, it would appear that fishes are an important component of the national environment reporting system for Australian estuaries and the sea, with both ' fish populations ' and ' fish stocks ' being proposed as suitable indicators to track environmental conditions and the human activities that affect these conditions (Ward *et al.*, 1998). More recently, the development of core indicators in Australia and New Zealand has included ' estimated wild fish stocks ' and ' total seafood catch ' as possible indicators of trends in the condition of estuaries and the sea in these regions (Australian and New Zealand Environment and Conservation Council, 2000).

Within the European Union and U.S.A., the major pieces of legislation protecting the water environment and its component species, biotopes, habitats and ecosystems (European Commission, 1992; Kurtz *et al.*, 2001) are increasingly including measures of biological integrity. The Water Framework Directive and the Habitats and Species Directive of the European Commission (1992) highlights a set of estuarine species whose integrity should be protected (Elliott & Hemingway, 2002). Similar technical guidelines to evaluate the suitability of ecological indicators for monitoring programmes in the U.S.A. have been prepared by the Environmental Protection Agency's Office of Research and Development (Kurtz *et al.*, 2001). Although a considerable amount of information on the use of IBIs in North American rivers, streams, lakes and ponds is available (Simon, 1999; Seegert, 2000; Simon *et al.*, 2000), considerably less effort has been devoted to its application in estuaries, despite the concentration of human developments and impacts around these systems.

ENVIRONMENTAL AND ECOLOGICAL QUALITY OBJECTIVES AND STANDARDS

Although Elliott *et al.* (1988) showed that the setting of estuarine environmental quality objectives and standards (EQO/EQS) can be greatly facilitated by the assessment of the health of fish populations (using the ecology, pathology, biochemistry and contaminant bioaccumulation of fishes in relation to anthropogenic influences) this has seldom been undertaken. EQS are primarily environmental (e.g. concentrations of pollutants in the water column or sediment) and not ecological parameters, and although some objectives and standards for bringing about environmental improvements were linked to fish, most were not.

More recent developments have seen the emergence of ecological quality objectives (EcoQO) and ecological quality standards (EcoQS). EcoQO have been defined by Elliott (1996) as an overall expression of the structure and function of systems, and thus require greater inclusion of important ecological components such as fish communities (Table II). EcoQO were developed from the EQO approach whereby environmental statements were derived as the desired outcome for an area. These could be phrased in the form of a null hypotheses which could then be tested to determine whether the conditions were met or not (Costa & Elliott, 1991).

The relationships between ecological and environmental quality objectives and standards have been outlined by Elliott (1996). The extension of EQS to EcoQS requires further development in that EQS have usually been derived for chemicals (e.g. trace metals, DO, NH_4^+) or in biological terms for microbes (e.g. sewage pathogens), whereas EcoQS include ecological or biological health variables that are more difficult to measure and monitor. Examples of how fish species and communities can be used in the development of both ecological and environmental quality objectives, and the setting of ecological and environmental standards for compliance with these objectives, are shown in Table II.

Table II. Some examples of the ichthyological elements in EQO/EQS and EcoQO/EcoQS

Environmental quality objectives/standards	Ecological quality objectives/standards
EQO: Water quality will allow the passage of fishes at all states of the tide. EQS: Dissolved oxygen levels should always be >5 mg l^{-1}	EcoQO: Individual health of fish species does not compromise the health of the population/community. EcoQS: No excessive induction of detoxification mechanisms, and the parasite/disease incidence is within normal limits.
EQO: Fish community structure and abundance is not altered due to poor physico-chemical conditions. EQS: Resident fish community and species populations are consistent with an unpolluted environment.	EcoQO: To ensure that there is the appropriate balance of estuarine residents, marine migrants and diadromous species. EcoQS: The composition of the fish community (as diversity, species richness and biomass) is as expected for a particular estuary.
EQO: The fishes are not tainted nor have elevated levels of persistent pollutants. EQS: e.g. The level of mercury in fish flesh is not >0·3 mg kg^{-1} wet flesh.	EcoQO: The commercial fishery outside the estuary (upstream and at sea) does not affect the integrity of the estuarine fish community. EcoQS: The recruitment of juveniles of marine species using the estuary as a nursery ground is as expected.
EQO: Quality and quantity of fish is sufficient to support recreational/ subsistence/commercial fisheries. EQS: Amount and quality of fishes suitable for human consumption is above a certain level.	EcoQO: Predator-prey relationships within the fish community are not compromised. EcoQS: Quality, abundance and biomass of prey species are within certain limits and will support the prey community.

Thus biology, and especially ichthyobiology, has a role to play in defining measurable ecological objectives and standards as management and assessment tools.

ESTUARINE FISH ASSEMBLAGES AS ENVIRONMENTAL INDICATORS

Despite the fact that ichthyofaunal composition in estuaries is usually dynamic, reflecting the ever changing environmental factors and life history patterns of the various species, fish communities have often been used to illustrate changes in the condition of estuarine environments (Whitfield, 1997), particularly as they relate to organic and inorganic pollution of these systems (Elliott & Hemingway, 2002). Two well known examples from the United Kingdom include the Clyde and Thames estuaries.

CLYDE ESTUARY

Published fish records from the pristine (i.e. pre-industrial revolution) Clyde Estuary are unavailable. By 1845, however, fish populations in the upper estuary appeared to have been eliminated altogether and severely damaged in the lower estuary (Gordon, 1845). According to the 1872 Rivers Pollution Commission the demise of the fishes in the Clyde by *c.* 1850 was attributed to a long history of increasing organic and industrial pollution.

The return of fish species and changes in their abundance at several localities in the estuary coincided with reductions in organic pollution and increasing DO levels during the 1970s (Wharfe *et al.*, 1984). Further positive changes in the fish populations during the late 1970s and early 1980s were documented by Henderson & Hamilton (1986) and related to continuing improvements in water quality and recovery of the invertebrate benthos. During 1978–1979 a total of 18 fish species were recorded in the Clyde Estuary between Glasgow and Woodhall, increasing to 34 species in 1984–1985. Henderson & Hamilton (1986) emphasized, however, that although fishes had returned to most of the estuary, they were not well established throughout the year in the city reaches and several previously recorded species had not yet returned. Species presence alone is insufficient to determine the health of the community but the return of salmonids to the Clyde catchment, and the regular recording of migratory runs, is indicative of a rapidly recovering system (Mackay & Doughty, 1986). The risk of severe dissolved oxygen depletion in the estuary still existed in the late 1980s and was perceived to be a major threat to the safe seaward migration of salmon *Salmo salar* L. smolts and the prevention of the establishment of a normal estuarine fishery (Henderson & Hamilton, 1986).

THAMES ESTUARY

The sequence and timing of the decline and recovery of the Clyde fish populations were closely paralleled by events in the Thames Estuary. Evidence of the final decline can be gleaned from the writings of Yarrell (1836), who recorded the collapse of the smelt *Osmerus eperanus* (L.) fishery in the tidal Thames due to ‘ the state of the water ’ in this portion of the estuary. Similarly, salmon were also in the process of being eliminated from the Thames Estuary in the early 1800s, with Yarrell (1836) stating that ‘ The last Thames salmon I have a note of was taken in June 1833 ’. While smelt and salmon fisheries were of great value in the tidal and middle Thames reaches, the eel *Anguilla anguilla* (L.) supported a fishery of at least equal importance along the whole of the Thames and its tributaries. Although large catches of eels were still being recorded in the early 1800s, by the end of the century navigation locks and pollution had reduced eel populations to such an extent that most of the stocks sold on London markets were being imported from Holland (Wheeler, 1979). Another abundant migratory fish was the twaite shad *Alosa fallax* (Lacepède), which by the end of the nineteenth century was not found in the middle or upper reaches of the tidal Thames. Although the decline and demise of many of the above fisheries may be linked to increasing pollution levels in the Thames, overfishing may also have played a role. This overfishing applied not only to the target species but also to the bycatch of such operations, e.g. whitebait fishing targeted mainly sprat *Sprattus sprattus* (L.) and herring *Clupea harengus* L. but numerous fishermen gave evidence to the 1879 Buckland and Walpole Fisheries Enquiry as to the harmful effects of this activity on the fry of other fish species (Wheeler, 1979).

TABLE III. A simple estuarine classification system based on the responses of the two major fish community components to anthropogenic changes in water quality and quantity

Estuary class	Description	Migratory fish component	Resident fish component
A	Excellent	Water quality and quantity allows free passage of key indicator species.	Resident fish community normal.
B	Good	Water quality and quantity allows free passage of most key indicator species.	Resident fish community slightly modified.
C	Fair	Water quality and quantity restricts passage of most key indicator species.	Resident fish community modified.
D	Poor	Water quality and quantity allows little or no passage of key indicator species.	Resident fish community impoverished.

According to Wheeler (1979), the turn of the twentieth century saw improvements to the Thames (following attempts to treat London's sewage) and resulted in the return of some species of fish to the London region of the Thames Estuary. It was only during the 1970s, however, that species sensitive to anthropogenic environmental changes returned to the middle and upper Thames Estuary. Documentation of the return of fish species in the Thames was greatly facilitated by the collection of specimens from cooling water screens at various power stations along the banks of the estuary. This was highlighted by the capture of a live salmon in November 1974 on the screens of the West Thurrock power station and was followed in succeeding years by further specimens, one of which was collected by anglers above the tidal limit (Wheeler, 1979). As was the case with the Clyde Estuary, species presence alone is insufficient to determine the health of the fish community. Andrews & Rickard (1980) recorded an increase in both the abundance and variety of fish species at all seasons in the Thames, thus reinforcing the view that the estuary was being successfully rehabilitated. Supporting evidence for the recovery and breeding of smelt populations in the Thames Estuary during the late 1970s is provided by P. Hutchinson (pers. comm.). A regulated eel fishery commenced operations in 1980 (M. Pilcher, *pers. comm.*) providing further evidence of this rehabilitation process. Sprat were found as far upstream as Greenwich in 1996 and large shoals now feature regularly in the commercial catches from the outer Thames Estuary (S. Colclough, C. Dutton, T. Cousin & A. Martin, *pers. comm.*).

In hindsight, fish information from both the Clyde and Thames estuaries could have been used to construct a very simple classification system (Table III) that would have documented the most recent decline and recovery of these systems. Available information from both estuaries suggests that these systems deteriorated to a Class D status in the early to mid-nineteenth century before recovering to a Class C status during the mid to late twentieth century. Further improvements in water quality and associated environmental conditions would result in continued recovery in both the migratory and resident fish communities, such that these systems would then approach a Class B status. This estuarine classification scheme, a subset of that used by the U.K. environment protection agencies (Scottish Environment Protection Agency, unpubl. data), provides a summary of fish status in the system. By concentrating only on community structure it does not, however, include all elements of fish biological health. By inclusion of individual species elements (Table IV) into such a scheme, the final value of the assessment would also be enhanced.

Other studies in the U.K. have used changes in species abundance to document the recovery of estuarine systems, e.g. Potter *et al*. (2001) documented an increase between the 1970s and 1990s in the annual fish catches from the intake screens of the Oldbury Power Station in the Severn Estuary. These authors suggested that the marked increases in abundance of species such as sand goby *Pomatoschistus minutus* (Pallas), whiting *Merlangius merlangus* (L.), bass *Dicentrarchus labrax* (L.), thin-lipped grey mullet *Liza ramada* (Risso), herring, sprat and Norway pout *Trisopterus esmarkii* (Nilsson) reflects the great improvement that occurred in the water quality of the Severn Estuary between these decades (Little & Smith, 1994).

SOME FISH COMMUNITY INDICES USED IN ESTUARIES

Fish species richness and diversity in an estuary can directly assist in the evaluation of the importance and condition of a system. Ichthyofaunal surveys are generally presented as lists of species and their relative abundances, which require specialist interpretation and are usually beyond the comprehension of most coastal managers and planners (Cooper *et al*., 1994). A method of condensing these data into a more functional format is essential if this information is to be used in any planning or management process. A standard method of condensing biological community information is through the use of a single or composite index, some examples of which are described below.

ESTUARINE COMMUNITY DEGRADATION INDEX (CDI)

The CDI developed by Ramm (1988) is based on a comparison of the fish community present within an aquatic system, to the community that would exist in the absence of, or prior to, degradation. The index assumes that differences between the potential community and the present assemblage are due to habitat degradation. The CDI has been applied to South African estuaries (Ramm, 1990). A total of 62 KwaZulu-Natal systems were first classified into six groupings based upon eight physical and hydrological parameters. This classification procedure involved the use of detrended correspondence analysis, two-way classification techniques and principle components analysis. Since the entire biological community could not be sampled, the fish assemblage of each estuary was selected to represent the overall community in the analysis. Reference ichthyofaunal lists were then developed for each of the physical groupings, and CDI values were calculated for each system by comparing the reference faunal list with a species list from biological surveys on that particular estuary. Computed CDI values for KwaZulu-Natal estuaries ranged from 0·2 (undegraded) to 8·2 (severely degraded).

TABLE IV. Fish-based parameters that could be used in a single or composite scoring system (the higher the score, the more natural the system) for monitoring human induced changes within an estuary. Some of the indicators are subjective and qualitative whereas others are more objective and quantitative

Level	Indicator	Value	Score
1. Fish species	1(a) Species abundance/biomass	Artificially low	1
		Medium/high	3
	1(b) Keystone/indicator species	Present	3
		Absent	1
	1(c) Alien/introduced species	Presence of alien/introduced species	1
		Absence of alien/introduced species	3
	1(d) Fish species health	Toxic accumulations present	1
		Toxic accumulations absent	3
2. Fish community	2(a) Harrison *et al.* (2000)	Similarity with mean number of taxa:	
	Species richness index	>95% upper CI	5
		Within 95% CI	3
		<95% lower CI	1
	2(b) Harrison *et al.* (2000)	Similarity with reference condition:	
	Bray-Curtis presence/absence similarity index	>50th percentile similarity	5
		10th–50th percentile similarity	3
		<10th percentile similarity	1
	2(c) Harrison *et al.* (2000)	Similarity with reference condition:	
	Bray-Curtis percentage abundance similarity index	>50th percentile similarity	5
		10th–50th percentile similarity	3
		<10th percentile similarity	1
	2(d) Deegan *et al.* (1997)	EBI value (eight metrics used):	
	Estuarine Biotic Integrity Index (number and biomass)	Score 31–40	5
		Score 21–30	3
		Score 0–20	1

The Sezela Estuary was described by Begg (1978) as the most severely polluted estuary in KwaZulu-Natal. During the 1970s it was essentially devoid of fish life and consequently had a CDI of 9–10 (Ramm, 1990). As a result of concerted efforts to correct the problems in the Sezela Estuary between 1982–1984, the aquatic community began to recover. By 1984 surveys indicated a CDI of *c.* 8, by early 1986 the CDI was *c.* 6, and in 1987 it had improved to *c.* 5 (Ramm, 1990). This pattern of declining degradation clearly demonstrates the advantage of using this method to monitor the recovery of an estuary. On the same basis, the CDI could be used to document the faunistic degradation of an estuary over time and assist in the identification of types of estuaries where the fish communities are most threatened.

ESTUARINE BIOLOGICAL HEALTH INDEX (BHI)

Whereas the CDI measures the degree of dissimilarity (degradation) between the potential community and the actual community, the BHI developed by Cooper *et al.* (1994) modifies the CDI to incorporate a measure of the degree of similarity between the potential community and the actual community. The formula for calculating the Biological Health Index is $BHI = 10(J)[\ln(P)\ln(P_{max})^{-1}]$, where J is the number of species in the system divided by the number of species in the reference community, P is the potential species diversity (number of species) of each reference community and P_{max} is the maximum potential species diversity (number of species) from all the reference communities. The index ranges from 0 (poor) to 10 (good). Reference communities are usually determined by establishing the normal range of fish community components such as presence and absence of taxa in the most unimpaired waters representative of the area or region under consideration.

Although the BHI has proved a useful tool in condensing information on estuarine fish assemblages into a single numerical value, the index is only based on presence and absence data and does not take into account the relative proportions of the various species present. Furthermore, the BHI formula incorporates two separate measures, health and importance, and combines them into a single index. The health component (J) is a measure of the degree to which the present condition of an estuary deviates from some reference condition, while the importance component ($\ln P \,.\, \ln P_{max}^{-1}$) reflects its contribution to the region as a whole (Cooper *et al.*, 1994).

ESTUARINE FISH HEALTH INDEX (FHI)

More recently, a new series of indices have been developed by Harrison *et al.* (2000) that focus on both qualitative and quantitative comparisons with a ‘ reference ’ fish community. In the qualitative assessment, the number of species in each estuary is compared to the average number for the group to which it belonged. Each estuary is then rated according to whether the number of taxa exceeds the average (>95% upper CI), approximates the average (95% CI) or is below the average (<95% lower CI) of its reference group. Scores of 5, 3 and 1 are assigned to each of the ratings respectively (Table IV). The species assemblage of each estuary can then be compared with a reference assemblage of each estuary type based on the most frequently captured taxa. The most frequently captured species from each group of estuary types corresponding to the 95% CI of the mean number of taxa is selected as the reference. The species composition of each system is then compared to the reference assemblage using the Bray–Curtis measure based on presence and absence (Table IV).

In the quantitative assessment, the per cent abundance of the species within each estuary is compared to the per cent abundance of the species captured in the group to which it belongs using the Bray–Curtis similarity measure (Table IV). In both the qualitative and quantitative assessments, non-indigenous fish species are included in the fish assemblages for each estuary type, but are excluded from the reference condition. The reason for this is that the contribution of non-indigenous species to the fish community structure of an estuary is indicative of a deviation from the norm, particularly in terms of relative abundance (Harrison *et al.*, 2000).

ESTUARINE BIOTIC INTEGRITY INDEX (EBI)

The EBI developed by Deegan *et al.* (1997) is another useful fish indicator of estuarine ecosystem status which reflects the relationship between anthropogenic alterations in the ecosystem and the status of higher trophic levels. Their EBI was based primarily on fish trawl catches and included the following eight metrics: total number of species, dominance, fish abundance (number or biomass), number of nursery species, number of estuarine spawning species, number of resident species, proportion of benthic-associated species, and proportion of abnormal or diseased fish (Table IV).

Deegan *et al.* (1997) tested their EBI in two Massachusetts estuaries with different levels of habitat degradation. Fish assemblages in low-quality sites had lower number of species, density, biomass, and dominance compared to medium-quality sites. In addition, the abundance of fishes using estuaries as a spawning and nursery area was lower in low-quality sites compared to medium-quality sites. The individual metrics and the overall index were found to correlate with habitat degradation and the EBI based on biomass was not an improvement on the EBI based on number, indicating that the extra effort to obtain biomass was not warranted.

ESTUARINE FISH RECRUITMENT INDEX (FRI)

The FRI was developed by Quinn *et al.* (1999) in an attempt to use ichthyological information to assess changes in habitat integrity (as influenced by changes in river flow), especially the availability and suitability of estuarine nursery areas to marine migrant fishes. This need arose in South Africa as a result of the increasing competition for scarce water resources, and the need for river flow requirements of estuaries to be well articulated so as to assist in determining the optimal scheduling of freshwater allocations. The development of a fish recruitment index which will adequately reflect the suitability of freshwater release policies for fish populations utilizing estuaries was perceived to be a major need by both managers and scientists.

A major requirement of the FRI was that this index should be biologically meaningful and be readily understandable by biologists and water resource managers alike. The FRI is a management directed index that is based on the integration of three key information sets. The first set is the current understanding relating to the importance or significance (dependency score) of marine fish species in a particular estuarine environment and whether it is endemic to southern African waters or not. The second is the preferred timing of the immigration period for a particular species (optimal recruitment score). The third information set incorporates known environmental requirements for the recruitment by juvenile marine fish into southern African estuaries. Details of the formula and practical application of the above index in a temporarily open and closed and permanently open South African estuary are given by Quinn *et al*. (1998, 1999). In both systems, a variety of river flow situations were examined and the predicted changes in the magnitude of marine fish recruitment into these estuaries were assessed.

CONCLUSIONS AND FUTURE RESEARCH DIRECTIONS

The conceptual and qualitative understanding of changes within estuarine fish assemblages is good but the quantitative understanding is still poor. Although scientists have the ability to make policy decisions based on a qualitative approach, there is a reluctance to act decisively because of what is perceived to be inadequate data sets. There is a strong argument for scientists to provide input into environmental management issues, otherwise decisions will be made without the benefit of their insights.

The basis for using biological monitoring of fishes to assess environmental condition is that the relative health of a fish community is a sensitive indicator of direct and indirect stresses on the entire aquatic ecosystem (Fausch *et al*., 1990). Ideally, studies should determine stress at the community, population, individual, physiological and subcellular level, using techniques such as production ecology, biochemistry, bioaccumulation studies, pathology, genetics, behaviour and physiology. In addition to this being a sensitive indicator and scientifically and biologically relevant, it is argued that it is more meaningful in a policy, management and public perception manner. The present authors emphasize further the earlier recommendation (Elliott *et al*., 1988) that fish studies should be one component in a holistic approach, using chemical, hydrographical and other biological data to interpret the fish results. Although the repercussions at higher (human) levels as the result of changes in fishes and their populations are beyond the scope of this paper, it is recognized that a breakdown in ecological integrity also has socio-economic implications.

It is noteworthy that within southern African estuaries, a number of ‘ fish orientated ’ approaches have been adopted by scientists to assess environmental change, depending on the type of information required by planners and managers. In addition to the various indices outlined above, monitoring of fish pathology, trends in anglers catches, relative abundance of juvenile marine fishes in estuaries and sudden fish mortalities can all provide insights into the health of these systems. A combination of these techniques would assist in the identification of potential threats to the estuarine biota, especially the fishes. Although most current indices depend on juvenile and adult stock information, the increasing data on larval fish dynamics in estuaries (Hall, 1987) bodes well for future ecological and environmental health indices.

Increasingly, modern legislation and agreements attempt to safeguard the health and integrity of ecosystems, and the determination of carrying capacity is an integral part of this. The carrying capacity of any system can be for any of the defined resources required by organisms, but principally for space and food. Carrying capacity is also influenced by the number of available niches and thus the protection of biotope complexes becomes an important issue. There is therefore a need for estuarine fish studies to measure carrying capacity and the effects of stress at all biological levels affecting that capacity, e.g. behavioural change caused by pollution leading to changes in food and space utilization.

This review, together with assessments presented in Elliott & Hemingway (2002) and Fausch *et al*. (1990), all suggest that future research in biological monitoring by means of fish communities should focus on: (a) standardization of methods of sampling and data analysis; (b) documentation of natural variation in fish communities, against which changes due to degradation can be compared; (c) experimental manipulation to test assumptions underpinning indices. Although there is extensive information for some areas, including southern Africa and Europe as shown here, the necessary infrastructures to conduct biological monitoring of fish populations need to be established in a range of biogeographic regions throughout the world. Aspects to be addressed by these regional institutions would include community assessments, fish health indices, bioassessment protocols, acute and chronic bioassays, pathology and tissue residue assessments. The information generated from a broad spectrum of biomonitoring options in both estuaries and their associated catchments would facilitate an increase in ecologically focused management of estuaries, thus benefiting both fishes and people who utilize these waters.

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