

Assessment of Techniques for Determining the Health of Tropical Estuarine Ecosystems

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Acronyms Used In This Report

AIMS Australian Institute of Marine Science
CSIRO Commonwealth Scientific and Industrial Research Organisation
FQAI Floristic quality assessment index
IBI Index of biological integrity
JCU James Cook University
MTSRF Marine and Tropical Sciences Research Facility

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Executive Summary

Estuarine ecosystems around the world are suffering the deleterious effects of human activities, and the threats to the ecosystem health of estuaries are likely to be amplified by the global climate change. Increased understanding of these threats has prompted a widespread catchment-to-coast focus on understanding ecosystem health, with explicit recognition of the pivotal role of estuaries. Understanding the forces affecting the health of tropical estuaries and of methods for measuring environmental health is crucial for developing appropriate and effective environmental management strategies. While there is a large amount of information about detecting impacts and measuring ecosystem health in temperate estuaries, the extent to which temperate approaches are transferable to tropical/subtropical systems is unclear. There have been no location-specific studies evaluating the appropriateness of extrapolation from temperate to tropical understanding. In particular, biochemical processes such as toxicity, persistence and accumulation rates are likely to differ between cooler temperate and warmer tropical systems). Contrasts in functioning of tropical compared to temperate estuaries are likely to be compounded by the much higher biological diversity present in tropical estuaries, which potentially leads to more complex ecological processes. High diversity might also equate to high variability, adding another layer of complexity.

Ecosystem health is a combination of three factors: resistance, organisation and vigour, all of which are ultimately functions of ecological processes. Ecosystem health might therefore be considered as more closely aligned with the integrity of ecological processes than the health or abundance of individual species or groups of species. Definitive measurement of ecosystem health therefore requires approaches that provide measures of the integrity of ecological processes.

Objective

This study assessed techniques that can be employed to determine the ecosystem health of estuaries and coastal wetlands in Australia's tropical regions, evaluated the sensitivity of those techniques to detect the effects of specific stressors, and evaluated their ability to separate natural variations from deleterious anthropogenic impacts.

Techniques and methods of biological assessment

Methods of detecting biological responses to stressors are as wide as the range of responses themselves, and encompass a continuum of options from reliance on a **single species** to assessment of entire biological communities. Single species methods are best suited to the detection of specific stressors but, although they provide cost-effective measures, their ability to detect a broad spectrum of impacts or to reflect the overall integrity of ecological processes is limited.

Multi-species approaches provide a greater capacity to assess higher ecological functions. The use of **multiple (indicator) species** as proxies for ecosystem-level reporting substantially expands the level of information available, but still suffers from most of the limitations of single species approaches. **Index-based** multiple species approaches and **multi-metric methods**, specifically incorporating many taxonomic groups, produce more realistic measures of ecosystem health, and potentially detect a broader spectrum of stressors than simple indicator species approaches. However, because they distil information into a single summary index they lack the ability to link effects with specific causes, and often fail to highlight valuable assemblage level information. Assemblage/community level methods using **taxonomic** and, even more so, **functional groups**, provide the opportunity for a more holistic assessment of ecosystem health. They provide powerful methods of

assessing the integrity of ecological processes, the attributes most closely aligned with ecosystem health.

Specific applications

Studies focused on specific stressors have been useful for directly assessing impacts of those stressors, but extrapolation of results to broader ecological contexts has been difficult. The greatest advances towards more holistic assessments of ecosystem integrity are emerging from work assessing the environmental flows required to support estuarine functioning. In this area, the complexity involved in understanding impacts of altered flows has been recognised and embraced. In contrast, advances in contaminant research have mainly been confined to technical factors and recognition of “new” contaminants in the environment. Although our knowledge of lethal/sub-lethal effects of contaminants has also advanced considerably, the broader ecological impacts of contaminants on estuarine ecosystems are not well understood, except in extreme circumstances.

The way forward

Early warning of ecosystem change from human impacts requires: a) a clear understanding of the task and potential problems, b) clear and ecologically meaningful biological objectives, and c) a clearly enunciated plan that focuses on gaining the knowledge necessary to substantially advance our understanding. In developing techniques for determining the health of estuaries and coastal wetlands in Australia’s tropical regions, we therefore need to not only develop methods of detection, but also advance our understanding of the links between stressors and environmental outcomes.

Estuarine indicator work in the MTSRF project “Marine and estuarine indicators and thresholds of concern” will focus on fish. It will be guided by previous work using fish as indicators of estuarine health in South Africa, and work in southern Queensland in freshwater systems. These previous efforts have mainly used measures of organisational structure as indicators of ecosystem health, especially indices that summarise the differences between fish species expected to be present against those actually sampled. We intend to extend work on fish as indicators beyond this to include ecological processes that, together, should offer a more comprehensive indication of ecosystem health. We have conceptualised three estuarine fish indicators likely to represent important components of ecosystem health: scavenging pressure, non-detritivore trophic composition, and predator success.

The combination of a range of techniques and their use at multiple sites in close proximity but having different potential impacts, provides an opportunity to greatly advance our ability to assess the health of estuaries and coastal wetlands of Australia’s tropics.

Introduction

Over the past 50 years estuarine ecosystems around the world have faced ever accelerating ecological change (Ong 1995, Kennish 2002). Initially, this increased rate of change was largely a function of the fact that estuaries occur at the confluences of rivers and the sea, areas that are particularly attractive for urban and industrial development (Kennish 2002, Smith et al. 2001). More recently, the pace of change and level of future threat has been amplified by the increasing impacts of global climate change (Sheaves et al. in press a). Increased understanding of these threats has prompted a widespread catchment-to-coast focus on understanding ecosystem health. With this focus has come explicit recognition of the pivotal role of estuaries in the catchment-to-coast continuum (e.g. Hutchings et al. 2005, Gehrke & Sheaves 2006). However, despite this recognition and the fact that impacts on estuaries are increasing at the fastest pace in tropical countries (Blaber 1999), little attention has been paid to understanding or measuring the health of tropical estuaries. It is important that we improve our understanding of the forces affecting the health of tropical estuaries, and of methods for measuring environmental health and detecting environmental impacts in the face of the unique characteristics of tropical estuaries.

The driver for management, science and the public is the need understand how particular stressors impact on estuarine health. Understanding biological responses to environmental change is crucial for developing appropriate and effective environmental management strategies (Kennedy & Jacoby 1992, Cross & McInerney 2005, Isaak & Thurrow 2006). However, gaining a detailed understanding of biological responses has proved elusive, even in well-studied systems. Considering the detection of biological response to different environmental conditions is the focus of most environmental monitoring programs and ecological studies, there should be a large knowledge base that could provide direct cause and effect information. The knowledge base is certainly sufficient to indicate that if habitats are modified sufficiently there will be substantial shifts in community composition (Connolly et al. 2005), the loss of species from the area (Williamson et al. 1994) and impacts on local food web function (Waltham & Connolly 2006). However, detecting biological response to specific types of environmental change and at more subtle scales has proved more difficult. Although it is critically important to be able to detect biological change and attribute that response to particular causal factors, this has often proved a complex problem. One difficulty is that it is not usually known how much change a system can absorb without reaching a critical tipping point where biological change results (Kennedy & Jacoby 1992). Moreover, it is often not clear which biological components of the system will respond to particular types of change first, what consequences those initial biological changes may have for the functioning of the system, or how long before those initial changes are expressed in other biological components (Archer & Newsom 2002, Estevez 2002, Lohrer et al. 2004). Such information can only be determined once biological responses are understood at all levels and perspectives; individual, species, population, assemblage and functional groups. Rapid degradation of estuarine and coastal wetland habitats from a broad spectrum of causes means there is an imperative to understand biological response to all types of environmental change (Lee et al. 2006). In tropical Australian estuaries, key issues include environmental contamination by pollutants (nutrients, toxicants) and changes to freshwater inflows. In turn, pollutant contamination and changes to flow interact strongly with each other and with other factors, particularly habitat modification/loss and overexploitation of resources, which both influence contamination and flows, and are influenced by them (Fig. 1).

Consequently, as well as being key issues in their own right, these confounding factors must be considered as important covariates when measuring the effects of pollution and flow. Unfortunately, although these factors are usually under strict government regulation in Australia, there is generally little information about the biological responses they elicit in

tropical coastal ecosystems. As a consequence, in the past most understanding has arisen via extrapolation from temperate systems.

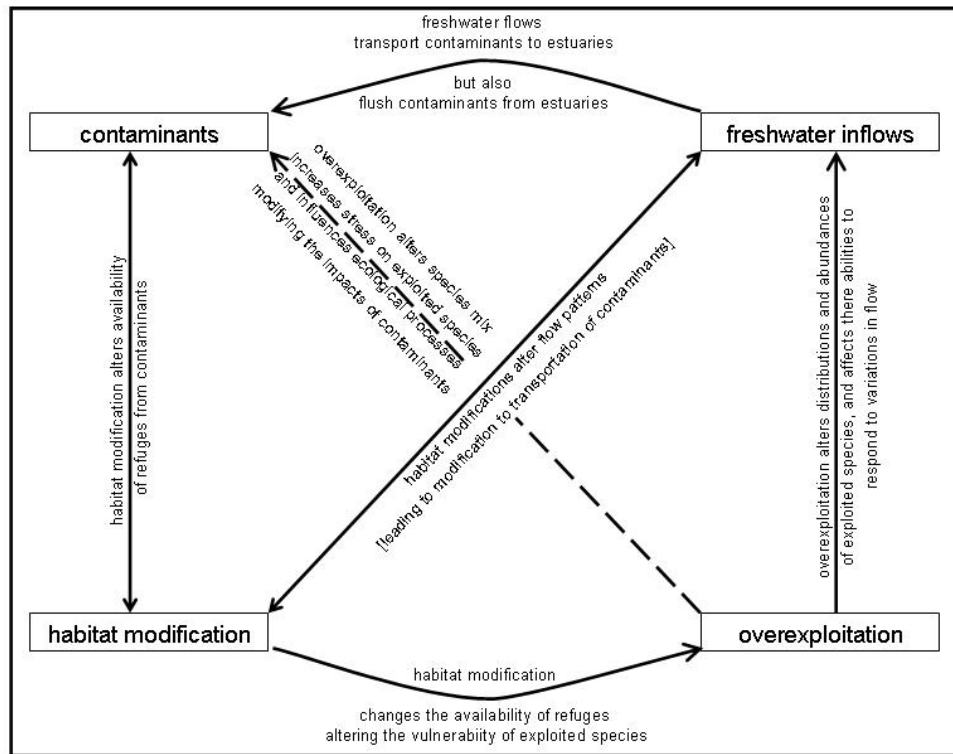


Figure 1: Some of the interactions between some major stressors of tropical estuarine ecosystems.

There has been far less reporting of successful detection of response to stressors in tropical estuaries than for tropical freshwater or temperate estuaries. Much of the recent tropical research in northern Australia (principally water quality based) has been focused at a catchment-to-reef scale but has lacked defined links to biological function, particularly for estuary components (Moss et al. 2005). Consequently, the ways in which stressors impact on tropical estuary function and the biological consequences of those impacts have been largely overlooked. This lack of success is symptomatic of the lack of tropical estuary research in general, and is consistent with an emerging theme in this report, the lack of integration of physical (e.g. water quality) and biological data.

The diverse temperate estuary literature provides valuable information about the strengths and weaknesses of different approaches to detecting changes in environment function. The details of much of this broad literature have been reviewed extensively (Table 1). However, the extent to which temperate approaches are transferable to tropical/subtropical systems is unclear. Extrapolation of temperate information to tropical systems appears risky because there is no location-specific understanding to allow evaluation of the appropriateness of this extrapolated understanding. In particular, biochemical processes such as toxicity, persistence and accumulation rates are likely to differ between cooler temperate and warmer tropical systems (Biney et al. 1987). Contrasts in functioning of tropical compared to temperate estuaries are likely to be compounded by the much higher biological diversity present in tropical estuaries (Robertson & Blaber 1992) which leads to more complex ecological processes (Sheaves 2005). High diversity might also equate to high variability, adding another layer of complexity. Variability can make detection of responses due to

stressors difficult to segregate from natural variability, thus constraining interpretations (Neckles et al. 2002, Stewardson et al. 2004).

In the past there has been considerable success in detecting species level responses to particular stressors, and there is an enormous volume of literature documenting both natural and human-induced changes in most of the measurable parameters in (temperate) estuaries (MacFarlane & Booth 2001).

Table 1: Recent reviews discussing the use of environmental indicators as proxies for aquatic environmental health.

Author/s	Subject of review
Boltovskoy et al. 1991	Foraminiferal test morphology as an indicator of change in ecological parameters.
Alve 1995	Foraminiferal responses to pollutants in estuaries.
Hilty & Merenlemer 2000	Selection of indicator fauna.
Seegert 2000	Indices of biotic integrity and associated sampling protocols.
Brown 2001	Assessing river condition using benthic macroinvertebrate assemblages.
Sures 2001	Parasites as bioindicators of heavy metals in aquatic ecosystems.
Carignan & Villard 2002	Selecting indicator species to monitor ecological integrity.
Estevez 2002	Biotic variables and analyses in estuarine inflow studies.
Whitfield & Elliot 2002	Fish as indicators of environmental change in estuaries.
Fichez et al. 2005	Indicators of pollutants in coral reef lagoon systems.
Stuer-Lauridsen 2005	Passive accumulation devices for monitoring organic micropollutants in aquatic environments.
Tom & Auslander 2005	Transcript and protein environmental biomarkers in fish.

This is evidence enough that detecting change is not the challenge; rather the challenge is in understanding how those changes impact the ecological integrity of a system. Although most studies successfully achieve their stated aims, the conceptual level of most of the conclusions does not match the complexity of environmental systems (Elliott 2002). In essence, much detection takes a reductionist point of view; measuring levels of individual stressors or changes in individual species, without consideration of interactions between stressors or impacts on estuarine ecology in a more holistic sense. This reductionist approach means we have a lot of information at sub-ecosystem levels, but we have little idea how to integrate this understanding into broader assessments of ecosystem integrity. Water quality data provide a good example of this problem. Water quality is relatively quick and easy to measure and changes in quality can be linked to a range of potential stressors (Moss et al. 2005). Consequently, water quality is frequently used as a proxy for estuary health (Whitfield & Elliott 2002, Cox et al. 2005). However, the lack of understanding of biological responses to changes in water quality at a variety of scales makes it clear that measuring water quality alone will not provide the necessary information to reliably assess the integrity of ecological function in tropical and subtropical estuaries (Whitfield & Elliott 2002). We can measure changes in water quality but our understanding of the ramifications of those changes is limited to direct effects on a few species (Moss et al. 2005), or to more general biological responses to extreme events such as development of anoxic conditions (Osgood & Stiegler 1990, Zou & Steuben 2006, O'Connor & Whitall 2007).

Recent development of “holistic” indices has provided a positive step away from reductionist approaches and towards ecosystem-level understanding. These indices are typically compiled from a larger range of variables, thereby providing much greater representation. Such index-based compilations are management-friendly because they distil complex ecological information down to a simple index (Whitfield & Elliott 2002). However, these approaches are better termed “quasi-holistic”, because they do not really address ecosystem-level assessments comprehensively. Substantial information is lost in integrating complex data into indices (Maddock 1999, Seegert 2000), meaning outputs do not represent the diversity in ecosystem functions in a truly holistic fashion (Kennard et al. 2006a).

In essence, under reductionist approaches, and even under quasi-holistic approaches, we are left with a poor understanding of the impact on overall ecosystem integrity, even though this is what usually motivate such analyses. We use the term ecosystem integrity as a synonym for ecosystem health. Ecosystem health can be seen as the combination of three components of systems (although many variations on these have been suggested): resistance, organisation and vigour (Rapport et al. 1998). Although organisation is usually measured in terms of variables such as species richness or diversity, and vigour in terms of rates of processes, all three components really result from the conservation or integrity of ecological processes. In a conceptual sense, then, ecosystem health is more closely aligned with the integrity of ecological processes than the health or abundance of species or groups of species. Consequently, the definitive measurement of ecosystem health really requires approaches that provide measures of the integrity of ecological processes.

Objective

This study assessed techniques that can be employed to determine the ecosystem health and integrity of estuaries and coastal wetlands in Australia’s tropical regions, evaluated the sensitivity of those techniques to detect the effects of specific stressors, and evaluated their ability to separate natural variations from deleterious anthropogenic impacts.

Techniques And Methods Of Biological Assessment

Methods of detecting biological response to stressors encompass a continuum of options from reliance on a single species to assessment across entire biological communities. These reflect the range of different responses that are theoretically possible (Fig. 2). Within the assessment continuum a range of approaches has been used to detect ecological impacts, however most of these can be summarised into two principal categories: biochemical (bioaccumulation of stressors and/or measurable biochemical changes due to stressors) and avoiders of stressors (organisms that relocate in the presence of stressors). Both approaches can be used to measure biological responses to stressors from the level of a single species to a whole-of-ecosystem level. Biochemical approaches are primarily used to measure impact on in situ (sessile) organisms, whereas avoider approaches are used when in situ or mobile organisms are absent (e.g. due to recruitment failure).

Levels of investigation

i. Single Species Approaches

Single species can be used in a variety of contexts, from basic biochemical studies that determine if a particular stressor is present in a system, to an indicator role where one species is used as a proxy for higher level ecosystem condition (Table 2).

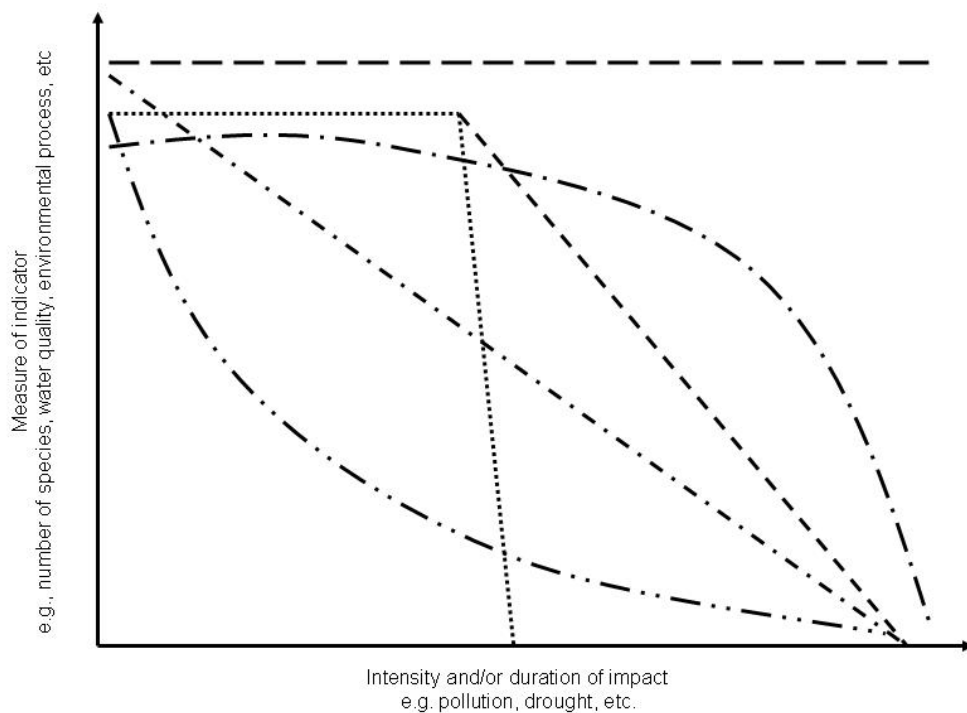


Figure 2: Theoretical responses to stressors in an estuarine environment. (Source: R. Pearson).

The use of a single species to assess impacts is best suited to the detection of specific stressors. This method has the advantage that experiments can be employed to document specific responses to a particular stressor (Everitt et al. 2002, Clavier et al. 2005). However, those responses are usually obtained in isolation from potential interactions with other environmental stressors (Das 2000). Although attractive to managers because they provide cost and time-effective measures (Carignan & Villard 2002), the single indicator species

method is limited by its inability to detect a broad spectrum of potential impacts. Information derived from a single indicator species can provide an alert to the presence of stressor/s in the system of interest, however on its own does not provide evidence of impairment of ecological function. Evidence of impaired function is limited to the indicator species itself because niche partitioning (Gause 1934) means that the status of a single species will rarely be indicative of that of its ecosystem (Rapport 1990). Single species approaches are often studied with biochemical approaches, however it is also possible to use an avoider-based approach.

Biochemical-based methods can be subdivided into two general categories: bioaccumulation and biomarkers. Bioaccumulators are organisms that incorporate contaminants into their tissues, while biomarkers are chemicals, such as hormones, that are produced in an organism's body in response to stressors. For biochemical studies, sessile organisms provide useful information about specific events and are useful for tracking long-term trends in levels of particular contaminants (Markich et al. 2002, Lazareth et al. 2003). The value of mobile organisms for biochemical studies is less certain because of the difficulties of establishing if the organism was present at the time of potential exposure to the stressor (Hilty & Merenlender 2000), although fish have proven useful in tracking sewage plumes within estuaries in southeast Queensland where their home ranges are relatively small (Schlacher et al. 2005). Both bioaccumulator and biomarker approaches are frequently used as a proxy for water quality (Whitfield & Elliott 2002). In this way, the biochemical approach is used as an index of overall environmental health, although information is really limited to conclusions about the particular contaminants examined. Bioaccumulation-based approaches generally use no more than a small suite of species. This usually means easier data acquisition in the field but may require highly specialised laboratory procedures (Whitfield & Elliott 2002). However, using biomarkers such as stress hormones can require specialised and time consuming field techniques (e.g. Katsiadaki et al. 2002, Hallare et al. 2005) making it less attractive logistically as well as limiting the amount of data that can be collected. On a positive note, biomarkers can be successful and relatively easy methods of detecting contaminants that are difficult to detect with alternative techniques (Humphries et al. in review).

Organisms that avoid sub-optimal conditions are by necessity mobile. Use of avoiders is often assemblage-based and considers shifts in assemblage composition (relative abundances, species richness, size distribution) as indicative of changes in environmental health (Whitfield & Elliott 2002). This approach works best when data are available that pre-date the stressor and provide reliable information about reference states for the system of interest (Reynoldson et al. 1997, Stewardson et al. 2004). Without reliable information about reference states, preferably long-term data, the ability to detect changes may be limited to extreme deviations because of the difficulties in isolating more subtle changes from background variability (Boulton 1999, Neckles et al. 2002, Stewardson et al. 2004). Avoider-based data may require greater field commitment of time and equipment than biochemical-based approaches, however this cost can often be offset by having little or no requirement for laboratory studies.

Table 2: Summary of applications and limitations of different approaches for assessing environmental integrity.

Level of investigation	Has been used for detection of specific stressors	Success at detecting specific stressor	Ability to detect a broad spectrum of impacts	Limitations
<i>Single species approaches</i>				Little biological application beyond specific stressor.
Biochemical	Yes	Good	Poor	
Avoider	Yes	Good	Poor	
<i>Multiple species approaches</i>				Lack specific causal information about ecosystem change.
Multiple indicator spp.	Yes	Good	Good	
Index-based	Yes	Good	Good	
<i>Assemblage level approaches</i>				Reference states may be difficult to determine. May increase field and analytical difficulties (offset by level of information gained).
Taxonomic groups	Yes	Good	Good	
Functional Groups	Infrequently	Good	Good	
<i>Process level approaches</i>				May increase field and analytical difficulties (offset by level of information gained).
	Infrequently	Good	Good	

ii. Multiple Species Approaches

Multiple species approaches are generally just extensions of the single species approach. The advantage of multi-species approaches is that they provide a greater capacity to report on higher ecological functions than single species approaches.

The use of **multiple (indicator) species** as proxies for ecosystem-level reporting substantially expands the amount of information gained. Apart from the obvious advantage of opportunities to monitor a greater spectrum of potential stressors, this method allows for comparisons of responses among species (Carignan & Villard 2002). In particular, it provides a greater potential for isolating responses to stressors from natural background variation. The use of multiple indicator species has similar limitations to a single species approach in that outcomes should generally be constrained to the specifics of stressors and organisms examined.

An advancement beyond multiple indicator species is a more holistic combination of multiple species specifically incorporating many taxonomic groups. Such studies move a step closer to producing realistic measures of environmental health and have a potential to detect a broader spectrum of stressors than the more focused indicator species approaches. Data at this level are usually collected for index-based (including multi-metric methods) assessments of environmental integrity.

Index-based and **multi-metric methods** use biological assemblages rather than individual species. These include protocols such as indices of biological integrity (IBIs), floristic quality assessment indexes (FQAI), multi-metric approaches (essentially IBIs) as well as many developed for estuary and coastal wetland applications (Table 3). Because these are integrated approaches to measuring environmental health they are rarely concerned with, or

able to detect, specific causes and effects (Seegert 2000). They are generally used to track slow-moving changes associated with rehabilitation (Lopez & Fennessy 2002) or potential degradation of sites (Toham & Teugels 1998), or for comparisons of ecological health among sites (Lougheed & Chow-Fraser 2002). Despite having an assemblage level focus, changes detected using these approaches can occur at an individual species level. However it appears that this level of information is often missed because the indices are frequently focused at other conceptual levels, such as species richness (Kennard et al. 2006b). This is particularly so for indices developed for estuarine applications (see Whitfield & Elliott 2002 for descriptions of recently developed indices).

Another problem with IBIs is that complex ecological information is distilled to a single index value and similar values can be obtained for very different reasons. Consequently clear explanation for particular outcomes is missing (Maddock 1999, Kennard et al. 2006a). Furthermore, these approaches are sometimes applied using a single biological variable such as species richness, and/or limited to a single environmental descriptor, thereby providing an output with a high probability of error (Kennard et al. 2006b). In addition, different indices developed for the same system can produce vastly different assessments of system health (Seegert 2000).

Table 3: Index-based approaches that have been used for assessing estuary and coastal wetland health. Climatic regions the index was developed for are abbreviated as: **Temperate** = Tem, **Subtropical** = Sub, **Tropical** = Tro.

Application of the index	Region	Index	Source
Assess gradients of pollution	Tem	Immunotoxicological index	Auffret et al 2006
Assess system health	Tem	Benthic Index of Biotic Integrity	Carr et al 2000
Assess level of pollution stress	Tem	Shannon-Wiener Diversity Index (H') Maturity Index (MI)	Gyedu-Ababio 1999
Assess habitat quality	Tem	Estuarine index of biotic integrity	Meng et al 2002
Assess vegetation response to environmental flows	Tem	Normalized difference vegetation index Chlorophyll normalized difference index Structure independent pigment index Photochemical reflectance index	Thorhaug et al 2006
Assess water quality	Te/Sub	Diatom Index	Bate et al 2004
Assess estuarine health	Te/Sub	Estuarine Fish Community Index	Harrison & Whitfield 2004
Assess environmental flow requirements in estuaries	Te/Sub	Estuarine Community Degradation Index Estuarine biological health index Estuarine biotic integrity index Estuarine fish recruitment index South African Scoring System Estuarine Health Index	Ramm 1988 Cooper et al 1994 Deegan et al 1997 Quinn et al 1999 Vos et al 2000 Adams et al 2002
Assess water pollution using diatoms	Sub	Generic index	Wu & Kow 2002
Assess impact of crude oil pollution and natural remediation	Tro	Biological Index of Pollution	Essien & Antai 2005

iii. Assemblage Level Approaches

Assemblage and/or community level methods represent a step beyond index-based methods. Assemblage/community level methods are usually considered to provide a more holistic assessment of ecosystem health than those discussed above. Application of multivariate techniques to assemblage/community level data allows information such as assemblage/community composition to be examined and compared over time and, provided appropriate variables have been measured, offer a robust method of linking community response to potential stressors.

A **community** is a set of interacting species in a system. Collecting data on an entire community is very time consuming and expensive, and may require the use of multiple sampling techniques to provide representative data for all taxa (Blaber et al. 1989). **Assemblages** are a suite of species able to be collected from a system using a particular method, and assemblages are more often the target in ecosystem health studies. Collection of the logistically more achievable assemblage-level data, concentrating on well known and easily sampled groups (e.g. fish), allows evaluation of ecosystem condition from two information rich perspectives: taxonomic groups and functional groups. Consequently, assemblage level approaches provide more reliable proxies for ecosystem level interpretations than less data intensive methods such as indicator species (Mahaney et al. 2004).

The use of **taxonomic groups** allows data to be used at species, generic and/or family levels (Warwick 1993). When used in conjunction with appropriate multivariate analyses, information about assemblage dynamics (assemblage composition [relative abundance], size distributions) can be extracted easily and matched with environmental variables to provide robust information about responses to stressors. This provides much more extensive and versatile information than simple aggregate measures like species richness or diversity.

Functional groups represent groups of organisms that fulfil the same ecological function at a given point in time (Wilson & Sheaves 2001). Functional groups can include trophic groups, both in the traditional sense or as specific aggregations of organisms (such as the prey of a particular predator), reproductive groups and life-history groups (Kennard et al. 2006).

iv. Process Level Approaches

The use of taxonomic groups, and to a greater extent functional groups, provides powerful methods of assessing ecological processes (the attributes that are most closely aligned to ecosystem integrity), because it is at the functional group level that organisms participate in ecological processes. Assemblage level data can provide most of the information required to examine important ecological processes such as recruitment success, food web structure, nursery ground value, predation and scavenging pressure, habitat utilisation and habitat integrity. The ability to examine process level information is an important step in determining fundamental changes to the way ecosystems work (Mahaney et al. 2004). For example, changes in flows to estuaries can alter nutrient delivery and primary and secondary productivity, thereby changing the way food webs function. Such changes can be detected by shifts in trophic composition of fish assemblages (Whitfield 1996, Whitfield & Elliott 2002). Despite enormous promise, direct process-based approaches have rarely been used for assessing ecological health of estuarine systems (Elliott 2002), although assemblage and community level data have been widely used to measure ecological change. In studies that have used assemblage/community level data the focus has rarely been on process. Rather it has usually been less specific and broadly aimed at “detecting ecological change”, usually changes in biodiversity and/or abundances of particular organisms, at some specified (but sometimes unspecified) and usually localised (sub-system) level.

The collection of process level data may increase logistic and analytical difficulties compared to less data-intensive approaches, but it also presents a far greater opportunity to detect anomalies in ecosystem function. Although collecting more complex data has often been seen as detrimental because of increased analytical difficulties, that is not necessarily the case. Rather, complex data lend themselves to analysis with multivariate statistics that can facilitate outcomes that provide far greater explanatory power than is possible with simple data and univariate techniques (Boulton 1999, Harris & Silveira 1999, Kennard et al. 2006a). The ability to simultaneously examine multiple environmental descriptors and a range of biological variables allows for rapid identification of factors possessing sufficient variability to sensitively reflect biological responses. Descriptors that vary little provide little useful information that can be related to changes in ecological function. The usefulness of multivariate approaches, as with all approaches, hinges strongly on selection of appropriate variables in the first instance (Karr 1999).

Process-based approaches have a great capacity to provide ecosystem level information. However, to inform process level understanding, integration of several approaches may be more informative than any of the approaches individually. For example, process level data can detect changes in the way a system functions, but process level understanding does not provide direct information about sub-lethal physiological effects of stressors, such as reduced fecundity. Therefore the use of appropriate biochemical approaches on selected species, in conjunction with process-based approaches, would facilitate a more specific interpretation of ecosystem integrity by revealing the underlying mechanisms of process-level change.

The combinations of complex fauna and sparse background understanding means we are a long way from completely understanding most aspects of tropical estuarine ecology. At present, therefore, it is unrealistic to expect to be able to determine ecosystem health at a whole-of-estuary level, or gain an overview of generic "estuary health". We should instead focus on understanding the particular impacts of specific stressors. Even then there needs to be a trade-off between desired outcomes and logistic considerations. It is not possible to measure every possible parameter in tropical estuaries (Carignan & Villard 2002), so decisions about which parameters are measured become crucial to achieving desired outcomes (Karr 1999). However, to reliably represent ecological integrity, studies will have to both acknowledge and embrace the complexity of tropical estuarine ecosystems (Elliott 2002). This does not require that every possible parameter is measured. Rather, what is required is a focussed approach that first identifies a specific factor of concern, then determines the key ecological process(es) that are most likely to be affected, and finally identifies the most useful measure of the status of that process.

Specific Applications

Studies focused on specific stressors have been useful for directly assessing impacts of those stressors, however extrapolation of results to broader ecological concepts has been difficult. The greatest advancements towards more holistic assessments of ecosystem integrity are emerging from work associated with environmental flows. This is the one area where the overall complexity involved in the impacts of altered flows has been recognised and embraced. In contrast, advancement in the field of contaminants has been mainly confined to technical components and recognition of “new” contaminants in the environment. Although our knowledge of lethal/sub-lethal effects of contaminants has also advanced considerably, the broader ecological outcomes of contaminants in estuaries are not well understood except in extreme circumstances.

a. Contaminants: Toxicants, Nutrients

The nature of any impact of contaminants is closely linked to the type of pollutant, the manner in which it is delivered (rapid, chronic), its dispersal, degradation and elimination rates, bioavailability and rate of uptake, biological implications of uptake, physico-chemical nature of the wetland, and wetland connectivity (Petty et al. 1998, Das 2000, Magni & Montani 2000, Kolpin 2002, Petty et al. 2004, Bainbridge et al. 2006). Our ability to detect biological responses to contaminants will depend on knowledge of all these factors. For example, a rapid, short-term pulse of a pollutant in a well flushed system may have a relatively low impact signature across a broad spatial scale, and thus would be more difficult to detect than chronic pollution. Identifying a point source may also be difficult unless the pollutant is endemic to a single potential source.

To assess ecological integrity of a system it will be necessary to gain an improved understanding of how contaminants interact with each other and with the broader environment. For example, bioavailability is often assessed using bacteria (Davies et al. 1998) which are used as a proxy for other organisms in the environment, but bioavailability can also be determined by chemical processes without a clearly demonstrated link to measurable biological outcomes. Uptake of contaminants can be measured but may require a history of other environmental variables before a clear interpretation can be delivered. For example, the rate of mercury uptake in prawns is strongly influenced by salinity (Das 2000), so measuring mercury levels alone only confirms the presence of mercury. The salinity history (and probably movement patterns) for the prawn is required to infer broader ecological implications. There is a considerable amount of literature examining the implications of exposure to different contaminants for specific species. Documented effects include lethal and sub-lethal outcomes that include reduced reproductive potential, exclusion from habitats and minor impairment, but these studies generally lack the data to relate their findings to ecological outcomes. Connectivity, particularly flushing potential and movements of contaminants, is another area that has received little attention in estuaries. Physical dispersal of contaminants can and has been modelled for many systems (e.g. Baumann & Niessner 2006, Carafa et al. 2006, Palancar et al. 2006) but there are few data linking those dispersal patterns to ecological outcomes. For tropical estuaries in particular, the retention times for contaminants may be significantly modified by the presence of mangrove forests, due to lateral trapping at forest margins (Wolanski et al. 1990, Victor et al. 2004). Residence times for contaminants in dry tropics estuaries may be further extended by reduction of tidal water exchange as the salinity pattern becomes inverse during dry seasons (Wolanski et al. 1992, Ridd & Stieglitz 2002).

i. Past approaches to detection

Past approaches to detection of contaminants were primarily based on water/sediment quality measurements, including the use of indicator species as bioaccumulators and/or avoiders. Detection was generally approached from a contaminant-specific/species-specific basis, with thresholds for biological response (e.g. uptake rate, toxicity) validated experimentally (e.g. Everitt et al. 2002, Clavier et al. 2005), although bioavailability of contaminants was often assumed from chemical rather than biological experimentation (e.g. Verrengia Guerrero et al. 2007). This practice should only be applied subsequent to validation to provide a clear understanding of the rates of uptake among the species involved (Simpson et al. 2006). Often, data were restricted to simple water quality measures which were used as a proxy for estuary health (Cox et al. 2005). In some instances indicator species were used as a proxy for water quality and subsequently estuarine health, but generally when indicator species were used, data were used in conjunction with water quality parameters. In recent years, passive samplers have begun to replace indicator species in the detection of contaminants (Petty et al. 2004). These have the advantage that they can be placed anywhere they are required and remove the reliance on indicator species being present when and where they are needed. Moreover, passive samplers facilitate relatively rapid detection of a broad spectrum of contaminants.

Water quality parameters are often used to infer biological responses using correlative measures (e.g. fish and turbidity Cyrus & Blaber 1987, 1992). However samples of both water parameters and fish are often point samples in location and time and therefore may not reliably represent the long-term physical regime in an estuary. Many organisms that use estuaries are capable of tolerating extreme and rapid shifts in physical parameters such as salinity and turbidity in the short term (Craig & Crowder 2000, Morgado et al. 2007), and their distributions are therefore more likely to be influenced by long-term patterns in physical parameters (Davis 1988, Sheaves 2006).

ii. Alternative approaches

Suites of passive samplers (e.g. semipermeable membrane devices [SPMD], polar organic chemical integrative sampler [POCIS], passive integrative mercury sampler [PIMS](Petty et al. 2004)) provide useful means of quantifying a broad range of pollutant levels/water quality. One big advantage of passive samplers is that they can provide valuable information about which stressors vary in the environment in a way that can be related to biological variability. An array of passive samplers could be used in conjunction with biological samples (taxonomic and functional groups to elucidate process-level information) to assess biological responses to a broad range of contaminants, but because we are almost always dealing with a complex mix of contaminants (Petty et al. 2004), assigning direct cause/species-specific effects would be difficult. However, this approach at least provides information that can be used to eliminate sources of stress.

Current research (Humphries et al. in review) suggests that biomarkers in fish are potentially useful in tropical estuaries. Humphries et al. detected the presence of organophosphate insecticides from changes in cholinesterase activity in muscle tissue of barramundi (*Lates calcarifer*), an important fisheries species. Biomarker studies represent a successful and relatively straightforward approach to detecting compounds such as organophosphate insecticides that are difficult to detect in the environment using other techniques. The challenge for biomarkers studies is to understand their place in broader ecosystem health assessments.

iii. Problems

There is a broad range of contaminants of concern, from those introduced historically (organochlorine pesticides, polychlorinated biphenyls, polychlorinated dioxins and furans, polycyclic aromatic hydrocarbons, heavy metals, nutrients) to more recent concerns such as pharmaceutical chemicals and personal care products (Petty et al. 1995, Petty et al. 1998, Kolpin et al. 2002, Petty et al. 2004). One immediate problem is simply what do we monitor? A further complication is that contaminants don't occur in isolation but generally represent a complex mixture in the environment (Petty et al. 2004).

Most current detection approaches are unable to provide a holistic exposure assessment because:

- 1) they lack the capability to integrate samples through time (sampling represents one or more points in time),
- 2) analytical sensitivity and selectivity are insufficient to detect and quantify ultra-trace to trace levels of complex mixtures of contaminants in water,
- 3) site conditions (e.g. water quality) impact the survival of sentinel organisms and exposure information is lost due to species-specific metabolism and depuration of contaminants of interest,
- 4) causal links between observed biological effects and contaminant mixtures are seldom established (Petty et al. 2004), and
- 5) it is simply too difficult and too expensive to monitor every possible contaminant.

b. Environmental flows

Direct consequences of alterations to flows can be pervasive across many spatial and temporal scales and have been reported for most biological components of rivers and estuaries. For example, the impacts on estuarine (and freshwater) fish faunas include disruptions to life-history migration pathways which are often crucial to successful reproduction (Connor & Pflug 2004, Walsh et al. 2005), growth and survival (Borges Barthem et al. 1991, McCormick et al. 1998, Cooke & Learch 2004), alterations to community structure (Kennish 2001, Gillette et al. 2005, Sheaves et al. in press b) and faunal distribution (Lafaille et al. 2001, Daufresne et al. 2005, Sheaves et al. 2007). Further, they impair ecological processes such as spawning (Connor & Pflug 2004, Walsh et al. 2005), influence habitat provision (Schofield 2003, Grismer et al. 2004, Khan et al. 2004) and habitat structure (Kennish 2001), disrupt the supply of nutrients that stimulate primary (Cloern et al. 1983, Salen Picard et al. 2002, Mallin et al. 1993) and secondary productivity (Kalke & Montagna 1991, Livingston et al. 1997), decrease juvenile survival (Salen Picard et al. 2002, Kraus & Secor 2005, North et al. 2005), and prevent the dissemination of cues needed for successful recruitment of juveniles (Strydom & Whitfield 2000). In turn, flow-on effects can impact indirectly on higher trophic levels (Omundson et al. 2002), change predator/prey dynamics and modify energy flow (Breitburg et al. 1997), both of which can feedback to impact community structure. Extended low- or no-flow periods can lead to complete changes in the way a system functions (Livingston 1997, Livingston et al. 1997); changes that may be irreversible in extreme cases (Bate & Adams 2000).

Studies of the effects of river flow on tropical estuarine species initially focussed on prawns, both in Australia (Vance et al. 1998) and overseas (Ahmad Adnan et al. 2005). Only two projects have been directed at understanding freshwater flows and tropical estuarine fish in Australia, one at the southern margin of Queensland's dry tropics (Staunton-Smith et al. 2004, Robins et al. 2006, Sheaves et al. 2006) and one in the central dry tropics (Sheaves et al. in press b). However, there is a large body of literature that examines biological response

to changes in parameters, such as salinity, that vary in response to changes in freshwater inflow.

Reduced flow in estuaries can induce both higher long-term salinity and increased inland incursion of saline water (Whitfield & Harrison 2003). This leads to changes in habitat availability and shifts towards more marine faunas in affected areas (Sheaves 1992, Whitfield 2005, Sheaves et al in press b). In contrast, increased flow can lead to reductions in salinity and promote mixed marine/freshwater faunas (Bate et al. 2002, Sheaves et al. in review) however long-term maintenance of those faunas would require input of consistent freshwater baseflow (Bate & Adams 2000). The effects of altered environmental flows on biological processes have been reported at many organisational levels, including individual species (Adams & Bate 1995, Staunton-Smith et al. 2004, Robins et al. 2006), assemblage (Albaret et al. 2004, Teels et al. 2004, Gyedu-Ababio & Baird 2006) and functional (Livingston et al. 1997). Although detectable at various levels, effects of altered flows are pervasive and impact on every aspect of estuarine ecology.

i. Past approaches to detection

The main methods of assessment discussed above have all been applied to the issue of environmental flows in freshwater systems. In recent times, reduced environmental flow has emerged as one of the most critical problems facing rivers and estuaries. As a result, there has been significant development of assessment approaches in freshwater systems ranging from those that principally consider hydrological characteristics (Hydrological Index, e.g. Richter et al. 1997, Hydraulic Rating e.g. Gippel & Stewardson 1996) through to those that merge physical and biological components at many organisational levels. The latter methods include the "Expert Panel Method" in which a team of experts arrive at appropriate flow regimes to meet the needs of biota, "Prescriptive Holistic Method" (e.g. Arthington et al. 2004) which attempts to build structured links between biota and their flow requirements and "Flow Stressor Response Models" (e.g. O'Keefe et al. 2001, Palmer et al. 2005) which extend the prescriptive holistic methods to include explicit risk assessment.

ii. Alternative approaches

In general, the methods applied to this issue have developed towards more process-based outcomes, albeit by proxy. Although data collected may provide the capacity to be used to examine processes, process itself is rarely examined. The objectives seem to be focused on taxonomic composition and abundance rather than specifically on function or process. Examination of ecological processes is a more reliable way of identifying shifts in the way a system functions than existing approaches. In fact, focussing on taxonomic/functional groups and examining process can be seen as simply an extension of some existing index-based methods. However, in selecting functional groups that relate to specific processes, methods that integrate functional groups and processes provide more relevant information but logistically simpler data collection than most index-based methods. This is because the index-based methods that purport to be holistic because they measure a broad range of taxa, do so at the expense of obtaining representative data across any one taxonomic group. Despite excellent work on these aspects in freshwater systems, little has been done on estuaries.

c. Habitat modification

Habitat modification occurs in response to both altered flows and the input of contaminants into estuaries. Estuaries are also affected by other anthropogenic factors, which can be classified into three broad categories:

- 1) complete habitat loss (physical removal and/or change to habitat characteristics [e.g. land reclamation, marina developments, aquaculture ponds] and removal of access for organisms [e.g. barriers to movement]),
- 2) partial habitat loss (reduced extent of habitat, disruptions to habitat contiguity),
- 3) habitat restoration.

All habitat modifications inevitably cause biological changes that can range from reduced abundances of one or more species in a system to complete ecosystem collapse. Habitat restoration is being used increasingly in temperate estuaries in Australia and elsewhere. However, while habitat modifications associated with restoration certainly cause a shift in community composition, the mix of species that returns after restoration is often not the same as that in place prior to the original habitat degradation (Smallwood 2001, Teels et al. 2004).

i. Past approaches to detection

Past attempts to quantify the effects of habitat degradation in estuaries have generally been based on changes in the aerial extent of broad habitat types (e.g. mangrove forest), and related to either use of the habitat type by particular organisms (rarely done, but see Manson et al. 2005) or in the case of commercial organisms, to commercial catch data (e.g. fisheries catch statistics, e.g. Lee 2004, Loneragan et al. 2005). Possibly because of the perceived difficulties in obtaining directly comparable data from the diverse range of habitats in tropical estuaries (Blaber et al. 1989), there have been few studies of habitat preferences of estuarine fauna (Johnston & Sheaves in review).

ii. Alternative approaches

As with detection of environmental change due to contaminants and altered flows, a shift of focus to process-based approaches seems the most promising pathway to gaining an improved understanding of habitat-related ecosystem function in tropical estuaries. Just as changes to flows can be detected in changes to the trophic structure of fish assemblages, with appropriately designed sampling protocols, based on taxonomic/functional groups, information on the effects of habitat modification could be obtained from the same data required to assess food web integrity.

d. Overexploitation of resources

There is considerable concern about the level of exploitation of resources provided by tropical estuaries (Ong 1995). However, beyond issues of environmental flows, contaminants and habitat modification, there has been little research in this field in tropical Australia. This is despite the fact that tropical estuaries are important nursery areas for many species of fish and crustaceans (Robertson & Duke 1990). Given that estuaries do provide crucial nursery habitats, it is likely that there is a direct link between whole-of-estuary function and the provision of key habitat resources at appropriate times. Questions of scale therefore become particularly relevant, because of relationships between different species/life history phases and species/phase-dependent and use-dependent (food, refuge) habitat use.

i. Past approaches to detection

Past approaches to detection have generally been driven by commercial interests, mainly fisheries, but quantifying the extent of change has proved difficult. In part this relates to the limited amount of research money committed to this issue. The lack of definitive information has contributed to low levels of management success for estuarine fisheries. The inability to produce clear outcomes, and subsequent failure of management initiatives, emphasise the difficulties associated with attempts to estimate abundances of fish, particularly in open

systems (Allen et al. 1992), and highlights the problems associated with reliance on correlative relationships between historical and current fisheries data and mangrove area/fisheries production.

ii. Alternative approaches

For this issue in particular there is a clear need to move towards assemblage level and particularly process-based methods. For example, key processes such as spawning and recruitment success, nursery ground provision and food web integrity could be examined at scales appropriate to fisheries management requirements.

e. Problems common to all issues

Problems common to all issues include the transferability of data and scale. Of particular relevance for tropical estuaries is the extrapolation of conclusions from temperate systems. One problem is that biochemical rates not only differ across climatic regions but differ in response to changes among more dynamic variables (e.g. rate of mercury uptake in prawns influenced by salinity [Das 2000]). This makes biological responses difficult to quantify even before potential interactions between pollutant and chemical environment are considered, and renders the transferability of information from temperate systems problematic (Biney et al. 1987).

Tropical estuaries tend to have location-specific faunal compositions (Sheaves 2006). In addition, seasonal factors need to be considered in tropical estuaries because tropical estuarine fish faunas have higher levels of compositional similarity during summer recruitment periods than during the cooler months (Sheaves 2006). This means that sampling protocols need to take account of among-estuary differences that may be substantially influenced by temporal factors. Similarly, when pre-impact data are not available, it may be difficult to quantify reference states, a problem that imposes inferential limitations on outcomes (Stewardson et al. 2004). However, despite differences in taxonomic compositions, there are strong similarities among estuaries at a functional level (Fig. 3) (Sheaves 2006), and presumably at a process level. Functional, process level approaches therefore overcome the problems associated with more conventional methods based purely on taxonomic comparisons.

A key problem in assessing the impacts of alterations to environmental flow regimes is that such events affect biological outcomes at the whole-of-estuary scale. Because it is important to study stressors at scales appropriate to the problem, this suggests that a whole-of-estuary scale is appropriate for investigating the effect of altered flows (Maddock 1999). Addressing environmental flows at such a large and encompassing scale is both logistically and biologically complex. Recognition of the complexity of this problem has clearly driven the shift towards more intensive data collection in recent years (Elliott 2002). Despite this, useful ecosystem-level information can be obtained from smaller-scale studies, particularly when sampling scales are matched to the biological problem at hand. In determining appropriate scales for studies, it is essential to understand the type and extent of the stressor, and to select variables appropriate to detection of the particular biological responses. For example, it is important to select biological variables for habitat remediation studies that relate to different levels of habitat sensitivity and potential recovery times (Maddock 1999).

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For the issues of habitat modification and overexploitation there are multi-level interactions (direct and indirect) between habitat use, nursery function, spawning and recruitment to fisheries. Because most research has had a direct fisheries focus, we have little understanding of most of those interactions for estuaries in general and particularly for tropical estuaries.

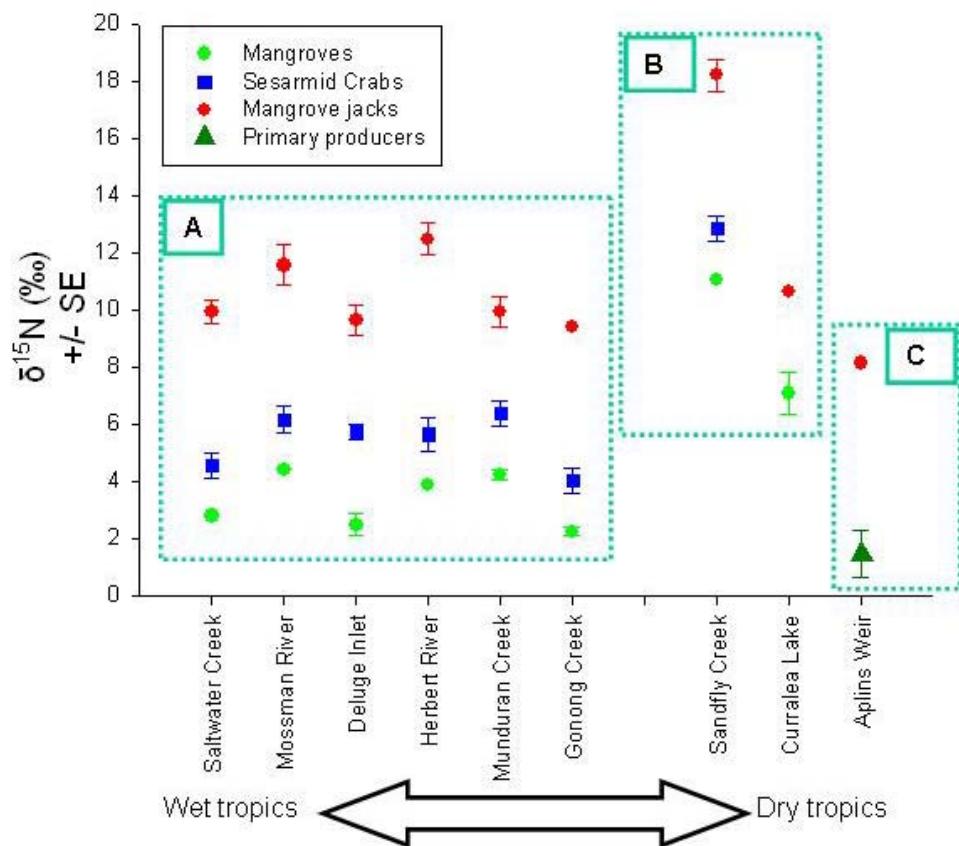


Figure 3: Relationships between mangroves (primary producers), sesarmid crabs (primary consumers) and mangrove jack (fish predator) for transfer of stable nitrogen through a food web. At each trophic step the fractionation of nitrogen remains consistent from mangroves to crabs and from crabs to fish for wet and dry tropics estuaries (boxes A and B). Estuaries and wetlands in this figure range from near pristine to modified however the values for nitrogen signatures vary substantially between systems in box A and boxes B and C. The high nitrogen values in box B, Sandfly Creek and Curralea Lake, are clear indications that processes are functioning differently from other systems. These are the only systems represented here with an adjacent sewage outfall or pumping station. Box C, Aplins Weir, has a generally low nitrogen signature for primary producers, and correspondingly low signature for mangrove jack. It is likely that this reflects low but persistent inputs of fertiliser from adjacent residential areas (M. Sheaves, K. Abrantes unpub. data).

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Another issue potential confounding impact assessment is the timing of sampling. At its most basic, it is necessary to consider tidal phase, particularly in respect to access to particular habitats. Moreover, the timing of repeated sampling for mobile organisms in tropical estuaries should remain consistent relative to tidal phase (lunar [spring/neap] and daily [high/low]) because many of those animals use different habitats during different parts of the tide (Connolly 1999, Thomas & Connolly 2001). Of greater concern is the timing of sampling for detection of the stressor. That timing is critical for contaminants delivered to estuaries by overland and downstream flows. Such contaminants are generally delivered by the first seasonal flows, which are often minor and localized in extent. Consequently, there is often also insufficient through-flow to flush the contaminants from the estuary, and they therefore persist in the estuary for longer periods (Bainbridge et al. 2006). Biological sampling protocols should not only be designed to collect pre-impact data but should be flexible enough to accommodate response sampling that commences when contaminants are detected (or expected), and be supplemented by ongoing data collections at regular intervals to examine the patterns and rates of recovery.

f. Risk assessment

Risk assessment is the crucial final step in determining the importance and degree of impact of environmental stressors (Palmer et al. 2005). In this context, risk is the probability of adverse effects on the environment resulting from exposure to a particular stressor, so risk assessment is the process of correlating environmental outcomes and stressors likely to cause those outcomes. Risk can be evaluated prospectively, by evaluating the likely impacts of predicted levels of known or suspected stressors, or retrospectively, by comparing specific impacts to known stressors. Prospective risk assessment provides the opportunity to impose management measures that will prevent environmental harm. Unfortunately, it relies on detailed prior understanding of the effects of specific levels of stressors and the associated uncertainty, information that is often unavailable in either a physical (eg. DanangPC & PEMSEA 2004) or an ecological context (eg. Berge et al. 2006). In contrast, retrospective risk assessment seeks to establish evidence of impacts based on causal linkages between stressors and observed ecological outcomes; in many cases probably a more achievable goal. Even here, however, a lack of appropriate information is often a major impediment to unambiguous evaluation. For instance, Barton & Metzeling (2004) found that a lack of information on ecological conditions in southeastern Australian streams prior to European settlement hampered their ability to set biological objectives for stream health. While establishing the link between causes and primary effects is difficult enough, causal links to secondary effects are even more difficult, further complicating risk assessment.

The Way Forward

a. The problem

Determining ecosystem health is clearly a complex and difficult task. While gross-scale environmental degradation is often obvious and relatively easy to detect, the development of reliable techniques to provide early warning of decline or to allow the detection of subtle or pervasive, low-level effects has often proved elusive. Given these difficulties, and the many ways in which tropical conditions are likely to complicate detection, simply applying methods that have been used in other environments would be a mistake. Clearly, understanding the range of options available and the problems that detection methods have encountered in the past are the first steps to successful detection of impacts. This review thus provides a sound basis on which to develop approaches that are specific to tropical estuarine conditions. However, even with extensive background understanding the task will not be simple, because the challenge is to develop approaches that are situation-specific, robust, easy to interpret and relevant, in an environment where limited biological and ecological understanding are major impediments.

To move forward we need to ensure that we have a clear understanding of the task and of potential problems, that we set clear and ecologically meaningful biological objectives, and that we have a clearly enunciated plan that focuses on gaining the types of knowledge necessary to advance our understanding. Consequently, in developing techniques for determining the health and integrity of estuaries and coastal wetlands in Australia's tropical regions, we need to not only develop methods of detection but also to advance our understanding of the links between stressors and environmental outcomes.

b. Where we stand now

We need to step beyond the use of water quality as a proxy of ecosystem health because, by itself, water quality provides limited information about biological function. With few exceptions, links between water quality, estuarine organisms and ecosystem functions are poorly understood. What are needed are approaches that are directly relevant to, and reflective of, biological outcomes.

Single species and single variable approaches can provide detailed information about specific issues. While they usually do not reflect ecosystem health as a whole, the specific information they provide can form important components of a holistic evaluation. An excellent example of a tight link between a specific water quality parameter and a biotic component of the ecosystem is provided by the study of seagrass biology and water clarity in the subtropical waters of Moreton Bay, southeast Queensland. Seagrass meadows declined when turbidity reduced secchi disk depths to < 1.7 m (Abal & Dennison 1996). A management objective was therefore set to keep annual turbidity in the bay below this level (as a median of monthly measurements). Even in this example, the link with overall ecosystem health was not fully determined, since the role of seagrass in ecosystem integrity was only assessed speculatively. More complex multispecies and multivariable methods provide more information, but again seem most useful as components of a broader understanding. It is really techniques that measure functional aspects of ecosystems that hold the most promise because these relate directly to ecological function, the conceptual entities that are linked most closely to ecosystem health and integrity. Although it seems that addressing functional questions should require a substantial increase in the complexity of data that need to be collected, this is not necessarily the case. Most of the information required is readily available from the data normally collected in single or multiple variables studies, or easily obtainable by standard ecological methods. The one additional factor that is

required to address functional questions is an advanced conceptual understanding of ecosystem functioning. This is a substantial problem in tropical estuarine studies where basic ecological understanding is sparse. However, if any real advances are to be made, these will come through improved abilities to understand and measure the status of ecosystem functioning rather than from measuring proxies with unknown relationships to the processes that impact integrity.

c. The future for tropical estuarine indicators

Although the way forward in the development of measures of tropical estuarine health and integrity is likely to be far from simple, there seems to be a straightforward course of action at a strategic level. This involves a logical sequence of study (Fig. 4).

- 1) Firstly, a particular stressor of concern needs to be identified (e.g. a specific pollutant) and its levels in the estuary measured at specific spatial and temporal scales. Here, direct single variable measures are likely to be appropriate in most situations. In the case of the specific pollutant, these would be physical measures of the pollutant in the water column and/or sediments or, where measurable levels are likely to be transitory but effects long-lasting, they may take the form of measurements of bioaccumulation in suitable organisms.
- 2) Secondly, a conceptual model of the way in which the stressor is likely to affect the ecosystem and the key processes likely to be impacted should be developed, based on the best available ecological understanding.
- 3) Thirdly, any changes in the key processes need to be identified and quantified. This could take many forms, but optimally would involve studies of the process at the putative impact site and a range of control sites, before, during and after the impact (essentially a BACI design (Green 1979, Underwood 1994)). If possible, any impacts should be firmly linked back to the stressor. Up to this stage the tacit focus has been on an impact by impact assessment. However, this third part of the process provides the basis for the development of a more holistic understanding and broad-based assessment of estuarine health, as the cumulative understanding from studies of the effects of different stressors builds up to contribute to a more complete picture, with the ultimate aim of producing a whole-of-estuary evaluation.
- 4) Finally, once the exact nature of impacts on key processes has been quantified, a range of simple measures or indices that are likely to unambiguously reflect the specific change in the process should be identified. These should be “traded-off” against complete assessment of the process to determine if, and to what extent, any of these measures can stand as simple but effective proxies.

This is easily enough articulated, but in practice will be complicated by interactions with other stressors and with environmental and biological variables, and by the complexities of time, space and scale.

This sequence specifically links indicators to ecosystem level outcomes of known impacts, and emphasises the pivotal role of process-level ecological understanding. Assessments of ecosystem health, the accurate interpretation of the meanings of changes in indicators, and eventual management decisions are only as good as the theoretical basis on which they are constructed.

For this approach to be translated into successful management, the frequent mismatch between the less detailed, ecosystem-level requirements of managers and the more focussed and very detailed aspirations of researchers needs to be addressed (Elliott 2002).

In particular, it is important for all protagonists to have a clear, common understanding of specific objectives and the conceptual scales at which they are to be applied. Central concepts such as “estuary health” need to be clearly defined to ensure clarity of direction for all parties involved, and to ensure that the tools developed by researchers are appropriate measures of the qualities that the community values. Concepts like estuarine health have many valid interpretations, and consequently there are likely to be many different research outcomes based on each researcher’s particular interpretations. This was well illustrated when three different groups of researchers independently developed Indices of Biologic Integrity for assessing river health for the same river. Each index was based on a particular interpretation of what was considered to reliably represent river health. Consequently, the indices produced contrasting outcomes from the same data; differences large enough to place the river in different health categories (Seegert 2000). To overcome such issues managers and researchers need to jointly develop specific questions that fit within specific frameworks; the first step is for managers to articulate their requirements in clearly defined ways so that researchers can tailor protocols to address specific management issues. It is then incumbent on researchers to address those issues at scales appropriate to management requirements.

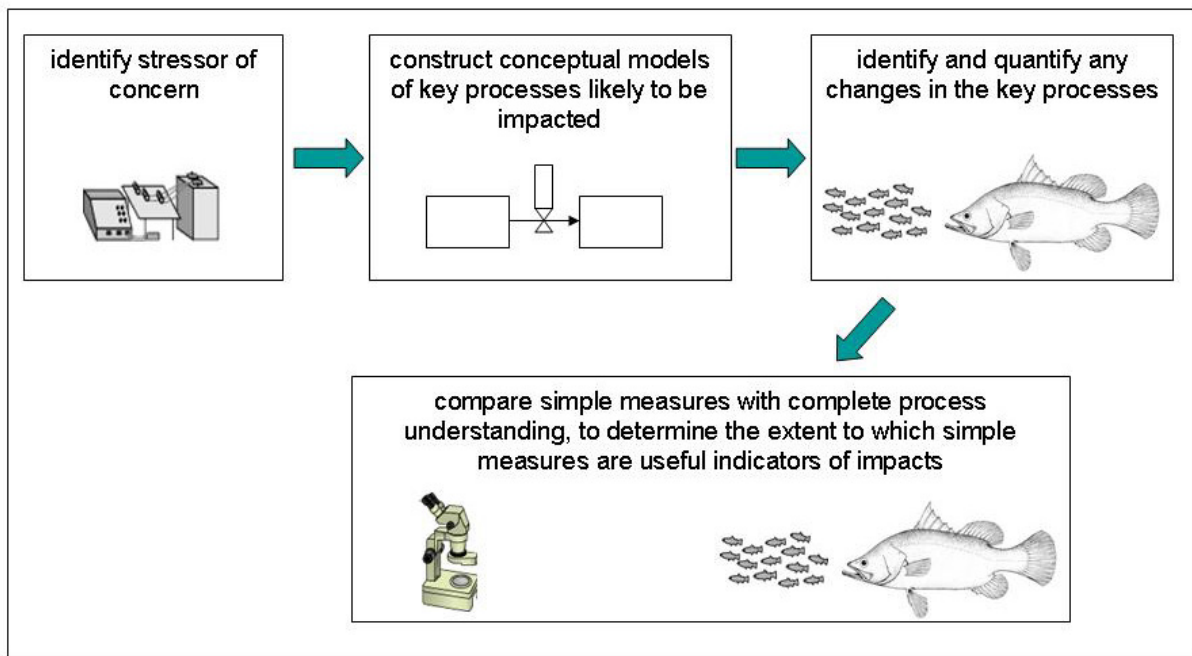


Figure 4: The series of steps in developing measures of ecosystem health and integrity for tropical estuaries.

d. Potentially useful indicators

Estuarine indicator work in the MTSRF project will focus on fish. We will be guided by previous work using fish as indicators of estuarine health in South Africa (Harrison & Whitfield 2004), and work in southern Queensland in freshwater systems (Kennard et al. 2006b). These previous efforts have mainly used measures of organisational structure as indicators of ecosystem health, especially indices that summarise the differences between fish species expected to be present against those actually sampled. We intend to extend work on fish as indicators beyond this to include ecological processes that, together, should offer a more comprehensive indication of ecosystem health. We have conceptualised three estuarine fish indicators likely to represent important components of ecosystem health: scavenging pressure, non-detritivore trophic composition, and predator success (Fig. 5).

One aspect deserving of attention in tropical estuaries is scavenging pressure. Scavengers perform the essential function of consuming carrion that would otherwise build up and generate toxic effects of decomposing bacteria. Scavenging also increases the rapidity with which nutrients in carrion are recycled to the environment. Scavenging is thus a high level ecological function, and one that in theory will be linked with water quality in estuaries. For example, estuaries with poorer water quality would have lower overall productivity and thus low carrion availability and high scavenging pressure. Alternatively, healthier ecosystems in estuaries with better water quality might have a greater abundance of scavengers and therefore higher scavenging pressure. A researcher at Griffith University (Webley unpubl. data) has developed a measure of scavenging pressure – the weight loss of replicate baits deployed for 30 minutes on the estuary floor. He has already shown the effects of tidal stage, time of day, soak time, bait size and type, and distance from estuary mouth, and his measure is now developed to the point of being suitable for measuring scavenging pressure quickly and efficiently across tropical estuaries. This measure of scavenging pressure is independent of the types of scavengers present. Furthermore, cameras are deployed to ascertain which species are responsible for scavenging, adding another component to the ecological interest in results.

Another component likely to prove useful in tropical estuaries is the non-detritivore trophic composition of fish assemblages (Fig. 5). Detritivorous fish dominate catches in all tropical estuaries, and potentially obscure effects of water quality. We therefore propose to examine the proportion of fish in functional groups other than detritivores (e.g. piscivores, microplanktivores, benthivores) as a measure of organisational structure in fish assemblages.

Finally, we propose to measure predator success (Fig. 5), one of the few aspects of fish ecology properly studied in tropical estuaries. The importance of river flow and connectivity between the main channel of an estuary and its associated wetland pools is well known (Sheaves et al. 2006, Sheaves et al. 2007). Predator success is part of an important ecological process, and this measure of ecosystem health will benefit from recent work demonstrating how best to measure it in tropical estuaries.

The study will focus on estuary systems of particular concern in the wet tropics of Queensland. Particular foci will be the Tully and Murray Rivers, systems with extensive agricultural development in their catchments. These rivers are recognised as heavily impacted by agrochemical pollution. Both systems are current study locations for CSIRO Land and Water, and are systems where hydrology, water quality and sediment transport are well understood (e.g. Mitchell & Furnas 2001). Consequently, focusing on these systems will allow links with CSIRO Land and Water and value-add to build more extensive outcomes from both projects. An adjacent, less impacted system, the Hull River, with substantially less development in the catchment, will allow comparison with a less impacted system nearby.

Systems such as Deluge Inlet and Zoe Creek on Hinchinbrook Island, with no development in their catchments, would provide the nearest examples of near pristine systems against which the health of the other systems could be judged.

Together the combination of a range of sites in close proximity, with contrasting levels of impact, and the variety of innovative techniques now available, provide an opportunity to greatly advance our ability to assess the health and integrity of estuaries and coastal wetlands of Australia's tropics.

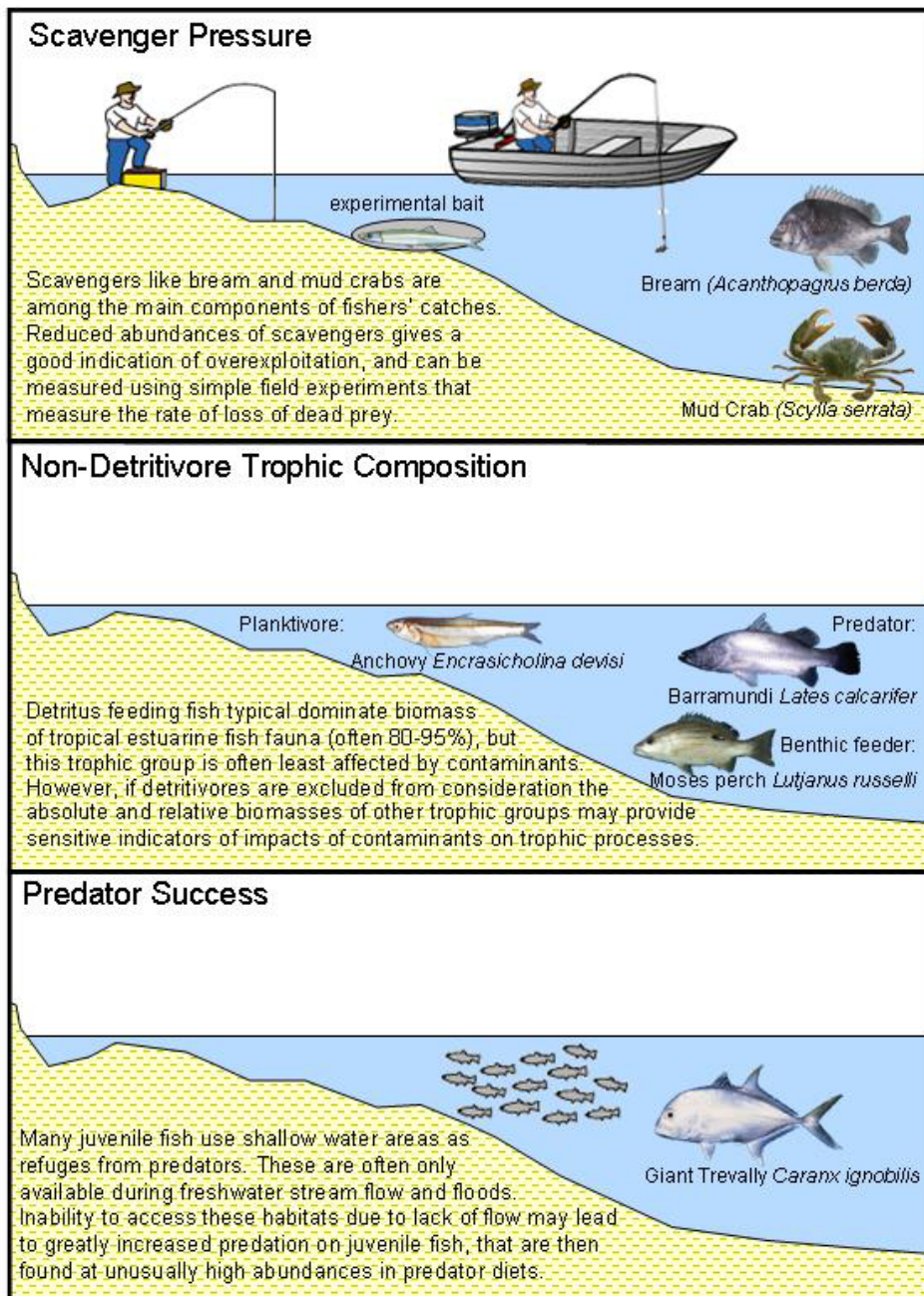


Figure 5: Three examples of simple biological indices with the potential to provide information on impacts to crucial ecological process in tropical estuaries.

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