AN INTEGRATED MODELLING APPROACH TO THE MANAGEMENT OF FRESHWATER INFLOW TO SOUTH AFRICAN ESTUARIES

by

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ABSTRACT

Estuaries are recognised for their biological diversity and productivity, as well as the vital role they play in providing habitat for organisms which utilise them as nurseries and feeding grounds. In many parts of the world concern has been expressed that the important functions and values of estuaries are being increasingly impacted upon by human activity. In South Africa diminishing freshwater inflow is a particular concern as this has led to an increase in the frequency and duration of mouth closure, which together with other factors has resulted in a marked deterioration in the condition of many estuaries.

Global environmental imperatives require an approach to ecosystem management that is defensible and sustainable in the long term. Current approaches to estuary management in South Africa do not meet these criteria, and consequently, this study set out to develop methodologies to address these shortcomings. Three modelling approaches are presented, which can be used independently, or conjunctively, in defining the freshwater requirements of estuaries. The models assess the consequences of change in freshwater inflow for (i) juvenile fish which utilise estuaries as nurseries, (ii) the availability of intertidal and species specific habitats, and (iii) the population structure and production of a common estuarine invertebrate (*Upogebia africana*), endemic to the region.

These techniques are applied in a case study of the Great Brak estuary (Western Cape, South Africa). The results indicate the utility of the approach and are supported, in part, by the findings of a long-term monitoring programme. The study also recognises the need for resource management to occur in the context of an integrated framework, which includes the explicit definition of ecological goals. Such a framework is presented, and is consistent with the Ecological Society of America's guidelines on sustainable ecosystem management.

As this approach has been devised to be applicable to South African estuaries, characterised by poor data availability, it is anticipated that methodologies will be equally applicable to estuaries in other developing countries with a similar lack of data. The methodologies also extend current international approaches to the management of estuary freshwater inflow, and would therefore be of value to estuaries in the United States of America, Australia and other regions where diminishing freshwater inflow has been raised as a concern.

PREFACE

The research described in this thesis was carried out at the Institute of Natural Resources, University of Natal, Pietermaritzburg, under the supervision of Professor Charles Breen (Institute of Natural Resources) and Professor John Hearne (Department of Mathematics and Applied Mathematics).

These studies represent original work by the author and have not otherwise been submitted in any form for any degree or diploma to any University. Where use has been made of the work of others it is duly acknowledged in the text.

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1 OVERVIEW

1.1 Introduction

This chapter provides an overview of the value and importance of estuaries as ecosystems, as well as providing a brief assessment of their present condition in South Africa. In outlining some of the threats facing South African estuaries, one of the most pervasive problems, that of diminishing freshwater supply, is highlighted. It is toward finding a management approach to addressing this issue that this study is directed. The remainder of this chapter will provide additional background, while Section 1.8 will set out the objectives and structure of this thesis.

1.2 The value of estuaries

Estuaries are undoubtedly important and valued features of the landscape. By virtue of their unique position at the interface of sea, river and land, estuaries support a distinctive assemblage of birds, fish and invertebrates, as well as characteristic floral communities such as mangrove swamps and salt marshes (Ketchum 1983; Kennish 1986). Although comprising only a fractional volume and area of the global hydrosphere, estuaries are considered to be highly productive ecosystems, sustaining an endemic flora and fauna and playing a vital role in the life history and development of numerous aquatic organisms (Odum 1970; Jefferies 1972; Day 1981a; McLusky 1981; Ketchum 1983; Kennish 1986).

The role of estuaries in sustaining both freshwater and marine populations of several species is now widely acknowledged. Thus marine populations of some species are considered to be dependent on the availability of estuarine nursery areas for juveniles of the species, and conversely, certain estuarine populations are known to be dependent on marine conditions for spawning (Whitfield 1994a). This functional interdependence of the freshwater, marine and estuarine environments creates a dynamic concept of estuary habitat which may be considered from the following perspectives :

(i) Estuaries as permanent habitats for endemic species

Certain species require estuarine conditions in order to exist, and consequently the continued survival of these species is dependent on the availability of estuarine habitats. In the development of an estuary association classification for fish, Whitfield (1994a) reports that 26% of the 142 estuary associated taxa are estuarine species which breed in estuaries, many of which are endemic. Cyrus (1991) lists 20 fish species endemic to South African estuaries; three of which are considered vulnerable, and five of which are considered rare (Skelton 1987). Day (1981a) reports 15 macrobenthic species restricted to estuaries.

(ii) Estuaries as permanent habitats for species which have an obligate marine phase in their life cycle

Some species, such as *Scylla serrata*, *Upogebia africana* and *Ambassis natalensis* are estuarine residents, but require marine conditions for spawning or larval development (Martin 1983; Robertson & Kruger 1994; Wooldridge 1991). In general, adults migrate to sea, spawn and return to estuaries. Eggs hatch at sea and then recruit into estuaries at a larval or postlarval stage. Alternatively, as in the case of *Upogebia africana*, larvae require marine conditions to complete a certain phase of their life cycle (Wooldridge 1991). In the latter case, eggs hatch within the estuary and postlarvae migrate to sea for a short period before returning to the estuary to continue their growth into adult prawns.

(iii) Estuaries as temporary habitats for species which have an obligate or opportunistic estuarine phase in their life cycle

Certain marine species such as *Rhabdosargus holubi* and *Penaeus* spp., recruit into estuaries as postlarvae or juveniles for varying periods of time. Cyrus (1991) has listed reasons for recruitment of marine species as; utilisation of calm waters and shelter (Blaber 1974), reduced predation pressure (Whitfield & Blaber 1978; Blaber 1980), food availability (Blaber & Whitfield 1977) and the presence of turbid water (Blaber & Blaber 1980; Cyrus & Blaber 1987a,b). In many cases it is not known whether recruitment is obligate or opportune. Some freshwater species such as the carid shrimps (*Macrobrachium* spp.) require brackish waters for larval development (Bickerton 1989), in which case they migrate from rivers into estuaries to complete this phase of their life cycle.

(iv) Estuaries as transient habitats for species which transfer between marine and freshwater conditions during their life cycle

Although anadromous species such as the salmon (*Salomar salar*) are absent from the South African fish fauna; Wallace *et al.* (1984) report that four species of catadromous eel migrate as adults from their freshwater environment to the sea to spawn. Juveniles then pass through estuaries back into rivers where they mature to adults.

(v) Estuaries as temporary habitats for opportunistic species

Estuaries are generally regarded as highly productive environments, enriched relative to the sea in both organic matter and nutrients (Kennish 1986). The detritivorous populations which this food source supports provides a resource base for piscivorous and benthic predators. Thus marine fish, many of which are favoured angling species, are frequently found in estuaries. In addition, a variety of bird species utilise estuaries opportunistically, using sand banks as roosting sites, and intertidal zones as a rich source of food.

In sustaining both freshwater and marine populations of fish and invertebrate species, estuaries are also in part sustaining the extensive recreational and commercial fisheries associated with the coastal zone. For example, over 55 % of the mass of the United States commercial fish and shellfish catch is dependent on estuaries, and these estuaries are known to have furnished an estimated 100 million days of sport fishing per year (Benson 1981). In Australia the gross value of production in the fishing industry was $ 1.4 billion in 1992/3 and recreational fishing was worth $ 2.2 billion in the 80's (Turner, Subak & Adger 1996), a significant component of which can be attributed to estuaries. In developing regions, such as the in the Pacific and Caribbean, nearshore fisheries are particularly important to humans as a food source, providing a substantial proportion (90 %) of coastal inhabitant's required protein intake (Lundin & Linden 1993).

Estuaries are also popular recreation venues and are subject to continually increasing utilization pressures. Being environments of high aesthetic value and offering a range of recreational opportunities; estuaries are prime areas for marina and resort development, and are consequently often the loci for the trend of increasing utilization of coastal resources in South Africa (Day & Grindley 1981; Sowman 1987), as well as in other parts of the world. According to Turner, Subak and Adger (1996) over half of the world's population lives within 60 km of the shoreline, and in many countries the growth rate of coastal populations is near to double that of the national growth rate. The trend of an increasing coastal population is expected to continue globally, particularly in Asia, Africa and South America, with a coastal population comprising two-thirds of the population of developing countries (3.7 billion) expected by the year 2000 (Turner, Subak & Adger 1996; World Resources Institute 1994).

In addition to direct utilisation pressures, estuaries are also subject to anthropogenic disturbances originating in the catchments which provide their freshwater inflow. Changes in land use, increased abstraction of water, increased soil erosion and the use of rivers for waste disposal all bring about change in estuaries. This change may be evident as an alteration in flow regime, water quality or circulation; or may result more directly in habitat degradation. Some have proposed that while estuarine ecosystems have evolved in concert with complex natural disturbance regimes, anthropogenic disturbances have altered the patterns and scales of natural disturbance to the extent that some systems have changed states (Breen, Whitfield & Wooldridge 1991). Although quantitative data supporting this hypothesis has yet to be collated, it is clear that population growth and concomitant urbanisation and industrialisation will impact on estuaries to an increasing extent (Day & Grindley 1981).

1.3 The condition of South African estuaries

Growing concern regarding the condition of estuaries in Natal (now KwaZulu-Natal) (Heydorn 1972, 1973) led to the initiation of a project funded by the Natal Town and Regional Planning Commission, to synthesise the available information on estuaries of the region. The study was commissioned in 1976 and completed in 1978, finding that only 20 of the 73 estuaries evaluated could be considered to be in a good condition (Begg 1978). Although Begg (1978) considered siltation to be the greatest threat to estuaries, flow diversion, and the construction of dams and weirs were also recognised as problems (Table 1.1).

A similar review of estuaries of the Cape coast was commissioned in early 1979 (Heydorn & Tinley 1980). Both these publications drew attention to the growing need for the formulation of policy to direct sustainable management in the coastal zone, and provided preliminary guidelines for land use planning and management in the coastal zone. Day and Grindley (1981) provided further insight into management needs in South Africa, assessing estuary management problems, reviewing appropriate legislation, and reiterating the need for sound management and the implementation of procedures such as Environmental Impact Assessments.

Table 1.1 : Summary of the various types and likely consequences of anthropogenic disturbance on the estuarine environment in Natal (Begg 1978).

<table>
<tr><th rowspan="2">FEATURE</th><th rowspan="2">EXPECTED ENVIRONMENTAL EFFECT</th><th colspan="2">PROBABLE RESULTS IN TERMS OF ESTUARINE PRODUCTIVITY</th></tr>
<tr><th>ADVERSE</th><th>BENEFICIAL</th></tr>
<tr><td>Incoming silt</td><td>Losses in storage capacity; losses of water area; increased turbidity.</td><td rowspan="2">Decreased production due to decreased light penetration; exclusion of visual feeders; suffocation of benthic fauna; decreased carrying capacity.</td><td>Reed encroachment; mud flat formation.</td></tr>
<tr><td>Dredging</td><td>Substrate disruption; turbidity; wetland disruption; increased tidal exchange.</td><td>Deepened areas offer refuge; increased passive transport of juvenile into estuary.</td></tr>
<tr><td>Disruption of riverine vegetation</td><td>Erosion and silt transport.</td><td>As above.</td><td>As above.</td></tr>
<tr><td>Disruption of wetlands</td><td>Silt transport; reduced winter flow; destruction of sources of detritus.</td><td>As above, with particularly severe decrease in productivity due to elimination of detritus sources.</td><td>None.</td></tr>
<tr><td>Drainage diversion</td><td>Reduced flow in estuary giving rise to mouth closure; or else increased water velocities giving rise to siltation.</td><td>Impeded immigration; decreased production at all levels.</td><td>None.</td></tr>
<tr><td>Dam construction</td><td>Modification of flow, giving rise to mouth closure.</td><td>As above; including diminished beach nourishment; barriers to movement.</td><td>Silt interception.</td></tr>
<tr><td>Road and rail construction</td><td>Reduced tidal exchange; reduced scour; water area losses; decrease in depth.</td><td>Decreased production at all levels.</td><td>None.</td></tr>
<tr><td>Weirs and causeways</td><td rowspan="2">Altered water levels; altered salinity; altered circulation.</td><td rowspan="2">Lowered carrying capacity; decreased utilisation; barriers to movement.</td><td rowspan="2">None.</td></tr>
<tr><td>Breaching</td></tr>
<tr><td>Pollution</td><td>Deterioration in water quality; increased biological demand for oxygen; presence of toxic compounds.</td><td>Modification of species composition and abundance; development of sludge communities; poor survival.</td><td>Increased birdlife; limited enhancement of fertility.</td></tr>
</table>

These concerns were again highlighted in a subsequent review of Natal estuaries (Begg 1984a), culminating in the publication of policy proposals for estuaries of the region (Begg 1985). Site specific assessments of the condition of certain estuaries (Blaber, Hay, Cyrus & Martin 1984) ensured that concerns remained in the public eye, ultimately leading to a national assessment of the status of South African estuaries (Heydorn 1986). This survey found that only 24 % of Cape estuaries and 28 % of Natal estuaries could be considered to be in a good condition. Twenty-two percent of the total number of estuaries assessed were considered to be in a poor condition (Figure 1.1) (Heydorn 1986).

The subsequent development of a community degradation index (Ramm 1988) again highlighted the extent to which some estuarine communities have become depauperate. Application of this index to 62 estuaries in Natal (Ramm 1990) revealed that only 21 % fell in the categories of relatively undegraded and slightly degraded, while 79 % were either moderately, strongly or severely degraded (Figure 1.1).

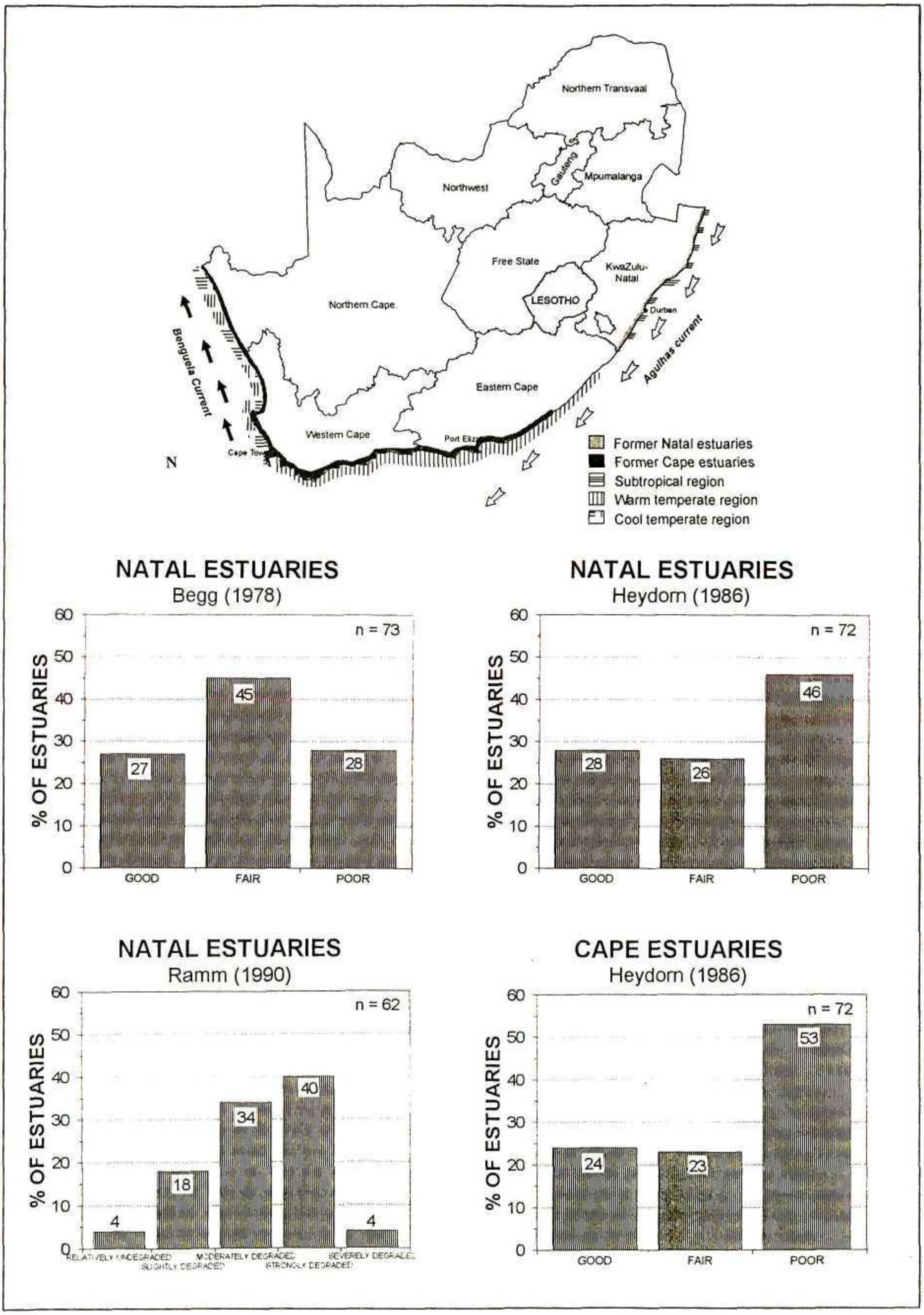


Figure 1.1 : Condition of Cape and Natal estuaries as reported in various studies (Begg 1978; Heydorn 1988; Ramm 1990).

The results of the most recent assessment of the condition of South African estuaries (Whitfield 1995) are shown in Table 1.2. Although this assessment finds that 60% of South African estuaries are in a good or excellent condition, it should be noted that this is primarily due to the large number of estuaries in an excellent/good condition which are located in the former Transkei region and were therefore not considered in previous assessments. For comparative purposes, Whitfield's (1995) assessment of Kwa-Zulu-Natal estuaries yields 48 % in a fair condition, 26 % in a poor condition, 25 % in a good condition, while only one estuary out of a total of 73 was considered to be in an excellent condition.

Table 1.2 : Current condition of South African estuaries (Whitfield 1995)

REGION	CONDITION			
	EXCELLENT estuary in a near pristine state (negligible human impact)	**GOOD** no major negative anthropogenic influences on either the estuary or the catchment (low impact)	**FAIR** noticeable degree of ecological degradation in the catchment and/or estuary (moderate impact)	**POOR** major ecological degradation arising from a combination of anthropogenic influences (high impact)
COOL TEMPERATE	1 (10%)	2 (20%)	2 (20%)	5 (50%)
WARM TEMPERATE	34 (28%)	52 (43%)	21 (17%)	13 (11%)
SUBTROPICAL	39 (33%)	22 (19%)	36 (31%)	20 (17%)
TOTAL	**74 (30%)**	**76 (31%)**	**59 (24%)**	**38 (15%)**

1.4 Key processes maintaining estuarine environments

While the popularity and recreational value of estuaries is related strongly to both the aesthetics and variety of recreational opportunities afforded by the coastal environment, the value of estuaries as ecosystems is more strongly linked to key physical processes occurring within the estuarine environment. Given the functional interdependence of estuaries and the adjacent environments referred to previously, processes which maintain an open mouth, or at least intermittent contact with the sea are therefore critical to the functioning of estuaries. Other processes such as nutrient cycling and flocculation, important for the maintenance of a productive environment occur as a result of the mixing of sea water with freshwater (McLusky 1981; Ketchum 1983; Kennish 1986). Freshwater inflow is therefore equally important, maintaining salinity gradients as well as keeping the range of salinity within limits and supplying nutrients, trace metals and organic matter. This section consequently focuses on tidal exchange and freshwater inflow as the key processes maintaining the estuarine environment.

(i) Tidal exchange

Tidal exchange is the movement of seawater into the estuary through the mouth during flood tides and the subsequent outflow seawards during the ebb tide. In addition to regulating the exchange of water, dissolved solids and suspended material between estuaries and the sea, tidal exchange is an important factor regulating the stability of the mouth (Day 1981b; Reddering 1988a,b). In particular, sediment loads of the flood tide play a critical role in mouth stability. The sediment load of flood tide water is in turn determined by coastal hydrodynamics and sea wave characteristics. When flow velocity into the estuary is reduced, the sediment load is deposited, resulting in an accumulation of sand on floodtide deltas. Subsequent ebb tides scour the deposited sediment, transporting it into the marine environment. When the rate of scouring of the mouth by tidal action is exceeded by the rate of deposition by wave action, sediment accumulates as a bar across the estuary mouth (Day 1981b; Reddering 1988a,b). The sand bar is removed by the erosive energy associated with a spring flood tide or by high fluvial flow velocities arising from a flood event.

(ii) Freshwater inflow

Although there is evidence suggesting some of the detritus supporting the typically extensive macrobenthic communities is produced within the estuary, freshwater inflow into an estuary no doubt supplements autochthonous production with nutrients and additional detritus (Day 1981c; Whitfield 1983). Furthermore freshwater inflow is a primary factor giving rise to the spatial variation in salinity, considered by many to be one of the definitive features of estuaries. Others have suggested that freshwater outflow provides valuable cues to the marine environment, enabling migration of catadromous species and recruitment of juveniles of many species of fish and invertebrates (Snedaker, de Sylva & Cottrell 1977). Recent findings indicate a more direct role, suggesting that the presence of strong longitudinal salinity gradients is a fundamental factor in determining the extent of recruitment of juvenile marine fish into estuaries (Whitfield 1994b).

The importance of freshwater inflow is possibly most apparent in the role it has in establishing and maintaining a free connection with the sea. Reddering (1988a,b) has shown that the channel dimensions of southeastern Cape estuaries are determined *inter alia*, by the flood discharge characteristics of the river. Reddering (1988a,b) suggests that these estuaries establish an equilibrium characterised by the flushing of tidally accumulated sediment by episodic river floods. The importance of these processes has also been highlighted in coastal ecosystems in KwaZulu-Natal (Quinn, Breen & Mander 1988).

1.5 Diminishing freshwater supply to South African estuaries

Earlier reviews of research needs identified diminished water supply to estuaries as a potentially serious problem (Begg 1978; Noble & Hemens 1978). Recent droughts in parts of the subcontinent have re-emphasised the need for research, and concerns about the effects of reduced freshwater inflow into estuaries have been expressed in the literature (Reddering 1988b; Whitfield & Bruton 1989; Whitfield & Wooldridge 1994; Allanson & Read 1995). In an assessment of surface water resource research needs in South Africa, the establishment of minimum flow requirements for estuaries was also identified as a research priority (Cousens, Braune & Kruger 1988). The findings of a national research programme on estuaries supported these concerns and gave impetus to the recognised need for the development of methodologies for determining the estuary freshwater requirements (Hay & Breen 1995).

The basis for the concern regarding diminished freshwater supply to estuaries is related to the continually rising demand for water, given a spatially and temporally inequitable supply. Most of South Africa (65%) receives less than 500 mm of rainfall annually and 21% receives less than 200 mm, while the average annual rainfall for the whole region (497 mm) is considerably less than the world average of 860 mm (Department of Water Affairs 1986). The growing demand for assured water supplies has necessitated the construction of large storage dams and several inter-basin transfer schemes. Numerous small agricultural dams, barrages and weirs have also been built to sustain stock watering and irrigation requirements. Instead of being available as streamflow to estuaries, water is stored and subject to consumptive losses, including high evaporation.

Possibly more significant than the loss of additional streamflow (volume) due to impoundment, is the alteration of the pattern of supply (volume & flow rate). Impoundments have the effect of attenuating floodpeaks, and the extent to which this occurs depends on the capacity of the reservoir and the characteristics of the particular flood event. As Reddering (1988a) has noted, the equilibrium between successive periods of deposition by wave action and scouring by episodic events can be readily disturbed by modifications in river discharge. If the scouring potential of flood events is reduced by impoundment attenuation, it follows that estuaries may open to the sea at a reduced frequency, thereby limiting the opportunity for key processes necessary for maintaining estuarine habitats to occur. Begg (1978) has argued that closure of the estuary mouth in KwaZulu-Natal estuaries has become an artificial norm, and furthermore that this often occurs at the most critical time for inshore spawning of estuarine dependent fish species.

Whitfield and Wooldridge (1994) have presented a summary of the likely responses of a typical estuarine system along the Eastern Cape coast of South Africa in response to differing freshwater supply regimes (Figure 1.2). The nett consequence of reduced freshwater inflow for these systems is a state of low variability and low diversity.

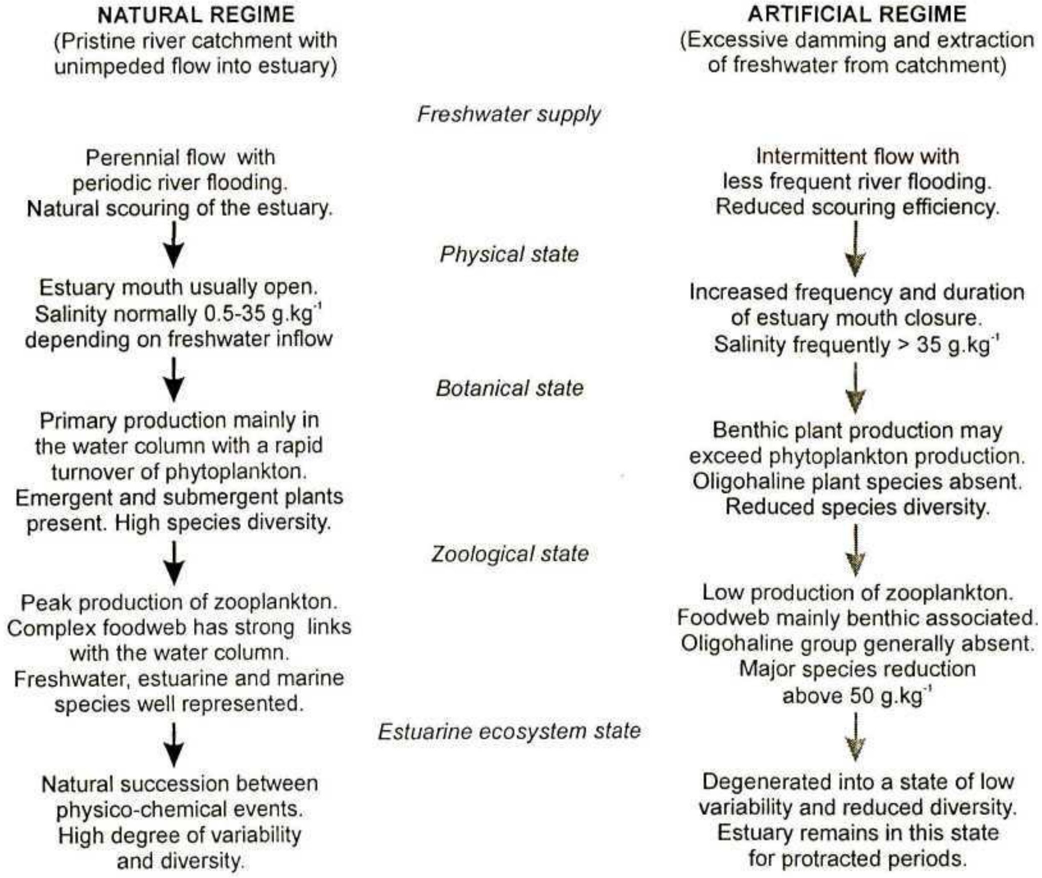


Figure 1.2 : Flow chart summarizing broad hypotheses arising from different freshwater supply regimes and estuarine response in a typical tidal Eastern Cape estuarine system (Whitfield & Wooldridge 1994)

1.6 Recent policy developments

Environmental issues have achieved greater prominence in international affairs in recent years, peaking in 1992 with the United Nations Conference on Environment and Development (the 'Rio Earth Summit') and the associated Rio Declaration. One of the major highlights of the Earth Summit was the opening for signature of the Convention on Biological Diversity. The objectives of the convention are "*the conservation of biological diversity, the sustainable use of its components and the fair and equitable sharing of the benefits arising out of the utilization of genetic resources.*" (Anon. 1994). More specifically, Article 10 of the Convention requires that Contracting Parties "*integrate consideration of the conservation and sustainable use of biological resources into national decision making*" and "*adopt measures relating to the use of biological resources to avoid or minimise adverse impacts on biological diversity*". Contracting Parties are also required to implement environmental impact procedures for proposed projects which may have significant adverse effects on biological diversity, with a view to avoiding or minimising such effects.

Although the signing of the Convention on Biodiversity by South Africa in 1994 exacts a greater commitment to the sustainable utilisation of natural resources such as estuaries, South Africa has been obliged to protect important estuaries since the signing of the Ramsar Convention by the South African Government in 1975. The Ramsar Convention (Convention on Wetlands of International Importance Especially as Wetland Habitat, 1971), was the first of the modern global nature conservation conventions, with the primary purpose of preventing the decline of wetland habitats (which includes estuaries and lagoons), and ensuring that they remain ecologically viable (Matthews 1993; Simon 1993). The signing of the convention by South Africa, committed the country to safeguarding and promoting the 'wise use' of the nation's wetlands and estuaries. With the publication of the World Conservation Strategy in 1980 (IUCN 1980), conservation or 'wise use' came to be defined as "*the management of human use of the biosphere so that it may yield the greatest sustainable benefit to present generations while maintaining its potential to meet the needs and aspirations of future generations.*"

South Africa has more recently embarked on a process aimed at developing a new environmental policy. This has resulted in the publication of a draft Environmental Policy for

South Africa, which is currently open for discussion (Department of Environmental Affairs and Tourism 1996). The Department of Water Affairs and Forestry (1996) has initiated a similar process aimed at producing a new Water Act for the country, and has presented for discussion, a summary of basic principles upon which the new Water Act will be based. A regional initiative to develop policy for marine environmental conservation in KwaZulu-Natal has recently been launched and a draft document (Anon 1996) is currently in circulation.

The draft Environmental Policy for South Africa stresses the importance of sustainable resource management, with specific objectives to improve biodiversity conservation, ensure the sound management of fragile ecosystems, and to integrate management and sustainable development with utilisation of the marine coastal zone. A specific objective is to ensure the sustainable and rational utilisation, conservation and management of water resources based on ecosystem and community needs (Department of Environmental Affairs and Tourism 1996). The latter objective is a key theme of the Water Law Principles (Department of Water Affairs and Forestry 1996) whereby "*the quantity, quality and reliability of water required to maintain the ecological functions on which humans depend should be reserved so that the human use of water does not individually or cumulatively compromise the long term sustainability of aquatic and associated ecosystems*". Indeed, it is intended this principle be specifically enshrined in the law in that "*the water required to meet peoples' basic domestic needs and the needs of the environment should be identified as 'the Reserve' and should enjoy priority of use.*"

1.7 Synthesis

Estuaries have come to be recognised for the diverse and unique plant and animal life endemic to their environments and the vital role they play in supporting aquatic organisms which utilise them as nurseries and feeding grounds. To a large extent, their efficacy in fulfilling these functions is dependent on the maintenance of at least a periodically open connection with the sea, allowing the exchange of nutrients, cues and organisms between the freshwater and marine environments. Increasing demand for freshwater resources in South Africa has resulted in the construction of impoundments with the consequence that freshwater flowing into estuaries is attenuated with respect to both volume and peak flow characteristics. As most South African estuary mouths are maintained in an open state by freshwater outflow, a reduction in outflow generally results in an increase in the frequency and length of mouth closure periods, thereby limiting biological potential of estuaries to fulfil the dimensions of habitat as defined in Section 1.2. The extent to which estuaries in certain regions of the country have already become degraded, and the likelihood that this trend will extend to other regions as development pressures increase, suggests that a conservation policy and associated management strategy for the region's estuaries is an urgent priority. In addition South Africa has recently committed itself to the conservation of biodiversity and the sustainable utilisation of natural resources. New policy developments require that the water requirements of natural environments such as estuaries are reserved; necessitating in turn, that the requirement of each estuary needs to be specified. This thesis will contribute to addressing these needs by presenting a range of techniques for determining the freshwater requirements of estuaries as well as a structured framework for managing the freshwater flow to the nations estuaries.

1.8 Objectives and structure of the thesis

Given the needs identified above, the primary objective of this thesis is to develop a resource management approach for estuaries, which is able to demonstrate ecological sustainability. Demonstration of ecological sustainability requires the establishment of a direct link between management action and the resultant functioning of an ecosystem in relation to the goal of ecological sustainability. The second objective of this thesis is therefore to develop and demonstrate these links. This in turn, requires the formulation of models representing the current state of our understanding of components of estuarine ecosystems. The third objective of this thesis is to assess the proposed management framework and associated models in relation to the requirements South African resource managers as well as those of other nations. It is hoped that in addressing the above objectives, a final objective will also be met. This objective is to rise to the challenge set out in the Water Law Principles (Department of Water Affairs and Forestry 1996), and develop a methodology for determining the freshwater requirements of estuaries.

This introduction has illustrated the need to reconcile the expanding human utilisation of estuaries, both direct and indirect, with their ecological value. The following chapter will review past and current approaches to estuary management in South Africa and abroad, particularly with regard to the freshwater requirements of estuaries. The purpose of the next chapter is to identify current management needs and will suggest an appropriate framework for the integrated management of estuaries, illustrating the need for the development of predictive capability in determining the freshwater flow requirements of estuaries. The subsequent three chapters will focus on the development of several alternative modelling approaches which can be used independently or conjunctively in defining the freshwater requirements of estuaries. The three approaches are demonstrated in a case study of the Great Brak estuary in Chapter 6. Chapter 7 presents a discussion of the utility of the three approaches, highlighting weaknesses and identifying further development requirements. The final chapter revisits the management framework identified in Chapter 2, providing recommendations for implementation and identifies further research needs.

2 APPROACH AND OBJECTIVES

2.1 International approaches to estuary freshwater requirements

Since the 1970's, growing concern has been expressed in the international literature regarding the reduction of freshwater flow to estuaries, primarily in semi-arid regions such as Australia and parts of the United States of America. This section highlights the problems of freshwater flow reduction in these countries in particular, and briefly reviews approaches to the management of estuary freshwater requirements.

2.1.1 Australia

Ruello (1973) suggested that the continued reduction of freshwater flow to Australian estuaries would have a persistent adverse effect on stocks of the commercially important prawn, *Metapenaeus macleayi*. In a review of aquatic resource management in Australia, Bayly (1975) reiterated this concern and endorsed Ruello's (1973) suggestion that cost-benefit studies and impact assessments of water conservation and flood mitigation projects should also consider the value of commercial and recreational fisheries. Ruello's (1973) concerns were highlighted in a subsequent study (Glaister 1978), which found a direct relationship between discharge from the Clarence River, New South Wales and the production of *Metapenaeus macleayi*. Staples (1985) later showed that annual catches of the prawn *Penaeus merguiensis* in the Gulf of Carpentaria were directly correlated with freshwater flow into the Gulf.

Pollution problems related to reduced freshwater inflow were also raised during the 1970's. In a study of the Hawkesbury estuary near Sydney, Wolanski and Collis (1976) showed that water quality reaches unacceptable limits during dry weather periods as pollutants are not diluted and removed by outflowing water. Connell, Bycroft, Miller and Lather (1981) later reported on the impacts to the Fitzroy River estuary in Queensland due to the construction of a barrage approximately 60 km upstream from the estuary mouth. The Fitzroy River is the largest river system in Queensland, and one of the few permanent river systems in the state. In addition to increasing the residence time of estuary water, construction of the barrage resulted in substantial reductions in dissolved oxygen concentration and increases in nutrient and chlorophyll *a* concentration during conditions of low discharge (Connell *et al.* 1981).

Construction of two dams on the Gordon River in Tasmania for hydroelectric power development resulted in the elimination of the estuarine underflow of salt water upon which meromixis in three riverine lakes, thought to be the shallowest and most striking examples of meromictic lakes in the world, are dependent (King & Tyler 1982). In addition variability in inflowing water has been replaced by flow of almost invariant temperature and chemical composition, characterised by higher summer flows and lower winter flows than those which occurred under natural conditions (King & Tyler 1982).

According to Pigram (1991), the water resources of many of Australia's rivers are already overallocated, and rivers in Australia have been regulated for irrigation, for stock, domestic and industrial water supply, hydroelectric power generation, and even to exclude estuarine salt water intrusion (McMahon & Finlayson 1991). McMahon and Finlayson (1991) suggest that river regulation on the Australian continent has been characterised by a lack of knowledge about the environmental consequences of this activity as well as an apparent lack of concern.

According to Brown (1995) between 1988 and 1994 at least six major enquiries into coastline pressures in Australia identified a common set of environmental problems; declining fisheries, degraded foreshores, diminishing water supplies, vanishing estuaries and polluted coastal waters. In an assessment of the state of the marine environment in Australia, Zann (1995) lists increased sedimentation, flow alteration, eutrophication and acidification due to drainage from disturbed acid soils. Sixty-four percent of estuaries in New South Wales and 22% in Victoria are considered to have poor water quality. Poor water quality and a reduction in habitat is thought to be a threat to fisheries in 21% of estuaries in New South Wales and 23% in Victoria (Zann 1995). Coastal lakes, largely restricted to the densely populated south-eastern coastal regions, are of particular concern. These systems, as well as the Peel-Harvey system in Western Australia are subject to declining water quality and increased eutrophication due to reduced tidal flushing. Loss of coastal saltmarshes and particularly, temperate seagrass beds has been raised as a serious concern (Zann 1995).

The top five concerns raised in the assessment of the state of the marine environment in Australia were; (1) declining marine and coastal water / sediment quality, particularly as a result of inappropriate catchment land use practices, (2) loss of marine and coastal habitat, (3) unsustainable use of marine and coastal resources, (4) lack of marine science policy and lack of long term research and monitoring of the marine environment, and (5) lack of strategic, integrated planning in the marine and coastal environments (Zann 1995). As the source of the key marine environmental threats lie inland, integrated catchment planning is considered by Zann (1995) to be "*almost as important to the sea as it is to the land*". Australia has formally adopted integrated catchment management in several states (Mitchell & Hollick 1993), with the inter-state Murray-Darling Basin Initiative (Blackmore 1995) perhaps being the most well known. The Australian approach is based on extensive community participation and has yielded considerable success (Mitchell & Hollick 1993; Blackmore 1995).

Of the three states comprising the Murray Darling Basin, Victoria was the first to initiate a legislative and policy framework which most explicitly recognises the environment as a legitimate user of water (McPhail & Young 1991). The 1989 Water Act permits the Minister to allocate water for environmental purposes in two ways. Firstly as a bulk entitlement which may be granted as a volume, level of flow or a share of flow or storage, and secondly as a licence for instream use of water. The latter would be issued to ensure that a storage facility's capacity is managed to maintain the volume and timing of flow required for the purposes of maintaining the downstream environment (McPhail & Young 1991). Furthermore, in new water resource developments, the proponents must firstly estimate the potential impacts of the proposed development on the environment and secondly determine the water requirements of the downstream environment and include these in the design of the project (McPhail & Young 1991).

Recognising the need to address the environmental water requirements of aquatic ecosystems, an international seminar and workshop on 'Water Allocation for the Environment' was convened by the Centre for Water Policy Research (Pigram & Hooper 1991). The seminar examined approaches to determining instream flow requirements as well as pricing mechanisms to improve water use efficiency (Syme & Nancarrow 1991; Musgrave & Kaine 1991; Dudley 1991). The introduction of water markets and mechanisms for pricing water, feature strongly in the subsequently formulated National Strategy on Ecologically Sustainable Development (Department of Environment, Sport and Territories 1992).

The National Strategy on Ecologically Sustainable Development (1992) sets out Australia's policy objectives for water resource management, stressing the need to manage in an integrated way, the quality and quantity of surface and groundwater resources, and the need to develop mechanisms for water resource management which will maintain ecological systems while meeting economic, social and community needs (Table 2.1). The National Strategy for Environmentally Sustainable Development commits Australia to manage water allocations of instream and floodplain ecosystems, and presumably includes the freshwater requirements of estuaries, although this is not made explicit.

Table 2.1 : Objectives for water resource management in Australia's National Strategy for Environmentally Sustainable Development (Department of Environment, Sport And Territories 1992)

OBJECTIVE 18.1: *to develop water management policies which are based on an integrated approach to the development and management of water resources*
Governments will : • continue to encourage and support actions to develop and adopt an integrated catchment management approach to water resources • continue to improve coordinating mechanisms and policy initiatives for improved water resource management • develop improved measures for effective public participation in development of water pricing and allocation policies and water resource management measures • introduce legislative and policy frameworks for the protection of aquatic ecosystems which are based on an integrated catchment approach • improve management of water allocations to ensure the maintenance of in-stream and floodplain environmental values, including the development of more effective legal and policy frameworks • finalise consideration of the national water quality management strategy, in particular those aspects dealing with concerted action to address blue-green algal blooms • have regard to the wide range of action recommended to achieve the overall objective of the national water quality management strategy, this being sustainable use of the nation's water resources by protecting and enhancing their quality while maintaining economic and social development • consider the whole hydrological cycle in water management planning, including stormwater, waste waters and effluents • have regard to the recommendations arising from the Industry Commission's report on water resources and waste water disposal
OBJECTIVE 18.2 : *to develop and implement the most effective mix of water resource management mechanisms*
Governments will : • examine and determine within their jurisdiction, the most effective mix of water resource management mechanisms, including appropriate pricing policies, regulatory measures, long term monitoring strategies, adequate research support, better utilisation of existing infrastructure • give thorough consideration of the range of technological, economic, environmental, and social factors when upgrading or providing new infrastructure • continue working within their own jurisdictions to review the operation of their water management sections • continue to develop methodologies for the determination of environmental externalities into water pricing • encourage more rapid adoption of water pricing structures, including where appropriate, complete pay-for-use tariff policies, which more accurately reflect the price of delivery • focus on improving water markets and mechanisms for introducing more comprehensive systems of transferable water entitlements • continue to pursue institutional reform of water agencies • give consideration to ensuring that water agencies are given an opportunity to review particular types of land-use development proposals of particular concern to them • encourage further work on identification of sustainability research priorities for water resources and development of a national strategy for water information management • consider the feasibility of a national approach to establishing inventories on the condition and extent of wetlands, floodplains and riparian ecosystems, as a basis for ensuring their long term protection

In 1990, New South Wales initiated a State Rivers and Estuaries Policy, which incorporates an Environmental Flows Strategy. The latter provides for a variety of mechanisms to maintain the ecological integrity of rivers and downstream habitats, including storage operating rules, environmental contingency allowances and environmental flow provisions for surplus and unregulated flows (McPhail & Young 1991). The purpose of the contingency allowances is to provide flexibility for environmental flow managers during regulated flow periods when problems arise, for example, flushing to maintain water quality. If these problems do not occur the unused contingency can be reallocated progressively to other sectors during the season as risk to the environment declines (McPhail & Young 1991). The concept of environmental contingency flows is being currently tested, and interim environmental allocations have been announced (Department of the Environment, Sport and Territories 1996). Recently other states in Australia have introduced environmental water allocations. For example South Australia has prepared a 1995 State Water Plan which makes provision for water pricing, community involvement and water allocations for the environment (Department of Environment, Sport and Territories 1996).

2.1.2 United States of America

Gunter (1963) highlighted the significance of freshwater inflow to the productivity of estuaries, describing the section of the Gulf of Mexico at the mouth of the Mississippi River as the "*fertile fisheries crescent*" (Benson 1981). The importance of estuaries to fisheries was later synthesised (Smith, Schwartz & Massman 1966), with McHugh (1966) and Copeland (1966) stressing the potential consequences of flow reduction to estuarine fish productivity and ecology, while Chapman (1966) showed a direct relationship between commercial catch data and freshwater inflow for six Texas estuaries. The significance of freshwater inflow to estuaries has been reemphasised with the recent publication of a review of the effects of river regulation and diversion on marine fish and invertebrates (Drinkwater & Frank 1994).

According to Chapman (1977), the National Estuary Protection Act Study (U.S. Department of the Interior 1970) was the first attempt to assess the problem of freshwater inflow reduction at a national level, finding that although not a source of concern for all estuaries, it was sufficiently serious to warrant further study. Of the 10 biophysical regions identified and assessed, significant problems were identified in 6 of the regions; the Middle Atlantic, Caribbean and south Florida, Gulf of Mexico, Pacific Southwest, Pacific Northwest , and the Pacific islands. It was considered a less significant problem in Chesapeake Bay, the South Atlantic regions and in Alaska (U.S. Department of the Interior 1970). During the early 1970's several site specific studies focussed on the potential impacts of flow reduction, or documented deteriorations in estuarine productivity (Chapman 1971, 1973; Cronin, Gunter & Hopkins 1971; Copeland, Odum & Cooper 1972; Gagliano, Kwon & van Beek 1972). In response to these studies, legislation enacted by the Texas Legislature in 1975 (Senate Bill 137) provided a mandate for comprehensive studies of the effects of freshwater inflow to the bays and estuaries of Texas (Martin 1987).

Chapman (1977) considered the most serious areas of concern to be Florida Bay, the Ten Thousand Island Region near the Everglades, the Mississippi Delta, most of Texas estuaries and San Francisco Bay. According to Benson (1981), the adverse effects of reduced inflow have been noted in estuaries on the Atlantic and Pacific coasts and the Gulf of Mexico. Problems along the Atlantic coast are primarily; poor water quality, increased sediment load and modification of seasonal flow regimes. Diversion of water for irrigation and industry has resulted in a reduction of inflow to Chesapeake Bay and other Atlantic coast estuaries, with a corresponding increase in contaminants and sediments. Benson (1981) suggests that San Francisco Bay perhaps offers the most striking example of the effects of freshwater flow reduction, with inflow being half of natural volumes. Chapman (1977) urged coastal zone managers and scientists to "*determine how much freshwater is essential for each of our most valuable estuaries, when it must be received, and what its quality must be*", noting also that "*it is equally important, however, to document what values and benefits accrue by providing such water to the coastal zone and what values and benefits will be lost if it is not provided*". In the Second National Water Resources Assessment, approximations of the freshwater requirements of estuaries were developed (United States Water Research Council 1978), although Benson (1981) suggested that these needed to be refined if they were going to be used to sustain estuarine resources effectively.

Benson (1981) reported on progress in solving freshwater inflow problems, as discussed at a national symposium on freshwater inflow to estuaries (Cross & Williams 1981). Several multi-institutional teams initiated research efforts focussed on important estuarine systems. For example, at Chesapeake Bay the U.S. Army Corps of Engineers, the Environmental Protection Agency, the U.S Geological Survey, the U.S. Fish and Wildlife Service as well as universities, state and regional groups established a collaborative programme utilising the U.S. Army Corps of Engineers' hydrodynamic model to evaluate the effects of altering freshwater inflow on Chesapeake Bay salinities (Benson 1981).

The Texas Department of Water Resources initiated the development of several approaches to determining the freshwater requirements of estuaries, a research programme which has continued through until present times. Key developments are summarised below :

(i) Lambert & Fruh (1978)

Lambert and Fruh (1978) suggested a conceptual methodology, formulated as an open structure into which available information and analytical techniques could be inserted on a problem-specific basis (Figure 2.1). A particular requirement in the development of the approach was that it was to be responsive to the ecologically-oriented freshwater management goals for an estuary. The approach was tested using the Corpus Christi Bay as a case study.

The first step (Step 0) was considered to be a policy step arbitrated by a public agency, and requires the translation of an estuarine management policy into a set of management criteria. For example the management goal for the Corpus Christi Bay System was stated as "*the fresh and saline water resources of the Corpus Christi Estuarine System shall be managed in such a manner as to provide an estuarine environment conducive to the maintenance of current estuarine fisheries*" (Lambert & Fruh 1978).

The purpose of the second step (Step 1) is to determine the total net amount of freshwater required to meet the ecological goals stipulated in the previous step, and represents translation of qualitative policies into quantitative management criteria. Freshwater inflow is considered to comprise river inflow, local runoff, direct precipitation and return flows, less the evaporative losses. This step requires two tasks; (a) the identification of one or more estuarine organisms as indicator organisms, and (b) the identification of one or more environmental components which are responsive to freshwater inflows and to which the indicator organisms relate. Lambert and Fruh (1978) specify several additional criteria for each of these tasks :

The environmental control component(s) should;

- be amenable to routine measurement with a time delay between sample acquisition and analysis results less than the estuarine system response time,
- have a predictable effect upon the behaviour of the indicator organism(s), and
- be amenable to mathematical modelling

The indicator organism(s) should;

- be responsive to the environmental control components(s),
- be in some way indicative of the desired biological state of the estuary, and
- be useable as an indicator without extensive, complex or costly procedures.

In the implementation of the case study only a single organism, the spotted seatrout (*Cynoscion nebulosus*) and one environmental control component were selected, due to the limited availability of biological data. Available information indicated that salinity should not exceed 27 $g.kg^{-1}$ for this species during the spawning season. Available hydrodynamic models were utilised to demonstrate that an average net flow of 5.7 $m^3.s^{-1}$ between the months of April and September would be required to satisfy the salinity constraint. This step is obviously reliant on the availability of suitable models which permit evaluation of the behaviour of environmental control component under a wide set of freshwater inflow conditions. Lambert and Fruh (1978) suggest several criteria for the selection of such models.

The third step (Step 2), requires the input of available climatological, hydrological and demand information into a systems model of the supply area, minimum criteria for which are also provided by Lambert and Fruh (1978). The control variables of the systems model are then manipulated according to assumed or prescribed operation plans. The freshwater flow requirement then becomes part of the operational constraints, and the capability of the flow scenario to meet the policy requirements identified in the first step is measured by the resulting freshwater release schedules.

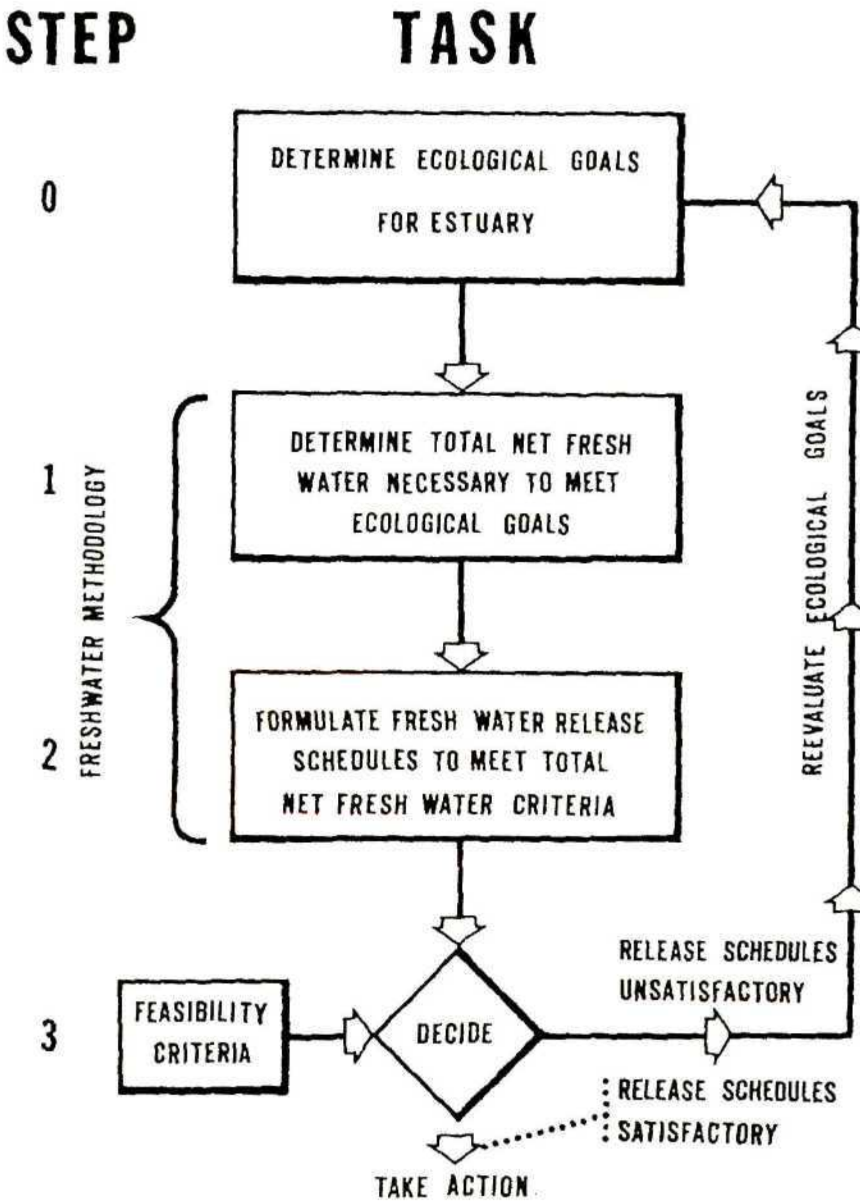


Figure 2.1 : Freshwater release schedule planning sequence containing the freshwater methodology (Lambert & Fruh 1978)

The final step again occurs under the arbitration of a public agency and requires a set of specific feasibility criteria against which the freshwater release schedules are evaluated, such as the risks associated with associated freshwater storage levels or the desired operating levels of upstream reservoirs. If the release schedules are considered acceptable they become the basis for management. Alternatively a re-evaluation of the original ecological goals and feasibility criteria would be required. The planning process becomes iterative until ecological and impoundment operating goals are achieved.

The implementation of the approach for the Corpus Christi system indicated that a substantial increase in the freshwater allocation to the estuary, while reducing storage in Lake Corpus Christi over a monthly, summer average and two-summer running average basis, would not result in a significant increase in persistent system stress and would maintain the spotted seatrout fishery. The final step (Step 3) was not implemented in the case study as adequate, quantified feasibility criteria were not available.

Although the use of a single control variable and single indicator organism was considered unacceptable for actual flow management, the study demonstrated a feasible methodology in addition to raising several important research needs. In particular a rational method for negotiating environmental goals for estuaries was suggested (Lambert & Fruh 1978).

(ii) Texas Department of Water Resources 1980a,b; 1981a,b,c; 1982; 1983

In response to the mandate of the Texas legislature, several studies of Texas estuaries were undertaken over a six year period during the late 1970's and early 1980's , with the purpose of determining the effects of reduced amounts of freshwater inflows due to existing and proposed inland reservoirs (Funicelli 1984; Martin 1987).

These studies addressed several approaches to assessing the effects of freshwater inflow, namely; the responses of key species, impact on the commercial fishery harvest; effects on nutrient inflow and innundation of deltaic marsh (Funicelli 1984). In these studies the physical, chemical and biological factors were empirically related (Martin 1987). For example, the anticipated flow reduction could be related to an anticipated increase in salinity, which could then be assessed in relation to the salinity preferences for the key species under consideration. Similarly, the availability of datasets relating inflow to harvesting enabled regression relationships to be developed which could then be used to determine the anticipated net reduction in harvest. These studies resulted in 19 statistically significant regression equations that related to eight harvest groups (shellfish, all penaeid shrimp, white shrimp, blue crab, oyster, finfish, spotted seatrout and red drum) (Funicelli 1984). As noted by Funicelli (1984), the additional advantage of the latter approach is the immediate transition from inflow reduction to economic impact.

(iii) Martin (1987)

Recognising the need for a more integrative approach, Martin (1987) reviewed the available studies (Texas Department of Water Resources 1980a,b; 1981a,b,c; 1982; 1983) and using components of previous approaches, presented a optimisation model for determining estuarine freshwater requirements (Figure 2.2).

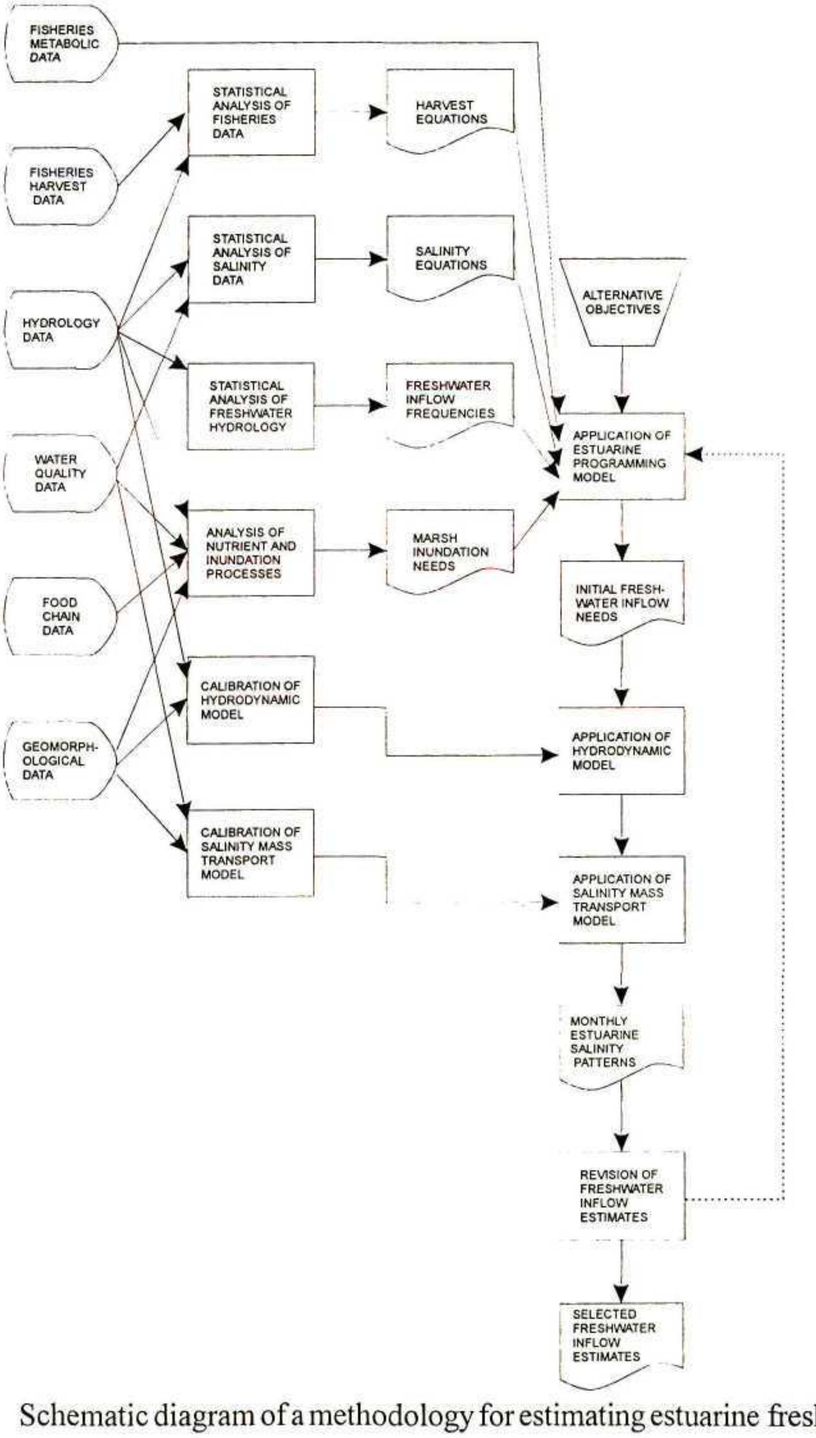


Figure 2.2: Schematic diagram of a methodology for estimating estuarine freshwater inflow needs under a management alternative (*after* Martin 1987).

The methodology is based on the analysis of six basic databases and the development of statistical relationships between the key indicator parameters of salinity, commercial fishery harvests and marsh innundation, and historical monthly and seasonal freshwater inflows. The estimated freshwater inflow needs are then determined by finding the 12 monthly inflow values from each of the major river basins which then either minimises the total annual inflow or maximises the annual fisheries harvest, subject to the constraints on average monthly salinities, annual commercial fisheries harvests, and monthly inflows (Martin 1987). The problem was usually constructed as a non-linear programming problem and solved either by a steepest descent, gradient algorithm, or the GRG2 nonlinear programming algorithm (Lasdon *et al.* 1980). Having determined an initial estimate, the flow regime is then routed through a one-dimensional transport and mass balance model to determine whether the inflowing freshwater will result in acceptable salinity distributions throughout the estuary. If not, initial estimates are revised and the process is repeated (Martin 1987).

(iv) Tung, Bao, Mays and Ward (1990)

Recognising the limitations of Martin's (1987) approach, Tung, Bao, Mays and Ward (1990) presented a revised methodology for determining the freshwater requirements of Texas estuaries. According to these authors the weaknesses of Martin's (1987) approach were; (1) the nonlinear aspects of the problem were suppressed by linearisation of all constraints, (2) the full multiobjective nature of the problem, e.g. minimisation of inflow and maximisation of harvest, was not considered, and (3) regression equations with considerable statistical uncertainty were used as deterministic constraints in the linear programming model.

In order to overcome these limitations Tung *et al.* (1990) developed a method using chance constraints in a nonlinear programming model to explicitly account for the uncertainty in the salinity regression equation. The methodology was applied to seven major estuaries along the Texas coast and in addition to providing revised freshwater inflows, demonstrated the associated achieved reliabilities for the salinity and harvest constraints. The latter is significant in that this term reflects the combined effect of uncertainty in regression equations, range of salinity bounds and any other constraints (Tung *et al.* 1990). However, the authors recommended that an alternative approach to the use of salinity regression equations be sought. The reason for this is that the equations do not represent the complex hydrodynamic processes which give rise to spatial and temporal variations in salinity. These processes, representing the interaction of tides, currents, winds and advection / dispersion contribute to the high variance about the regression, giving rise to large uncertainty in the dependence of salinity on inflow and consequently the low achievable reliability. Although the authors suggested that the approach could be extended to consider the multiobjective nature of the problem, this was not undertaken in their study.

(v) Bao and Mays (1994a,b)

In subsequent approaches, Bao and Mays (1994a,b) interfaced a nonlinear programming optimizer in an optimal control framework, with a hydrodynamic transport model to implicitly solve the hydrodynamic-salinity constraint equations for salinity levels. Previous approaches thus treated salinity on a steady state basis, whereby seasonal variation in salinity was accommodated by subdividing the year into several seasons and solving the steady state problem separately for each season. In the revised approach the salinity regression equations are replaced by a two-dimensional hydrodynamic transport model to simulate circulation and spatio-temporal variability in salinity. Application of the approach to the Lavaca-Tres Palacios estuary indicated an improvement over the previous approaches (Martin 1987; Tung *et al.* 1990), as the Tung *et al.* (1990) model was found to underestimate the minimum annual optimal flow for low reliability values and overestimates the annual total flow for high values.

(vi) Zhao and Mays (1995)

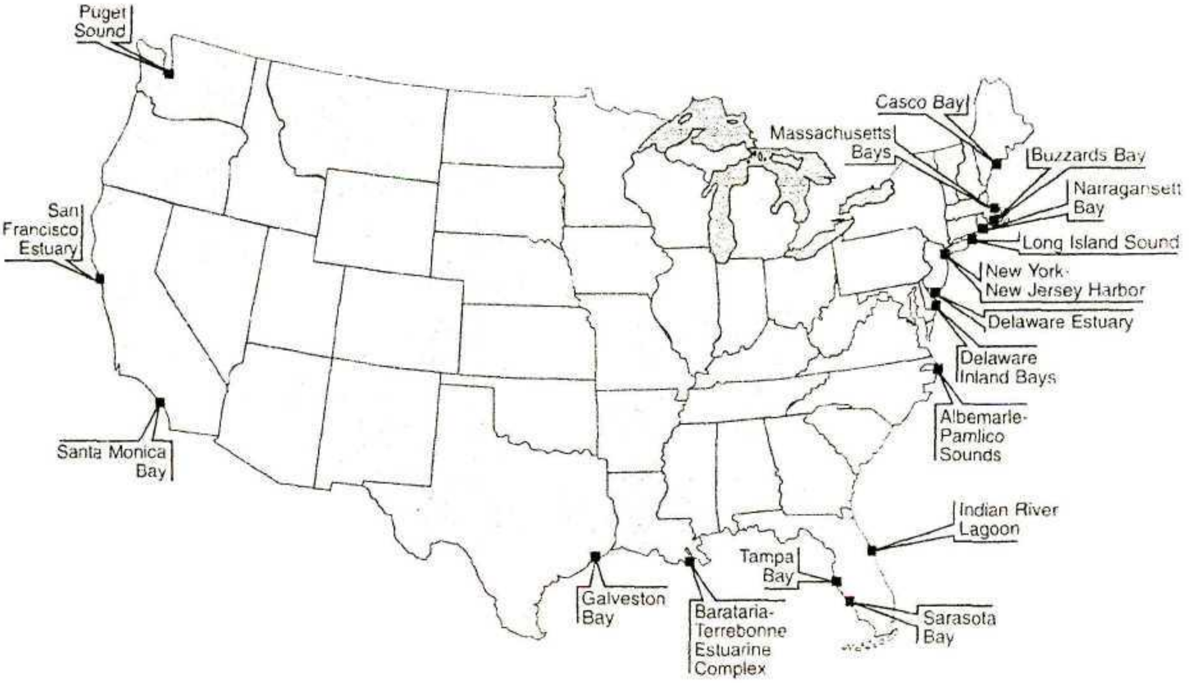
Zhao and Mays (1995) present a new approach based on discrete-time stochastic linear quadratic control. The feedback-control model enables decision makers to determine the upstream reservoir releases immediately subsequent to measurement of salinity and nutrient levels at specified locations in the estuary. Optimal upstream reservoir releases are determined so that the salinity and nutrient levels at these locations are as close as possible to the prescribed levels for the remaining months, as determined by statistical expectation. The approach was demonstrated for the Lavaca-Tres Palacios estuary, providing managers with a real time tool for the management of releases from impoundments in the system.

Texas estuaries have been the focus for the development of techniques for determining the freshwater requirements of estuaries, and are the subject of a continuing multi-institutional research programme (Jensen 1994; Gerston 1995). Elsewhere in the United States, the focus of estuary management has been the 28 estuaries designated as 'Estuaries of National Importance' (as of June 1995). The National Estuary Program was established by Congress in 1987 to protect and restore the health of estuaries while supporting their economic and recreational values (Gerston 1995). For each estuary a Comprehensive Conservation and Management Plan is developed, and addresses the following areas;

- water and sediment quality, focussing on pollution abatement and control,
- living resources, including restoration as well as protection of special habitat areas, and
- land use and water resources, using regulatory and non-regulatory means to conserve land and water (US Environmental Protection Agency 1992).

The first five years of the National Estuary Programme ended in 1992. The progress report (US Environmental Protection Agency 1992) outlines the priority problems for the then 17 estuaries of national significance, to be addressed in the subsequent phase. Figure 2.3 shows the location and characteristics of these estuaries and Table 2.2 summaries the priority problems in each estuary. Freshwater inflow related problems are highlighted for emphasis.

As indicated in Table 2.2 below, hydrological modifications of freshwater inflow are of particular concern in the case of the San Francisco Bay / Sacramento - San Joaquin Delta estuary. According to the San Francisco Estuary Project (1993), aquatic resources in the estuary have declined considerably due to prolonged drought and large diversions of freshwater. However, as a consequence of the complex configuration of the Delta and estuary, as well as the complex withdrawal and diversion network, it is not possible to measure freshwater discharge into the estuary directly. Recognising the need to identify a management index which is based on the response of the estuary to fluctuations in the input of freshwater, the San Francisco Estuary Project convened a series of workshops. In addition to this need, additional attributes were identified, such as; the need for the index to be simple and inexpensive to measure accurately, the index should integrate a number of important estuarine properties and processes, and should also be meaningful to a large number of constituencies.



ESTUARY	PHYSICAL and HYDROLOGIC FEATURES: Estuarine Drainage Area (100 sq.mi.)	Total Drainage Area (100 sq.mi.)	Water Surface Area (sq.mi.)	Average Depth (ft.)	Avg. Daily Fresh-water Inflow (100 cfs)	Volume (billion cu.ft.)	NATURAL RESOURCES: Wetlands (sq.mi.)	Classified Shellfish Waters (sq.mi.): Approved	Classified Shellfish Waters (sq.mi.): Total	ECONOMIC ACTIVITIES: Population Density, 1980 (per sq.mi.)	Land Use (% of EDA): Urban	Land Use (% of EDA): Agriculture	Point Sources of Pollution: Industrial	Point Sources of Pollution: MWTP	SUSCEPTIBILITY TO POLLUTION: Dissolved Concentration Potential	Particle Retention Efficiency
Casco Bay	12	12	164	42	21	191	61	144	168	172	14	10	40	10	M	H
Massachusetts Bays	27	27	981	180	65	2014	141	79	141	5409	122	6	115	34	M-H	M-H
Buzzards Bay	6	6	228	34	12	215	75	184	199	780	13	6	7	5	H	H
Narragansett Bay	13	18	165	30	32	139	155	110	165	1065	26	9	113	24	M	H
Long Island Sound	72	172	1281	62	300	2192	315	1122	1342	1008	25	14	226	87	L	H
Hudson River/Raritan Bay	85	165	298	21	267	172	269	0	257	1471	24	25	582	287	M	M
Delaware Estuary	48	135	768	21	198	448	641	549	623	1082	24	42	181	153	M	M
Delaware Inland Bays	3	3	32	4	3	4	N/D	19	30	126	10	46	9	7	H	M
Albemarle-Pamlico Sounds	116	296	2949	13	460	1081	1768	2524	3088	182	9	59	103	84	M	M
Indian River Lagoon	12	12	280	7	14	51	161	34	106	327	17	30	14	54	H	H
Sarasota Bay	3	3	44	6	4	8	N/D	N/D	N/D	923	28	26	6	11	H	M
Tampa Bay	26	26	346	13	24	123	394	9	88	476	17	40	69	29	H	H
Barataria-Terrebonne Bays	38	38	1326	11	101	200	1458	623	782	270	9	30	1	51	M-H	M
Galveston Bay	45	245	540	6	152	92	374	0	547	665	16	50	747	566	M	M
Santa Monica Bay	5	5	211	314	9	1844	4	0	0	4088	60	2	6	4	M	H
San Francisco Bay	111	863	716	42	370	412	909	0	125	1321	27	66	178	87	M	M-H
Puget Sound	111	197	1292	585	942	7081	387	94	141	495	27	16	308	72	L-H	M-H
NEP TOTAL	733	2223	11621	1391	2974	16267	7112	5491	7802	19860	468	497	2705	1565		

Abbreviations: sq.mi., square miles; ft., feet; avg., average; cfs, cubic feet per second; cu.ft., cubic feet; EDA, estuarine drainage area; MWTP, municipal wastewater treatment plant; H, high; M, medium; L, low; N/D, no data

Source: NOAA, "Estuaries of the United States: Vital Statistics of a National Resource Base." A Special 20th Anniversary Report. U.S. Department of Commerce, October 1990

Figure 2.3 : Location and characteristics of the 17 estuaries in the National Estuary Programme (US Environmental Protection Agency 1992)

Table 2.2 : Priority issues in each of the 17 estuaries in the National Estuary Programme (US Environmental Protection Agency 1992)

ESTUARY	PRIORITY PROBLEMS	
Albemarle-Pamlico Sounds	• declines in fisheries productivity • impairment of nursery area function • anoxia-related fish kills • shellfish closures	• health of aquatic resources • eutrophication • habitat loss • changes in distribution of bottom dwelling organisms
Buzzards Bay	• pathogen contamination • increasing nitrogen	• toxic contamination
Long Island Sound	• hypoxia • pathogens • living marine resources	• toxic contamination • floatable debris
Narragansett Bay	• management of fisheries • impact of toxic contaminants • recreational uses • land use	• nutrients • health and abundance of living marine resources • health risk to consumers of contaminated seafood
Puget Sound	• control sources of toxic substances • increase protection of shellfish beds • support long term research • continue and maintain established programs • prevent spills and enhance response capability	• cleanup of toxic substances where sources are controlled • protect and stop loss of wetlands and other aquatic habitats • improve control and cleanup of non-point source pollution • support and improve public improvement and education
San Francisco Estuary	• decline of biological resources • dredging and waterway modification • intensified land use	• increased pollutants • **freshwater diversions and altered flow regimes**
Delaware Estuary	• water quality • **water supply**	• habitat • living resources
Delaware Inland Bays	• nutrient over-enrichment • land-use planning	• loss and alteration of habitats
Galveston Bay	• reduction / alteration of living resources • shoreline erosion	• public health issues • **water resource management issues**
New York - New Jersey Harbour Estuary	• pathogen contamination • toxic contamination • habitat loss and alteration / living resources	• floatable debris • nutrient and organic enrichment
Santa Monica Bay	• loss and degradation of wetlands • human health risks associated with disease-causing pathogens in the surfzone • impact of pollution on the benthic (bottom-dwelling) community • impact of pollution on the pelagic (open-ocean) community	• human health risk from eating contaminated seafood
Sarasota Bay	• decline in water quality • habitat loss • inadequate and inconsistent public access and overuse of resources	• stormwater and wastewater • decline in finfish and shellfish populations
Barataria-Terrebonne Estuarine Complex	• **hydrological modification** • habitat loss and modification • eutrophication • toxic substances	• reduced sediment flows • changes in living resources • pathogen contamination
Casco Bay	• toxic waste • nutrients • combined sewer overflows	• lack of enforcement • bacteria • balancing economic development with environmental protection
Indian River Lagoon	• increased nutrient loadings • lagoon circulation • increased levels of pathogens • loss of emergent wetlands and their isolation from the lagoon	• increased suspended matter loadings and sedimentation • loss of seagrass beds and increased stress on remaining beds • increased input of toxic substances
Massachusetts Bays	• toxics • pathogen contamination • habitat loss and modification	• bioaccumulation of toxics • water quality • sea level rise
Tampa Bay	• water quality deterioration / eutrophication • lack of community awareness • circulation and flushing • increased user conflicts between various recreational activities, industrial and navigational needs, and urban development	• loss of habitat, including seagrasses and emergent vegetation • lack of agency coordination and response • hazardous / toxic contamination

Salinity emerged as the most appropriate index for several reasons, namely; (1) salinity distribution is of direct ecological importance to many species, (2) salinity distribution is the result of the interplay of freshwater input, geometry of the estuarine basin, tidal regime and diversion of freshwater in the Delta , and (3) salinity measurements can be taken accurately, directly and economically (San Francisco Estuary Project 1993). Empirical statistical relationships between the position of the near-bottom 2‰ isohaline and various estuarine components have been developed, and show that the further downstream the 2‰ isohaline is displaced, the greater the abundance or survival of most species examined (San Francisco Estuary Project 1993; Powell 1995).

As a consequence the San Francisco Estuary Project (1993) recommended that standards should be developed using an index that establishes an upstream limit of the 2‰ near-bottom isohaline, averaged over different periods of the year, and furthermore that the downstream position of the 2‰ isohaline should not be constrained. The position of the 2‰ isohaline is considered to be a powerful diagnostic indicator of the condition of ecosystem components across a range of different trophic levels, and a sensitive indicator of the estuarine communities response to freshwater inflow (San Francisco Estuary Project 1993; Powell 1995). Moreover preliminary analyses indicated that errors in prediction using models which incorporate only the position of the 2‰ isohaline were comparable to errors using more complex models with additional flow related variables (San Francisco Estuary Project 1993).

However recognising that estuarine structure and functioning are not simply a function of the instantaneous salinity distribution, the San Francisco Estuary Project (1993) identified the ultimate goal of a predictive model which incorporates the position of the 2‰ isohaline and other appropriate physical and biological variables. The recommended approach was to formulate a matrix of the existing state of knowledge of the responses of a variety of estuarine organisms and communities as well as estuarine properties and processes, as one axis and the location of the near-bottom 2‰ isohaline as the other. A preliminary list of potential ecosystem components is shown in Table 2.3 below (San Francisco Estuary Project 1993).

Table 2.3 : A preliminary list of diagnostic estuarine properties and communities to be included in the salinity and flow-response matrices for different biologically important periods of the year (San Francisco Estuary Project 1993).

ESTUARINE PROPERTY OR COMMUNITY
Water quality for human use • Taste and odour • THM content • Salinity
Bathymetry changes
Hydrodynamic processes • Transport/circulation • Structure • Bay-ocean exchange • Residence times
Habitat area and volume
Suspended sediment dynamics
Water properties • Light availability • Temperature • Salinity distribution • Nutrient distributions
Fates and effects of toxins
Algal biomass, primary productivity, species • Bay • Delta
Nuisance blooms • *Macroalgal* • *Microalgal*
Organic carbon as food
Planktonic / neritic crustaceans • Copepods and mysids
Fish abundance • Estuarine residents • Estuarine spawners • Anadromous species • Euryhaline estuarine species • Euryhaline marine species
Benthic faunal abundance
Invasion likelihood, success
Marsh and mudflat communities • Plant species • Mammal species • Migrating, transient, and resident waterfowl; shorebirds, raptors, and passerine species • Amphibian and reptilian species • Invertebrate species as prey

The Bay-Delta Accord sets out a framework for the current management of freshwater inflow to the estuarine system and includes components of real-time management, consensus building and adaptive management (Okamoto 1995). The latter approach provides increased flexibility in water resource management in that rather than imposing a fixed standard and continuous limit on, for example, export pumping, water managers are given an annual budget and biological goals (e.g. to double the salmon population). Thus while salmon are present in the system, pumping may drop below previously mandated limits, whereas when salmon are not present pumping may exceed diversion limits.

In addition to the research conducted within the National Estuary Programme, the US Environmental Protection Agency (1993) is conducting a regional survey of the condition of estuaries (EMAP Estuaries). The survey, due for completion in 1997, assesses water quality, marine debris, water quality, fish health, benthic health and sediment quality.

2.1.3 Estuary freshwater reductions in other countries

As is the case with Australia and the United States, problems associated with reductions in freshwater flow to estuaries have emerged in other regions of the globe. For example as noted by Petts (1984), increased penetration of salt water due to reduced freshwater flow has been reported for the Zambezi estuary in Mozambique (Hall, Valente & Davies 1977), as well as the Dnieper estuary in the former Soviet Union (Zalumi 1970). Rozengurt (1991) suggests that dam construction in the southern regions of the former Soviet Union has resulted in a reduction in inflow to the estuaries and nursery areas of the Azov, Caspian and Black Seas of between 30% and 97%. Brackish water ecosystems associated with the Black Sea have been lost due to significant changes in the hydrology of the Danube, Dnepr, Dnestr, Don and Kuban rivers, resulting in a decrease of 90% to 98% in the catch of commercially viable species (Rozengurt 1991).

Construction of impoundments has resulted in a significant change in coastal hydrographic and circulation patterns associated with the Nile delta in Egypt and the St Lawrence estuary in Canada (Petts 1984). In the case of the latter, the seasonal variation in the halocline, the seasonal temperature budget of both the estuary and nearshore zone, as well as nutrient supply and marine fish reproduction have been effected (Neu 1975; Petts 1984). In the former, sediment supply to the Nile was reduced from approximately 150 x $10^6 t.yr^{-1}$ to almost nil, causing shoreline erosion and resulting loss of habitat and an 80 percent reduction in fish harvest subsequent to the construction of the Aswan High Dam. Similar problems are reported for the Indus River in southern Asia, as well as some large African rivers (Halim 1990). Frempong (1995) has noted a decrease in fisheries in the coastal lagoons associated with the Volta delta in Ghana, attributable to the construction of the Akosombo dam on the Volta river.

Fennessy, Forbes, Schleyer, Fielding and Robertson (1997) recently reviewed the effects of river regulation on prawn fisheries. These authors noted that Da Silva (1985), Gammelsrod

(1992) and Hoguane (1997) have all emphasised the importance of outflow from the Zambezi in maintaining prawn populations on the Sofala Banks off central Mozambique. Correlation between river flow patterns and prawn catches have been observed, and it has been shown that flow regulation by Cahora Bassa dam has had a distinguishable, negative effect on both prawn abundance and the size structure of prawn catches. Hoguane (1997) however, predicts that managing water release timing appropriately could increase off shore prawn population by 20%. Fennessy *et al.* (1997) raise similar concerns regarding the proposed construction of impoundments on the Thukela river, South Africa, suggesting that prawn fisheries on the Thukela Banks, are likely to be adversely impacted.

In an assessment of the 45 countries listed in the University of Rhode Island's Coastal Resource Centre (CRC) Database of Coastal Management, as at November 1996, 12 nations were found to have coastal management programmes which address reduction in freshwater inflow in some way. These nations are Albania, Australia, China (Jiangsu Province), Comoros, Ecuador, Mexico, Mozambique, Oman, Portugal, Thailand and the United States.

2.2 A brief history of estuarine freshwater requirements in South Africa

(i) Commission of Enquiry into the threat to St Lucia (1964 - 1966)

Acknowledgement of the freshwater requirements of estuaries in South African can be traced to the St Lucia Commission of Enquiry (Kriel 1966), set up to investigate the *"alleged threat to plant and animal life in St Lucia Lake*" (Crass 1982). As a consequence of reduced inflow under drought conditions, salinity in the lake during the late 1950's and early 1960's had risen to the extent that public concern resulted in the matter being raised in Parliament (Crass 1982).

The St Lucia Commission of Enquiry (Kriel 1966) recognised two primary needs, namely the supply of sufficient freshwater to the Lake and the maintenance of an adequate link with the sea. More specifically, the Commission recommended that 35% of mean annual runoff be reserved for the Lake from any storage scheme that might be built in the catchment, and that additional water be transferred from the Pongola catchment. With regard to the maintenance of an adequate link with the sea, it was recommended that the dimensions of the estuary channel be doubled (Crass 1982).

(ii) Commission of Enquiry into Water Matters (1970)

Although the Commission of Enquiry into Water Matters (1970) recommended that in planning water resources development, provision be made for the reasonable needs of nature conservation, it was stressed that a thorough study would be required to ensure that water would not be wasted (Department of Water Affairs 1986). The only two 'nature conservation' areas which were identified were the Kruger National Park and Lake St Lucia, with a total water requirement of 220 x 10^6 m^3 per year, extending to 290 x 10^6 m^3 per year by the turn of the century, representing one percent of the then estimated total water requirement of South Africa (Department of Water Affairs 1986).

During the late 1960's, McHugh (1968) expressed concern regarding the consequences of diminished freshwater flow to estuaries along the Gulf of Mexico, stating that '*reduced runoff could be the most crippling blow*" to estuarine systems. Similarly in South Africa, the role of freshwater in the conservation of estuaries was stressed by Grindley (1970), at the 1970 Water Year Convention on Water for the Future.

(iii) First attempts at quantifying estuary freshwater requirements

The concept of allocating a proportion of runoff from all catchments to estuary management was first introduced to the engineering and water resource development community in 1983 (Roberts 1983). In projecting future national water demand, Roberts (1983) made allowance for the requirements of estuaries, wetlands and nature conservation, suggesting a figure amounting to 11% of the estimated total water requirements of all sectors in the year 2000. Acknowledging that the initial estimate was simplistic, Roberts (1983) urged engineers and scientists to undertake research to refine his estimate (King & Tharme 1993).

The first regional assessment of estuary freshwater requirements was produced three years later (Jezewski & Roberts 1986). These estimates were based on two separate components; a flooding requirement and an evaporative requirement. The former was considered necessary to open temporarily closed estuaries, flood wetlands and to flush out accumulated sediment, while the latter was considered important to counter the loss due to evaporation, thereby preventing the occurrence of hypersalinity in the estuary (Whitfield & Wooldridge 1994). Estimation of the flooding requirement was almost four times the evaporative requirement and was considered to be less accurate than the estimate of the evaporative requirement, leading the Department of Water Affairs (1986) to recommend that future research would have to concentrate on refining these estimates in particular.

Rather than being annual allocations of freshwater to estuaries, these estimates were compiled to provide better predictions of runoff available for utilisation and were incorporated into the review of South African water resources (Department of Water Affairs 1986). According to the Department of Water Affairs (1986), the then estimated total freshwater requirement of estuaries and lakes (Table 2.4) amounted to 5 % of the virgin MAR of rivers, and was anticipated to represent as much as 15 % of the utilisable resource. Total water required for environmental management (including nature conservation) was estimated to reach 2 954 x 10^6 m^3 per year by the year 2000, comprising approximately 13 % of the total demand, and representing almost 10 times the amount originally estimated by the Commission on Water Affairs in 1970.

Table 2.4 : Estimated water requirements of estuaries and lakes in South Africa and the former homeland areas (Department of Water Affairs 1986)

COASTAL REGION	REQUIREMENTS (million $m^3.a^{-1}$)					
	Evaporative		Flooding		Total	
	Quantity	%	Quantity	%	Quantity	%
Cape	150	37	1080	64	1230	58
Natal	260	63	620	36	880	42
Total (RSA)	410	100	1700	100	2110	100
Transkei	190		470		660	
Ciskei	6		14		20	
Grand total	**606**		**2184**		**2790**	

In recognising estuaries as valid 'users' of water the Department of Water Affairs (1986) also raised the interesting contention that not all estuaries are equally important, arguing that "*it may be found that certain estuaries have little ecological value and enjoy low priority when water is scarce, whilst others would be regarded as being so important that they would be allocated water in almost any circumstances.*" Furthermore the Department of Water Affairs (1986) recognised that the requirements indicated in Table 2.4 are in competition with other demands, suggesting that it may not be possible to meet the management demand of each estuary and that "*benefits would have to be weighed carefully against water use in each case*".

(iv) Improved estimates of estuary freshwater requirements

During the early 1990's the Department of Water Affairs and Forestry reaffirmed its commitment to establishing the water requirements of the environment (Department of Water Affairs and Forestry 1991a) and commissioned a number of studies to more accurately determine the freshwater requirements of certain estuaries (e.g CSIR 1992a, 1992b, 1992c). The principal methodology utilised in these approaches was to convene a workshop of interested parties, and particularly experts, in various aspects of estuarine dynamics and ecology. Through a discussion of issues, participants would aim to converge on a preliminary estimate, providing recommendations in the form shown in Table 2.5 below.

Table 2.5: Summary of the conclusions and recommendations of a workshop to determine the freshwater requirements of three KwaZulu-Natal estuaries (CSIR 1992b)

	MDLOTI ESTUARY	LOVU ESTUARY	MKOMAZI ESTUARY
Classification	Blind estuary	Periodically blind estuary	Permanently open system
Management objectives	1. Seasonally open mouth (approx. 25% open) 2. Flushed regularly	1. Seasonally open mouth (approx. 80% open) 2. Flushed regularly	Permanently open mouth (approx. 100% open)
Key environmental factors	1. Maintenance of character (nursery habitat) 2. Socio-economics (water quality is important)	1. Maintenance of character (nursery habitat) 2. Socio-economics (water quality is important)	1. Maintenance of character: *importance in regional* context is unkown. 2. Water level (upstream salinities)
First estimate of the freshwater requirement	10 to 20 million m^3/yr	35 to 45 million m^3/yr	220 to 330 million m^3/yr
Confidence level	65%	55%	50%
Further requirements	1. IEM 2. Baseline survey 3. Basic monitoring programme	1. IEM 2. Baseline survey 3. Basic monitoring programme	1. Baseline survey 2. Basic monitoring programme 3. Research into regional importance

During this period pioneering research on the response of estuarine macrophytes to freshwater was being undertaken (Adams 1992; Adams, Knoop & Bate 1992; Adams & Talbot 1992), culminating in an assessment of the importance of freshwater in the maintenance of estuarine plants (Adams 1994) and a decision support system for determining the freshwater requirements of estuarine plants (Adams & Bate 1994). Through funding made available by the Water Research Commission, the work leading to the methodologies outlined in this thesis was also initiated.

(v) Current South African approaches to the management of freshwater inflow

The current approach to the management of freshwater inflow to South African estuaries is best illustrated by an examination of the Great Brak estuary, arguably the most successful example of estuary management in South Africa. The Great Brak estuary is located approximately halfway between Mossel Bay and George on the southern Cape coast (Figure 2.4). The estuary has an area of approximately 79 ha while the total catchment of the Great Brak river extends to 200 km^2 (Figure 2.5), with an average annual runoff of approximately $37 \times 10^6 m^3$ per annum (CSIR 1990). The Wolwedans Dam was completed in 1989, and was constructed with the main

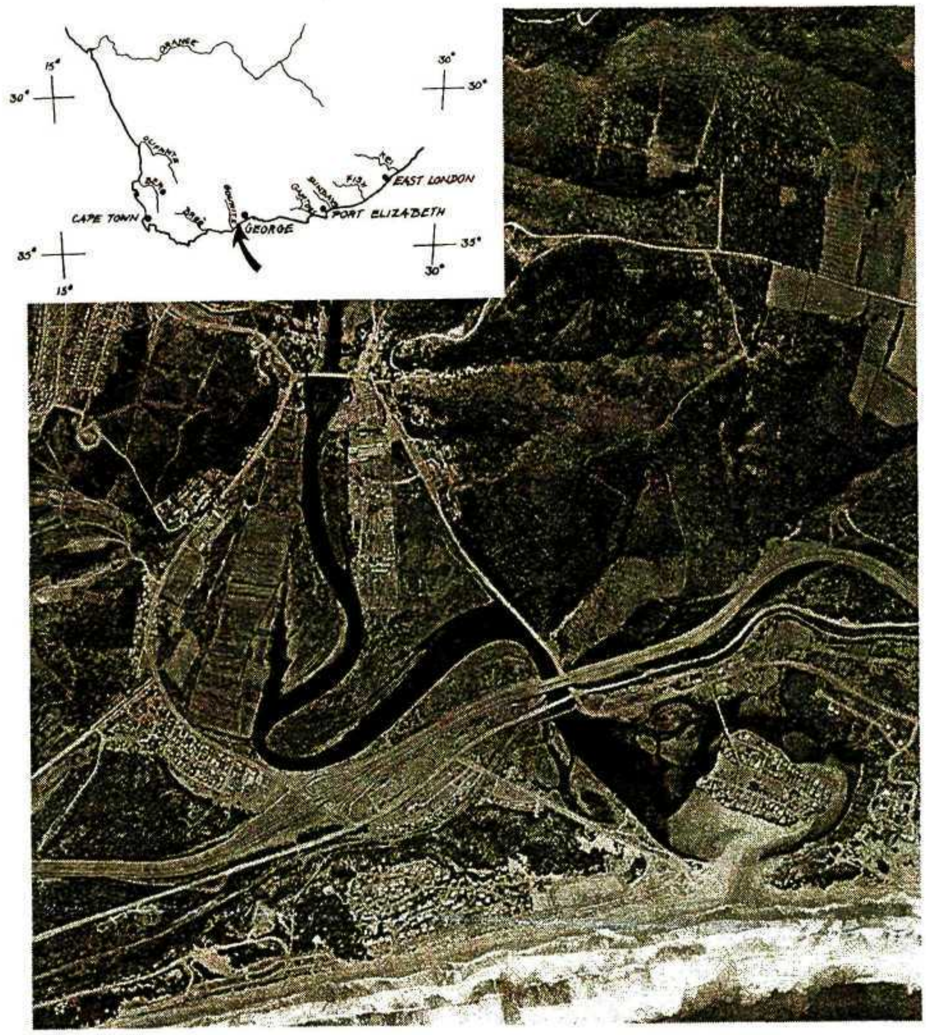


Figure 2.4 : Aerial photograph of the Great Brak estuary (CSIR 1990). Inset shows location of the Great Brak estuary (Morant 1983).

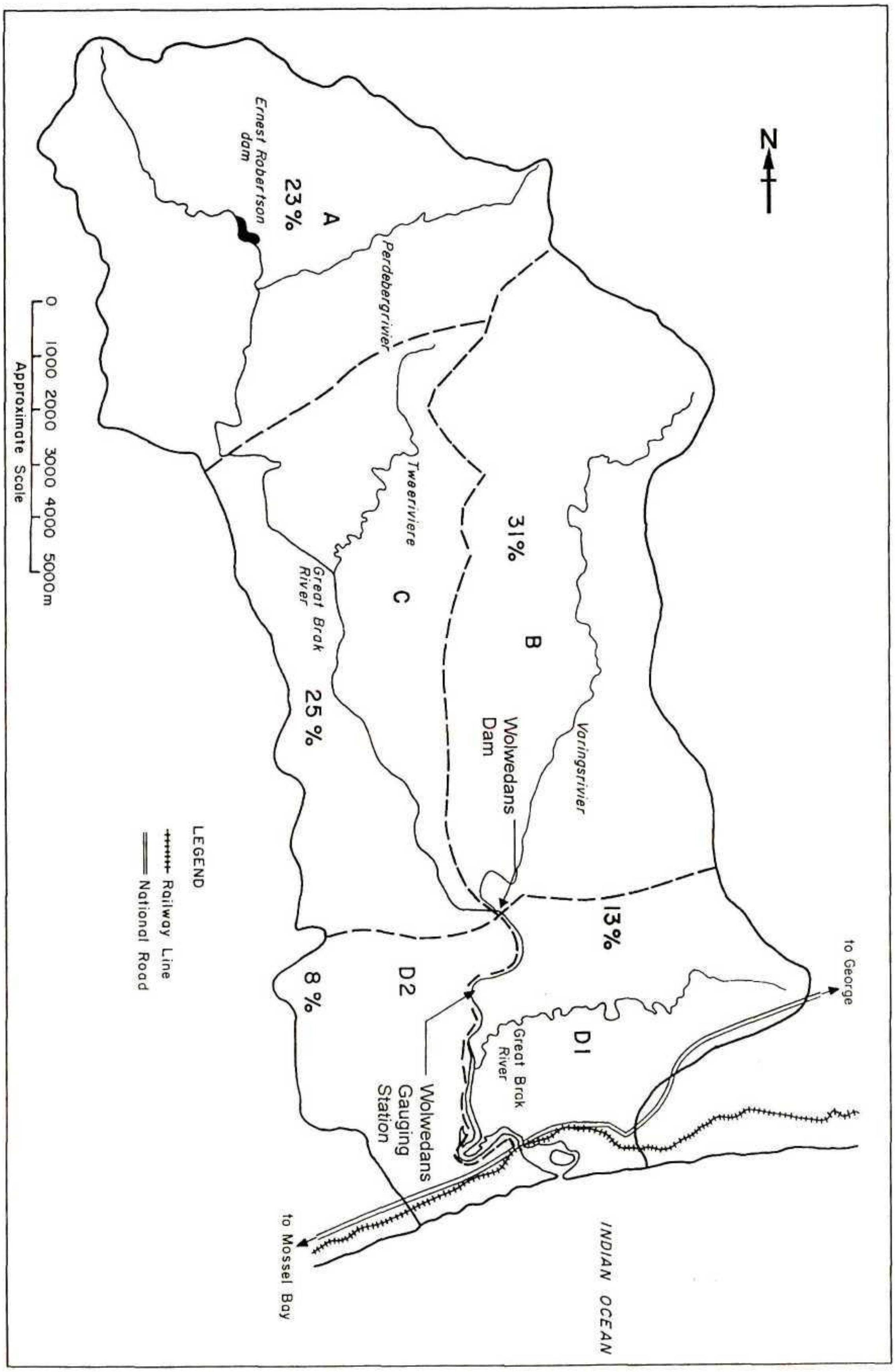


Figure 2.5 : Catchment of the Great Brak estuary showing rivers and the approximate location of the Wolwedans Dam (*after* Morant 1983).

purpose of supplying water to the Mossgass plant[1], and to supplement the growing water demand for the Mossel Bay region. Prior to construction, the water resources of the Great Brak river were largely unexploited, while currently 65% of the mean annual runoff is impounded (CSIR 1990).

According to Morant (1983), even prior to the construction of the Wolwedans Dam, normal flow to the estuary had been significantly reduced by afforestation, the damming of small tributaries for irrigation, and the building of larger dams such as the Ernest Robertson Dam (Figure 2.5). For example, along the Varing river alone there are six irrigation dams (Morant 1983). Afforestation was initiated as early as 1911, and approximately one third of the catchment is currently afforested. The cumulative effects of changing land use and streamflow attenuation by dam construction are shown in Figure 2.6 (CSIR 1990).

Subsequent to the announcement that the Wolwedans Dam was going to be constructed the Department of Water Affairs established a steering committee (Great Brak River Environment Committee (GEC)), with the specific aim of investigating the effect of the dam on the Great Brak estuary, and to establish an effective estuarine management plan for optimal use of the the water allocated for estuary purposes (CSIR 1990). In providing an allocation to the estuary a preliminary report by the Department of Water Affairs (DWA 1988/89), indicated that;

- an amount of 1×10^6 $m^3.yr^{-1}$ would be sufficient for the estuary,
- any further needs would be met during normal and above normal (wet) years when occasional overflow of the dam would occur,
- during dry years only the evaporative losses of the estuary would need to be met, and
- the capacity of the outlet works would provide the possibility of flushing the mouth of the estuary from time to time (CSIR 1990).

The subsequent investigation comprised studies of estuarine hydrodynamics, water quality and sedimentation as well as elements of the estuarine ecosystem including the distribution of vegetation and benthic fauna. The study on estuarine hydrodynamics concluded that the estuary mouth would be closed more often and for longer periods, resulting in higher water levels in

1 The Mossgass plant converts natural gas into fuel from gas reserves located below the Indian Ocean seabed, approximately 85km off-shore from Mossel Bay (Department of Water Affairs 1991b)

the estuary which would be subject to higher evaporative and seepage loss (CSIR 1990). The ecological study (CSIR 1990) concluded that;

- the ecosystem is in a fairly stable condition
- an open mouth is the most beneficial factor for maintaining estuarine ecology
- salinity in the estuary should be maintained between 7 and 40 g.kg^{-1}
- water levels in excess of 1.22 m (MSL) for extended periods should be avoided.

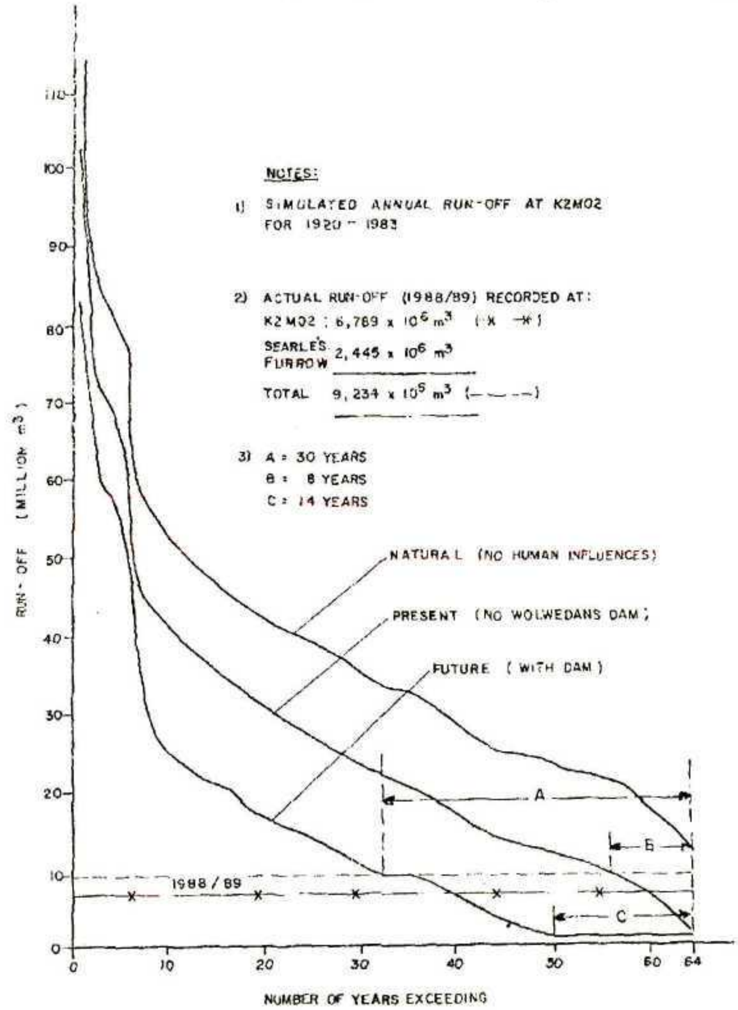


Figure 2.6 : Exceedance curve based on simulated annual run-off for the period 1920 to 1983. The three curves indicate the relative effects of land use change as well as the attenuation effects of the Wolwedans Dam (CSIR 1990).

Water quality was expected to decline due to less dilution and reduced flushing, leading to eutrophication and periodic oxygen depletion. However this assessment was based on the lack of an effective sewage system in the town of Great Brak, and consequently it was recommended that a new sewage system be constructed. A socio-economic assessment was also undertaken (CSIR 1990), and anticipated socio-economic impacts of the dam were investigated by means of a questionnaire circulated to residents. Table 2.6 shows a comparative assessment of possible socio-economic effects with and without the Wolwedans Dam (CSIR 1990).

Table 2.6 : Comparative assessment of the possible socio-economic effects with and without the Wolwedans Dam (CSIR 1990).

AREA OF CONCERN				SOLUTION	
CONCERN	PRE-DAM CONSTRUCTION	POST-DAM CONSTRUCTION	EFFECT OF DAM	REMEDIAL MEASURES	MONITORING REQUIREMENT
Reduced river flow to estuary	Gross MAR = 38.3 Mm^3 Net MAR = 15.2 Mm^3~~	Gross MAR = 38.3 Mm3 Net MAR = 15.2 Mm^3	Dam will reduce inflow to estuary	Water release plan Mouth management plan	Water level recording River flow recording
Effect on estuary mouth	Artificially breached 2 to 3 times per year due to natural high estuary water level	Reduced inflow, lower level, fewer breachings due to less river inflow	Estuary mouth will close more often and be closed for longer periods	Water release plan Mouth management plan	Water level recording River flow recording Continuous observations (Checklist)
Effect on Water quality of the estuary	Some natural stress, especially during closed mouth conditions and at high season	Increased risk of oxygen depletion, excess nutrients and high bacterial counts	Possible water quality deterioration due to reduced river inflow	Water release plan Mouth management plan	Water quality monitoring
Effect on aesthetic quality of estuary	Frequent open mouth and tidal action, especially during holiday seasons	reduced aesthetic quality due to more frequent mouth closure	Frequent mouth closures and lower estuary water levels	Water release plan Mouth management plan	Water quality monitoring water level recording
Effect on recreational value of estuary	Recreational activities concentrated around open mouth and tidal action	Activities will be influenced by mouth condition and estuary water level	Frequent mouth closure and lower estuary water levels	Water release plan Mouth management plan	Water quality monitoring Water level recording
Effect on biological aspects of estuary	Estuarine ecology appears healthy and productive	Water level change may cause salt marsh die-back and disrupt faunal production	Productivity may reduce and disrupt ecological processes	Water release plan Mouth management plan	Estuarine ecology monitoring Water level recording
Effect on property values	Valuable for holiday and retirement homes, especially on riverside and the Island	Sensitive to changes in aesthetic and recreational values	Expected trend not clear	Maintenance of aesthetic and recreational values as far as possible	
Effect on flooding	Major floods with loss of property have been recorded	Normal floods will be attenuated	Risk of damage by floods is reduced	Issuance of flood warning Evacuation plan	Estuary water level River flow Dam level
Effect on water rights	Industry uses 4 Mm^3/a but owns rights to 10.6 Mm3/a. Riparian farmers use 6.3 Mm^3/a~	At present utilised rights are acknowledged	Farmers are limited to a maximum for irrigation	Industry may be entitled to compensation	
Socio-psychological effect on the downstream residents	No effect without dam	Concern at the possible effect of dam failure and security, and effect on the river	Some downstream residents feel insecure as result of dam	Involve community in flood warning evacuation plan and in devising management plan	

~ Mossref acts as an agent for the DWA and operates the dam

~~ 1 Mm^3 = 1 million m^3

The primary recommendation of the CSIR (1990) was that *"the present estuarine environment be maintained by substituting active management for the natural processes operative prior to the construction of the dam. This implies that the natural effect of unregulated river flow should be replaced by controlled water releases, together with the excavation of the beach berm, to open the estuary mouth when required".*

It was also recommended that in order to determine when water releases or breaching would be necessary a monitoring procedure should be established. Monitoring should be undertaken on a monthly basis during the off-season period, but weekly during the holiday season. At each sampling interval a checklist would be filled in and scored according to the criteria shown in Table 2.7 below. The flowchart shown in Figure 2.7 would then be used to determine what management action should be taken (CSIR 1990).

Table 2.7 : Checklist and scoring criteria for determining whether management action is necessary (CSIR 1990)

CRITERIA	SCORE	
	YES	NO
Is the mouth open?	2	0
Is the estuary water level less than +1.22 MSL	2	0
Is there a bad smell and / or excessive algal growth in the water?	0	1
Is the E. coli level less than 1000?	2	0
Is the salinity level more than 7 and less than 40?	2	0
Are fish dying or under stress e.g. gaping at the surface for air?	0	2
Is it February? Is it June? Is it November?	-1 -1 -2	0 0 0

A monitoring programme was also recommended, with the purpose of evaluating the proposed management approach. Key aspects of the monitoring programme included; continuous monitoring of river inflow and water level in the estuary, an annual bathymetric survey of the lower estuary for at least the initial 10 years, and monitoring of plant communities as well as species diversity and abundance of the fauna along baseline transects on an annual basis, for a similar period. The monitoring programme has been running for several years now and three major monitoring reports have been produced (CSIR 1992d; CSIR 1993; CSIR 1994).

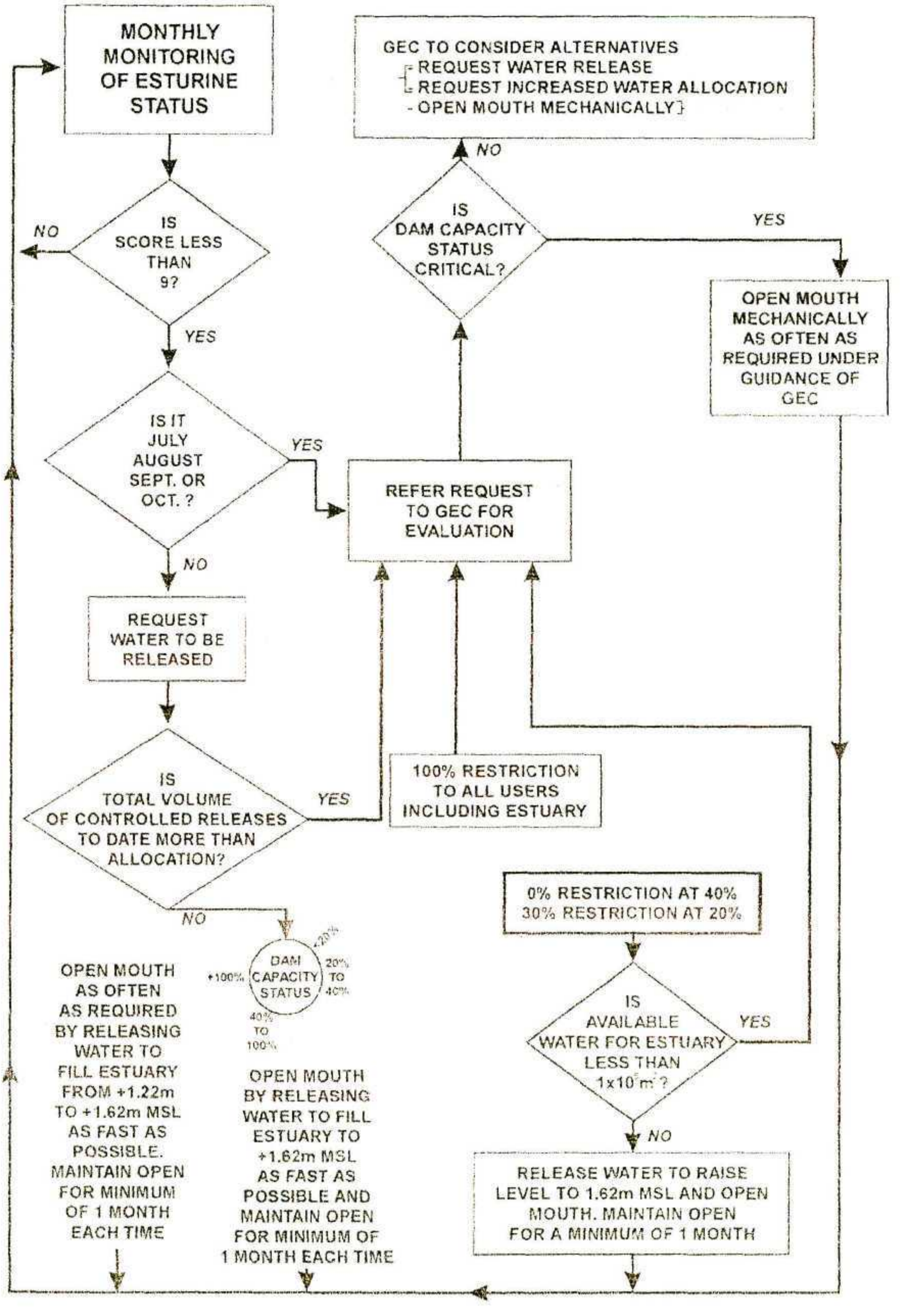


Figure 2.7 : Flow diagram to be used in association with monitoring checklist to determine what management action should be undertaken (CSIR 1990)

2.3 Discussion

Over a period of twenty-five years, the effects of reducing freshwater inflow to estuaries and coastal ecosystems have been documented for locations in both developed and developing countries. As a consequence, reduction in flow to estuaries is now acknowledged as a global rather than a regional threat to biodiversity (Norse 1993).

Although many countries have recently begun to recognise the water requirements of the natural environment, the emphasis in doing so has certainly been the determination and allocation of instream flow requirements for rivers and associated wetland and riparian environments. Over the last few decades considerable effort has been focussed on the development of methodologies to determine the instream freshwater requirements of these inland aquatic systems, and consequently a wide range of techniques for impact assessment and flow requirement assessment are now available (Bovee 1982; Gore & Nestler 1988; Milhous, Updike & Schneider 1989; King & Tharme 1993). In contrast, very little effort has been directed at the development of similar techniques for determining the water requirements of estuaries. A notable exception has been the formulation of approaches for the management of freshwater flows to the estuaries serving as nursery grounds for the commercially important coastal fisheries the Gulf of Mexico. Similarly, management of freshwater flow in the San Francisco Bay - Delta area has also received attention.

This review has identified that in addition to the acknowledged need for the development of techniques for estimating the freshwater requirements of South African estuaries, there is an urgent need to develop a general approach which may be applicable in other regions, particularly those of the developing nations. This thesis will present such an approach, which although formulated for South African conditions, may nevertheless be appropriate in other areas. Prior to setting out a conceptual framework for the management of South African estuaries several key issues arising out of the preceding literature review will be discussed, with a view to identifying the critical requirements for the framework.

(i) Shortcomings of previous South African approaches

As noted by others (Whitfield & Bruton 1989; Whitfield & Wooldridge 1994), the regional allocations determined by Jezewski and Roberts (1986) did not take into account the numerous biotic-abiotic relationships within estuaries, focussing instead on an estimate of the flooding and evaporative requirements. Indeed, these estimates were not intended as allocations, but merely approximations which would provide water managers with a more realistic estimate of the utilisable runoff from catchments. Thus while the determination of these first estimates was rooted in recognition of the water needs of the environment, it was not an assessment of the *ecological* water requirements of estuaries. In addition, the Jezewski and Roberts (1986) estimates did not provide an indication in the seasonal or monthly distribution of this annual allocation (Whitfield & Bruton 1989; Whitfield & Wooldridge 1994).

Recent approaches (CSIR 1992a, 1992b, 1992c, 1995), reflect the ecological water requirements of estuaries more explicitly as they are determined by consensus of specialists, whereby each specialist is an authority representing some component of the ecosystem such as fish or birds. However the approach is based on a collective and intuitive expert assessment rather than a scientifically defensible methodology, and consequently very low confidence assessments often accompany these estimates (Table 2.5). Furthermore these estimates are usually annual totals and do not specify monthly flow regimes or ranges in acceptable variability. Because explicit ecological goals are not specified, there is no way of knowing in the long term whether these estimates are realistic, and while the experts always stress the need for monitoring, it is seldom undertaken by government departments or authorities. Lack of monitoring and investment in basic research by government departments has led to a growing dissatisfaction among estuarine scientists and a reluctance to participate in estuarine flow requirement workshops. Scientists have been concerned that their initial estimates are accepted as a 'quick fix' solution tantamount to 'rubber stamping' without providing them the opportunity to re-evaluate estimates in conjunction with a structured research and monitoring programme.

Although the Great Brak estuary provides an example of where a relatively comprehensive monitoring strategy has been established it should be noted that the freshwater requirements of this estuary were allocated prior to any impact assessment, as was the decision to construct the dam. Nevertheless the management system is currently the only one of this nature in place and is widely regarded as being successful. The advantages and limitations of this management approach will be discussed more completely in subsequent chapters of this dissertation as the techniques developed herein will be tested in a case study of the Great Brak estuary.

(ii) Assessment of international approaches

Only the United States has appeared to have directed substantial research effort at formulating techniques for the determining estuary freshwater requirements. Two distinct approaches have evolved; one applicable to estuaries flowing into the Gulf of Mexico, the other an approach for the San Francisco Delta-Bay area. Although there are elements of both approaches which would be applicable to South African estuaries, there are equally several limitations.

The approach adopted for the San Francisco Delta-Bay is focussed on a single index of salinity, the upper boundary of the near bottom 2‰ isohaline. This index was developed as a proxy for freshwater inflow and integrated measure of the net effects of the inflow, due to difficulties in determining the actual inflow volumes into the San Francisco Bay-Delta area. While this index has clear value as a surrogate measure of freshwater inflow several concerns regarding its utility as a management tool for ecological systems are apparent.

This point may be best illustrated by initially referring to a South African example. In his policy proposals for the estuaries of KwaZulu-Natal, Begg (1985), recommended *'that management should seek, as far as possible, to preserve natural conditions such as natural vegetation, natural drainage patterns, the rate of freshwater discharges, and natural tidal and salinity regimes'*. Similarly, in a review of the conservation status of South African estuaries, Heydorn (1989) suggested that sound management procedures for estuarine environments should be based on maintenance of the natural processes operative in estuaries to the greatest possible extent. In essence, these policy guidelines suggest that the primary management goal is the maintenance of natural physical processes. While the value of guidelines such as these is unquestionable at the level of public policy, at a pragmatic and operational level, approaches which are based on maintaining 'normal' flow, 'typical' salinity gradients or 'characteristic' types of circulation are inherently problematic.

The rationale for these approaches very often lies in the fact that the physical characteristics of a particular estuary are determined largely by the interaction between river discharge, tidal action and basin morphology (Kennish 1986), and consequently proponents argue that since the basin morphology can be measured and both tides and river discharge can be relatively accurately predicted; it is possible to classify a particular system on the basis of the typical hydrodynamics. The objective of management in this case is to keep variation of salinity within 'normal limits', defined by a historic record or by consensus of experts. The question that needs to be addressed is; given the variability of the processes involved, how realistic, how meaningful and how useful is a management guideline which is based on 'normal' conditions? If the objective of the management policy is to maintain the hydrodynamic integrity of the system, how much variation does one permit, and what are the ecological consequences of variation? This question is particularly valid given that variability is often considered a key feature of estuarine systems. If one wishes to determine the performance of a management strategy for freshwater flow to an estuary, is it evaluated according to whether physical criteria have been met, or evaluated according to the current health of the ecosystem? If it is found that the ecosystem is in poor health due to insufficient inflow how does one determine an increase in allocation, other than by an iterative 'trial and error' approach?

It is therefore argued that approaches to the management of estuaries based on salinity variation and other hydrodynamic criteria are not particularly useful. The purpose of estuary management is not to protect and conserve salinity variation. Salinity will always vary where fresh and sea water meet, even in the absence of a single organism. Rather, it is argued that salinity variation is one of several unique features of estuaries, which together create conditions which are different from either adjacent environments. Certain organisms have become adapted to, and in certain cases dependent on the conditions found in estuaries. In this way estuaries, by way of their 'otherness' or uniqueness have developed ecological functions which sustain and compliment processes and populations in adjacent environments. It is these critical ecological functions which require protection in the management of estuaries. It is clear that investigation which is directed at the management of coastal resources needs to take an approach which is less centred on categorisation of the physical attributes of estuaries and more focussed on their functions as ecosystems. Although this need was recognised by the San Francisco Estuary Project (1993), relationships between their selected index of inflow and measures of ecosystem functioning have yet to be developed.

Unlike the approach adopted by the San Francisco Estuary Project (1993), the methodology developed for Texas estuaries does provide a direct link between inflow and measures of ecosystem functioning. Because of the substantial commercial fisheries present in the Gulf region, extensive time series of harvest amounts are available. As a consequence it has been relatively easy to determine site-specific statistical relationships between harvesting (representing recruitment success and therefore ecosystem functioning) and inflow. These relationships, represented as continually revised regression equations provide the basis for the methodology proposed for the region. The use of regression equations in a management approach has its own problems, as articulated by Tung *et al.* (1990), "*...the causal connection between flow and harvest may be obscured by unmeasurable parameters such as effort, selectivity, and skill, and may be corrupted by poor reporting or the difference between locality of landing and locality of catch, to say nothing of other environmental variables unrelated to inflow. The regression therefore tends to be noisy and statistically uncertain*". Nevertheless, the primary reason why this approach is of limited applicability in South Africa is that there are no comparable time series data. Commercial fisheries in South Africa are based primarily on pelagic species which are not dependant on estuaries.

(iii) Considerations specific to South Africa

Unlike the majority of estuaries elsewhere in the world, a high proportion of estuaries in South Africa are frequently closed to the sea for extended periods during the year. This arises due to the fact that catchments along the eastern and southern coasts are small and this together with the low rainfall of the region results in low runoff volumes delivered to estuaries. As discussed previously, abstraction and storage further reduce runoff volumes and consequently runoff is insufficient to maintain an open mouth. While this may occur in other regions of the world, it does not appear to have been raised as a major regional or national concern anywhere else. However the increase in frequency of mouth closure in South Africa does lead one to speculate that it could be a phenomenon more frequently encountered in other regions in the future, particularly those with low outflow systems, characterised by increasing levels of abstraction. The implication of this is that any freshwater determination methodology proposed for South Africa needs to address the potential for estuary mouth closure under highly variable and reduced inflow conditions.

Furthermore South Africa is characterised by situation of poor data availability. Very few estuaries in South Africa have flow gauging weirs near their mouths and even fewer have water level gauges. Only one set of comparable estuary health assessments has been undertaken for all estuaries in the country, and in addition, these assessments were not undertaken during the same time period (Cooper, Harrison, Ramm & Singh 1993; Harrison, Cooper, Ramm & Singh 1994; Harrison, Cooper, Ramm & Singh 1995). Whitfield (1995) has analysed the availability of information on South African estuaries (Table 2.8), showing that information availability could be considered poor or nil for 68% of estuaries. Fortunately whatever information is available has been compiled into a bibliography of information for each individual estuary (Whitfield 1995), which is updated on a continuous basis.

Table 2.8 : Synthesis of available information categories from individual estuaries in each biogeographical region (Whitfield 1995)

INFORMATION	REGION							
	Cool Temperate		Warm Temperate		Subtropical		South Africa	
	No.	%	No.	%	No.	%	No.	%
Nil	-	-	61	50	31	26	92	37
Poor	5	50	21	17	52	44	78	31
Moderate	3	30	23	19	28	24	54	22
Good	2	20	10	8	5	4	17	7
Excellent	-	-	7	6	2	2	9	3
Total	**10**	**100**	**122**	**100**	**118**	**100**	**250**	**100**

2.4 Redefining management needs

Over the past 25 years some effort has been directed toward characterising estuarine systems in South Africa and identifying problems and threats to their sustainable utilisation. For a number of years, researchers, managers and policy makers have been aware of the value of estuarine systems and cognisant of their role and value as ecosystems. However, although policy proposals for certain estuaries were tabled over a decade ago (Begg 1985), these have yet to be accepted or implemented. Similarly, although the need for estuary-specific freshwater allocations has been acknowledged for several years, an integrated methodology for determining these has not been developed.

This thesis proposes that objectives such as 'the sustainable utilisation of estuaries' or 'maintenance of biological diversity' will not be achieved on the basis of the formulation of policy alone. Indeed the failure of most policy initiatives can be attributed to formulation in the absence of a strategy for implementation, and particularly, the absence of a supportive and integrated management framework. Policy which is to be successful needs to be accompanied by a strategy which sets out a time table for the action plan, describing clearly the operational goals against which progress is going to be measured. The strategy should document the range of methods which will be employed to achieve these goals, and may include a mix of regulatory, non-regulatory and incentive based approaches.

Most importantly, successful policy also requires a 'toolbox' of decision making and evaluatory techniques. The purpose of natural resource policy is to set the constraints and conditions of resource utilisation, and as can be expected, as the demand for natural capital such as water or land increases, so do the complexities of resource management. Resource management in the latter half of this century has consequently been characterised by dissension between social, economic and political issues. Because of the dynamic nature of these interactions and their non-linear responses to change, consequences of potential actions are difficult to predict, thereby creating uncertainty and promoting conjecture. Under these circumstances, decisions are difficult to make, difficult to explain and difficult to defend. The result is often a debilitating, unproductive debate which stifles progress and wastes resources. Structured decision making based on predictive capacity can provide a significant contribution to this process by constraining the debate and focussing management on the real issues.

A central hypothesis of this thesis is that the development of appropriate predictive capacity, located within a framework which acknowledges the failings of previous approaches to adaptive management (McLain & Lee 1996), will contribute significantly to defining a management approach which will demonstrate achievable ecological sustainability. Breen, Quinn and Deacon (1994) have proposed a management approach for the natural environment of rivers which addresses many of these concerns (Figure 2.8). The concepts employed in this approach have been extended in formulating an approach appropriate for the management of estuaries.

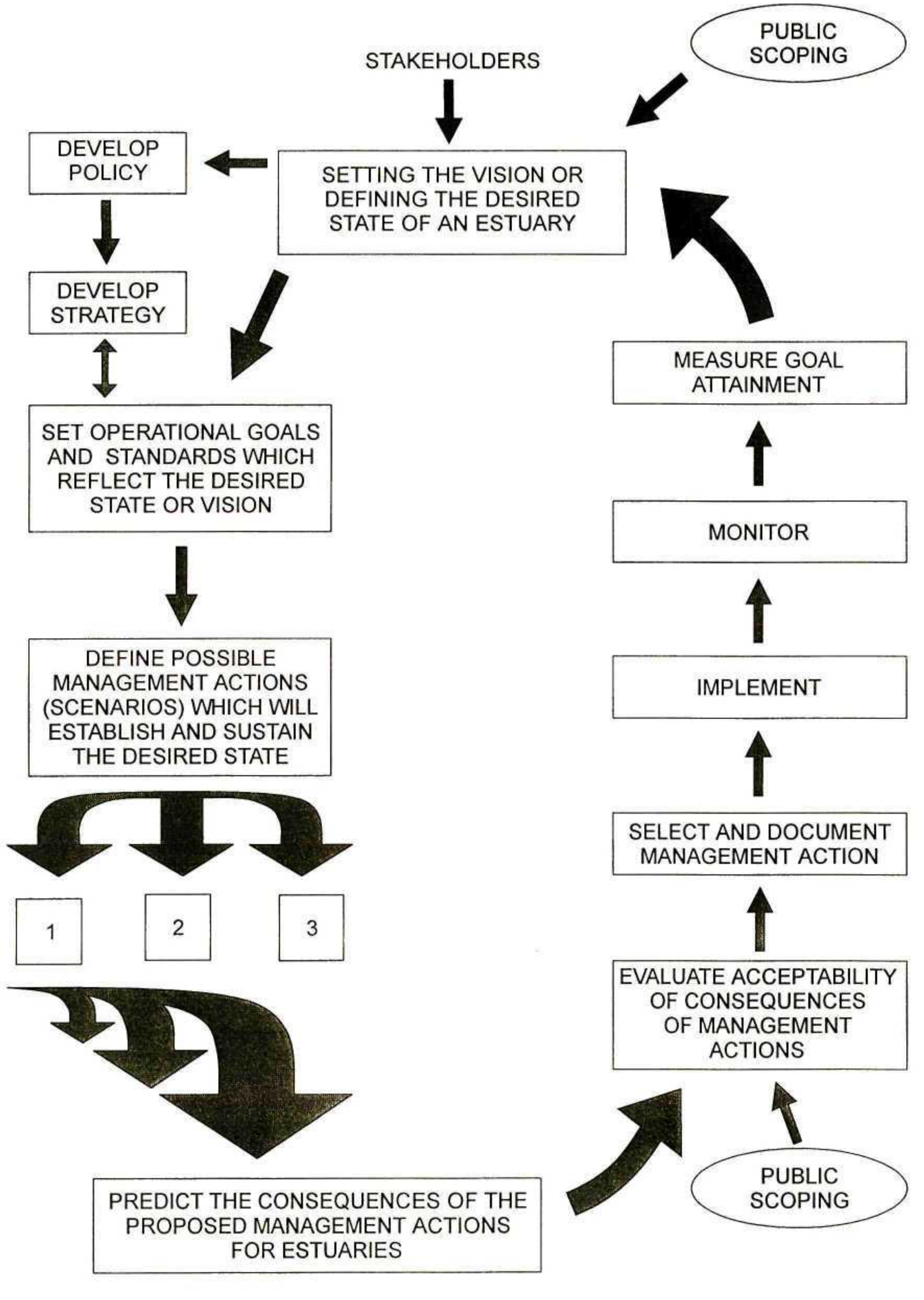


Figure 2.8 : A management framework for demonstrating ecological sustainability in estuaries (*after* Breen, Quinn & Deacon 1994)

(i) Setting the vision

A vision describes the state in which society wishes estuaries to be, reflecting the value society places on estuaries and their functions. For management to be meaningful its purpose must be to achieve a desired and predetermined state of the natural environment (Breen, Quinn & Deacon 1994). The vision in turn, forms the basis for policy. For example the discussion document for marine environmental conservation policy in KwaZulu-Natal states that '*the province of KwaZulu-Natal will, in the 21st century, have healthy coastal and marine ecosystems that will be rich in biodiversity and provide the people of this province with safe and sustainable opportunities for economic development, healthy food and quality recreation.*"(Anon 1996).

As the vision is a reflection of the wishes of society, it is important that setting the vision, as well as many other components of this management approach, occurs on the basis of public participation. Griffis and Kimball (1996) suggest that the "*sheer complexity of activities and institutions, and the sheer density of people in coastal areas, makes the people-management process the focus of Integrated Coastal Management*". Participation is a fundamental principle of the new environmental policy, and recognises that "*all interested and affected parties have a right to participate in environmental management and decision making*" (Department of Environmental Affairs and Tourism 1996). This tenet is equally recognised in the new Water Law Principles which acknowledge the need to devolve management to a local level (Department of Water Affairs and Forestry 1996).

(ii) Setting operational goals

Goals such as 'maintaining ecological processes', 'preserving genetic diversity' or 'ensuring sustainable utilisation' cannot be considered operational goals as there are no direct means of assessing achievement. These goals must therefore be redefined in precise and meaningful terms (Breen, Quinn and Deacon 1994). Setting operational goals has recently been expressed as a fundamental requirement of scientifically based ecosystem management (Christensen *et al.* 1996). The report of the Ecological Society of America Committee on the Scientific Basis for Ecosystem Management (Christensen *et al.* 1996) states that "*goals must be explicitly stated in terms of specific 'desired future trajectories' and 'desired future behaviours' for the ecosystem components and processes necessary for sustainability. Furthermore, these goals* *should be stated in terms that can be measured and monitored"*. As Meyer and Swank (1996) have expressed it, *"unless we provide precise definitions and specific measures for attributes such as ecosystem integrity, health, and vitality, these will not be achievable management goals..."*.

(iii) Defining and recording potential management actions

Management actions required to sustain the natural environment are usually based on wisdom in the absence of any structured decision making framework. While decisions are considered to reflect the expertise of participants, the process, interpretations and assumptions which underlie these decisions are seldom recorded. As a consequence it is impossible to review the steps in decision making to correct, improve or reverse decisions (Breen, Quinn & Deacon 1994). All too often it is not known why a particular decision was taken or by whom. Suggestions which were offered during the decision making process as alternatives to the final decision may be lost, despite the fact that they may have yielded better results on implementation. Apart from giving attention to documenting the process of defining management actions, the actions themselves must be documented in a clear and unambiguous manner. Alternative management actions need to be explicitly linked to their desired effect in the ecosystem. At the same time the social and economic costs of each alternative can be determined to ultimately facilitate trade-offs.

(iv) Predicting the consequences of management actions

Adaptive management is currently promoted as a desirable conceptual approach to ecosystem management (Christensen *et al.* 1996), and can be described as a structured system in which monitoring iteratively improves the knowledge base and helps to refine management plans (Ringold *et al.* 1996). Adaptive management uses experimentation as a strategy for managing ecosystems, whereby the results of studies of prior management decisions are incorporated into future management decisions (Okamoto 1995). Others have criticised the adaptive management approach (McLain & Lee 1996), claiming in a case study of three examples, that the approach is unsuccessful because it *"relies excessively on the use of linear systems models, discounts nonscientific forms of knowledge, and pays inadequate attention to policy processes that promote the development of shared understandings among diverse stakeholders."* Adaptive management also carries a certain risk, based on the assumption that monitoring will identify inappropriate action and that corrective management will restore the ecosystem to its previous state. In the words of Holling (1996), " *in adaptive management, policies are designed as hypotheses and management implemented as experiments to test those hypotheses. But the rule of good experimentation is that the consequences of actions be potentially reversible and that the experimenter learns from the experiment"* .

Being able to predict the consequences of management actions will obviously limit the risk associated with the adaptive management approach. According to Christensen *et al.* (1996), models " *can be useful in identifying particularly sensitive ecosystem components or in setting brackets around expectations for the behavior of particular processes. They can be especially useful in identifying indices and indicators that provide a measure of the behavior of a broad suite of ecosystem properties*". Some would argue that for models to be useful management tools they *need* to identify indices or indicators which provide a realistic measure of ecosystem functioning. The approach proposed in this thesis argues that policy, goals and management actions should be specifically focussed on the biological attributes of estuaries, and measured by one or more objective indices. This means that the consequences of potential action such as artificial breaching, or water release policies from proposed or existing impoundments should be evaluated by indices which indicate the extent to which the proposal impacts on the biological potential of the system. By this is meant the extent to which a given estuary can fulfil the dimensions habitat defined in the Introduction.

Such an approach necessitates the development of predictive abilities at two levels. Firstly, the hydrodynamic and physical consequences to an estuary, of an action such as artificial breaching or a water release from an impoundment, need to be determined. In other words the effect of such a change on the physical environment, measured for example by changes in water level, temperature, salinity or access to the sea needs to be established. The second level of prediction requires the translation of changes in these environmental variables into consequences for the biota. For example, how changes in inflow might alter habitat availability and consequently the dynamics of populations supported by these habitats, or alternatively how a decrease in freshwater inflow together with high evaporation may result in salinities above the tolerances of organisms within the estuary.

(iv) Evaluate the acceptability of consequences of different management actions

Anthropogenically induced changes in the structure and functioning of natural systems alters the asset value ascribed to natural systems (Breen, Quinn & Deacon 1994), and the response of society to the predicted change will reflect the extent to which the associated change in asset value is considered acceptable. If the likely consequences are considered unsuitable, other management actions need to be considered and routed through the predictive process. Obviously this process is dependent on public participation and requires protocols for establishing trade-offs and documentation of the entire assessment process. As Stanford and Poole (1996) have suggested public participation requires an unambiguous, public forum capable of policy, science and management debate, characterised by intimate citizen involvement in a process which is founded upon empirical analysis of cause and effect.

(iv) Selection and implementation of management action(s)

Hilborn (1992) has raised the question of whether institutions are able to learn from experience, suggesting that they often lack systematic plans for doing so, as well as mechanisms for retaining new knowledge in the 'memory' of the institution. If the process of adaptive management is to succeed it is essential that the selected management action as well as the rationale for its selection is documented.

(v) Monitoring and measuring goal attainment

Management is meaningless unless one is able to measure the extent to which management goals are being attained. Furthermore, monitoring needs to occur at a time scale that will permit the implementation of corrective action before undesirable changes in the state of the ecosystem occur. Implementation of monitoring programmes requires vision and commitment, the benefits of which are not always evident in the short term (Christensen *et al.* 1996). Ringold *et al.* (1996) have proposed an approach to monitoring consistent with the principles of adaptive management, in that the monitoring process itself is subject to a process of adaptive management. The most fundamental requirement of a monitoring strategy, however, is that it should specifically set out to measure the *actual* responses of components of the ecosystem in relation to the *predicted* responses. This requires that monitoring outputs and modelling outputs be directly comparable.

2.5 Objectives for developing predictive capability

The predictive capabilities alluded to in the previous sections, do not currently exist. As is evident from the preceding discussion, the proposed approach is dependent on two levels of predictive ability, the first focussing on the consequences management actions may have for the physical character of estuaries and the second focussing on the consequences the resulting physical change may have for the ecological functioning of the system. As the first level of prediction has been addressed elsewhere (Slinger 1996), this dissertation will focus on developing an approach to addressing the latter.

Over the past three decades modelling has come to be acknowledged as an integral part of problem solving in almost all disciplines (Starfield, Smith & Bleloch 1990), and particularly for natural resource management in situations characterised by limited data availability and poor understanding (Holling 1978; Starfield & Bleloch 1991). Numerous modelling methods, techniques and approaches to problem solving have evolved over the years, contributing to an enormous body of knowledge, numerous failures and some successes. Evaluation of this body of knowledge is beyond the scope of this dissertation and the approaches and techniques utilised herein are not defended against a myriad of potentially suitable alternatives. In developing the necessary predictive understanding the principles of 'boot-strapping' and incrementalism (Starfield & Bleloch 1991) have been embraced; guided by the goal of exploring the consequences of our present understanding, rather than a goal of deterministic inclusivism. 'Prediction' is used in the sense of 'likely outcome' rather than a measurable absolute, and by model is meant 'an integrated suite of predictive methods' rather than a unitary mathematical construct. Above all, it should be recognised that the current management of South African estuaries is based on limited understanding and almost no data to corroborate hypotheses.

The remainder of this thesis will focus on the development of predictive capabilities which are required if the management framework outlined in this Chapter is to be adopted. The processes, functions, or components of the ecosystem selected as representative of the link between management action and ecosystem response must be selected with great care, as they will be used as surrogate evidence that the ecosystem is being managed in an ecologically sustainable manner. With this need in mind the following objectives and criteria are specified.

(i) To develop an ecologically based modelling approach which will predict the ecological consequences of changes in the physical character of the estuarine environment. This model should meet the following criteria :

- The modelling approach should be specifically tailored to predict ecological functioning of the estuary.
- The modelling approach should include a representative range of organisms, preferably reflecting a range of trophic levels.
- The modelling approach should be sensitive to changes in estuary physico-chemical condition and therefore incorporate the ecophysiological tolerances of individual species
- The modelling approach should be generally transferable to all South African estuarine systems.
- As far as possible, the modelling approach should demonstrate applicability to other regions, particularly semi-arid regions characterised by growing demand for water resources.
- The modelling approach should generate a set of management indices which should be objective, ecologically based and readily understandable to the biologist and resource manager alike. The indices should not be biassed to a single species and should be generally regarded as representative of the state of health of the estuary.
- The modelling approach should generate indices which are easily and inexpensively monitored.

(ii) To test and evaluate the modelling approach and indices in a case study.

(iii) To identify further research needs, and in particular to identify further development requirements to operationalise the modelling approach and indices developed.

3 FISH RECRUITMENT INDEX

3.1 INTRODUCTION

As noted by Whitfield (1996), fishes have frequently been used as indicators of environmental health or biological integrity (Brown 1978; Hocutt 1981; Karr 1981; Elliot, Griffiths & Taylor 1988; Ramm 1988, 1990; Fausch, Lyons, Karr & Angermeier 1990). According to Whitfield (1996), both fish abundance and species diversity provide a good indication of the health of the system under consideration. Whitfield (1996) has also summarised the advantages and limitations of using fish as indices of biological integrity, concluding that the advantages far outweigh the disadvantages.

As identified in the Introduction, one of the primary ecological values attributed to estuaries is the role they play in providing habitats for juvenile fish, often referred to as the 'nursery function' of estuaries. As the modelling objectives established in Section 2.5 call for an approach which is not biassed to a single species or trophic level, attention was given to the formulation of a modelling approach which would provide an indication of the extent to which an estuary was fulfilling this role for a variety of fish species. It should be noted that data regarding the utilisation of estuaries in South Africa by juvenile marine fish is very sparse, limited to three principal components:

(i) Firstly a variety of juvenile fish make use of estuaries, although not all of these species are dependent on estuaries as several are able to utilise inshore areas as nursery grounds. In developing the index the assumption that greater emphasis should be placed on those species with an absolute rather than a prefered or opportunistic requirement, has been made. Section 3.2.1 thus describes a dependency categorisation of fishes in South African estuaries, and an associated dependency score.

(ii) Secondly, although juvenile marine fish recruit into estuaries predominantly during the summer months, each species appears to have a prefered recruitment period, comprising a peak period and periods of lesser recruitment activity before and after the peak. Something is known of the recruitment periods for 27 fish species. Section 3.2.2 thus summarises this understanding and describes an associated recruitment score.

(iii) Finally, there is evidence to suggest that recruitment occurs to a greater extent under conditions characterised by a high longitudinal salinity difference (the difference in salinity between the mouth and head of the estuary). In contrast, recruitment appears to be limited during periods of high flow. Section 3.2.3 presents the evidence for these claims and describes several alternative assumptions for incorporation into the index.

The index described below attempts to integrate this limited information and understanding as concisely as possible, and in a manner which is readily understandable by estuary managers, biologists and laymen alike. The result is a 'hybrid' index incorporating both published data as well as expert opinion and value judgements.

3.2 COMPONENTS OF THE INDEX

3.2.1 Dependency score (DS_i)

Whitfield (1994a) provides a revised estuary dependence categorization for fishes in South African estuaries, consisting of five major categories (Table 3.1). Of these, only species in divisions Ia and IIa are totally dependent on estuaries. Categories Ib, IIb and IIIc represent species which are at least partially dependent on estuarine systems, a large proportion of which could be regarded as marine / estuarine opportunists. Category III comprises mainly stenohaline marine species which occur in low numbers in estuaries. Category IV consists of mainly euryhaline freshwater species, for which the degree of penetration into estuaries is determined primarily by salinity tolerance. Finally Category V includes obligate catadromous species such as eels, which use estuaries as transit routes between marine and freshwater environments. The total number of estuary dependent or partially dependent species corresponds to 74% of the total number of estuary associated species (Whitfield 1994a).

In order to distinguish between those species which merely benefit from estuaries, and those which are dependent on estuaries, a simple scoring system was devised. Category I species were not included as they spend their entire life cycle within the estuary and do not translocate to the marine or freshwater environments for spawning. These species will therefore breed in estuaries even if the mouth is closed. Category IV species are euryhaline freshwater species and were excluded, as they are not dependent on estuaries and will also utilise estuaries when the mouth is closed.

Table 3.1 : The five major categories of fishes which utilize South African estuaries (*after* Whitfield 1994a)

I		Estuarine species which breed in South African estuaries. Further subdivided into :
	Ia	Resident species which have not been recorded spawning in the marine or freshwater environment
	Ib	Resident species which also have marine or freshwater breeding populations
II		Euryhaline marine species which usually breed at sea with the juveniles showing varying degrees of dependence on South African estuaries. Further subdivided into :
	IIa	Juveniles dependent on estuaries as nursery areas
	IIb	Juveniles occur mainly in estuaries, but are also found at sea
	IIc	Juveniles occur in estuaries, but are usually more abundant at sea
III		Marine species which occur in estuaries in small numbers but are not dependent on these systems
IV		Euryhaline freshwater species, whose penetration into estuaries is determined primarily by salinity tolerance. Includes some species which may breed in both freshwater and estuarine systems.
V		Obligate catadromous species which use estuaries as transit routes between the marine and freshwater environments

Based on the list of species provided by Whitfield (1994a), each species is allocated a dependency score (DS_i), reflecting the extent of its dependency on the estuarine environment (Table 3.2). However, although it is relatively easy to allocate scores to the extremes of the dependency continuum (completely dependent = 5 and completely opportunistic = 1), scoring intermediate classes is associated with somewhat more uncertainty.

As a consequence is it necessary to propose alternative scoring systems to establish whether adopting a different scoring system may materially alter any policy recommendations. Thus under Assumption 1 in Table 3.2 overleaf, Category V species are scored the same as Category IIa species as they are entirely dependent on estuaries in the sense that juveniles need to move through estuaries to obtain access to their freshwater habitats in rivers and streams. Thus Category IIc species, being only partially dependent on estuaries scored lower than Category IIa species which are entirely dependent on estuaries. Category IIb, IIc, and III represent a progressive decline in dependence and are therefore categorised accordingly.

In the case of Assumption 2, Category IIc and III were scored the same, and considerably lower than the next most dependent category, being those species which occur mainly in estuaries, but are also found at sea (Category IIb). In the final assumption the score of this latter category is lowered, giving greater emphasis to those species completely dependent on estuaries.

Table 3.2 : Dependency scores (DS_i) allocated to categories of fish included in the model according to their dependence on estuaries.

CATEGORY	DEPENDENCY SCORE (DS_i)		
	Assumption 1	Assumption 2	Assumption 3
I	not included in model	not included in model	not included in model
IIa	5	5	5
IIb	3	3	2
IIc	2	1	1
III	1	1	1
IV	not included in model	not included in model	not included in model
V	5	5	5
endemic	5	5	5

A further point of note is that not all juvenile marine fish are of equal conservation significance. Some may be widely distributed throughout the Indo-Pacific region, whereas others may be endemic to certain areas of the South African coastline. The latter are therefore considered of greater conservation importance, consequently the score of an endemic species, regardless of its category of dependance was increased to 5.

Twenty-seven species are included in the index formulation as these are the only species for which preferred recruitment periods are known. However, these species represent approximately 40% of all South African fish classified as Category II, III or V (Whitfield 1994a).

3.2.2 Optimal recruitment score (ORS_i)

Whitfield and Kok (1992) provide a summary of the recruitment periods for common coastal species in the Eastern and Western Cape Provinces, and Wallace and van der Elst (1975) give similar information for species from the KwaZulu-Natal region. Table 3.3 below summarises the recruitment periods of species for which information is available. The optimal recruitment score (ORS_i) is defined as being 5 for the main immigration period, indicated in Table 3.3 by the broad line, decreasing linearly to 0 towards the outer limits of the species recruitment period, indicated in Table 3.3 by the tapered line.

Table 3.3 : Recruitment periods for the 27 fish species, occurrence in the Great Brak Estuary is also indicated. Broad line represents peak recruitment period (*after* Wallace & van der Elst 1975; Melville-Smith & Baird 1980; Bruton *et al.* 1987, Whitfield 1990;Whitfield & Kok 1992).

SPECIES	JULY	AUGUST	SEPTEMBER	OCTOBER	NOVEMBER	DECEMBER	JANUARY	FEBRUARY	MARCH	APRIL	MAY	JUNE	COMMON NAME	Great Brak
Acanthopagrus berda													River bream	
Anguilla mossambica													Longfin eel	●
Argyrosomus japonicus													Kob	●
Diplodus sargus													Blacktail	●
Elops machnata													Ladyfish	●
Galeichthys feliceps													White seacatfish	●
Johnius dussumieri													Small kob	
Leiognathus equula													Slimy	
Lichia amia													Leervis	●
Lithognathus lithognathus													White steenbras	●
Liza dumerilii													Groovy mullet	●
Liza macrolepis													Large-scale mullet	
Liza richardsonii													Southern mullet	●
Liza tricuspidens													Striped mullet	
Megalops cyprinoides													Oxeye tarpon	
Monodactylus falciformis													Cape moony	●
Mugil cephalus													Flathead mullet	●
Myxus capensis													Freshwater mullet	●
Pomadasys commersonnii													Spotted grunter	●
Rhabdosargus globiceps													White stumpnose	
Rhabdosargus holubi													Cape stumpnose	●
Rhabdosargus sarba													Natal stumpnose	
Sarpa salpa													Strepie	●
Solea bleekeri													Blackhand sole	●
Stolephorus holodon													Thorny anchovy	
Terapon jarbua													Thornfish	
Valamugil cunnesius													Longarm mullet	

3.2.3 Longitudinal salinity difference multipliers (LSM_t)

Whitfield (1994b) shows that an important factor associated with the abundance of larval and juvenile marine fishes is the longitudinal salinity difference, being the difference in salinity between the head (upper limit of tidal influence) and mouth of the estuary. In permanently open estuaries with longitudinal salinity differences greater than approximately 20 $g.kg^{-1}$, Whitfield (1994b) notes significantly higher densities of 0+ juvenile fishes than those where longitudinal salinity differences are absent or poorly developed (Figure 3.1). Martin *et al.* (1992) indicate a similar finding, showing that the densities of post larval marine migrants in the St Lucia estuary increased significantly for almost all species following a cyclonic flushing of the system. Data presented by Martin *et al.* (1992) shows that the ebb tide longitudinal salinity difference following the flushing event remained greater than 20 $g.kg^{-1}$.

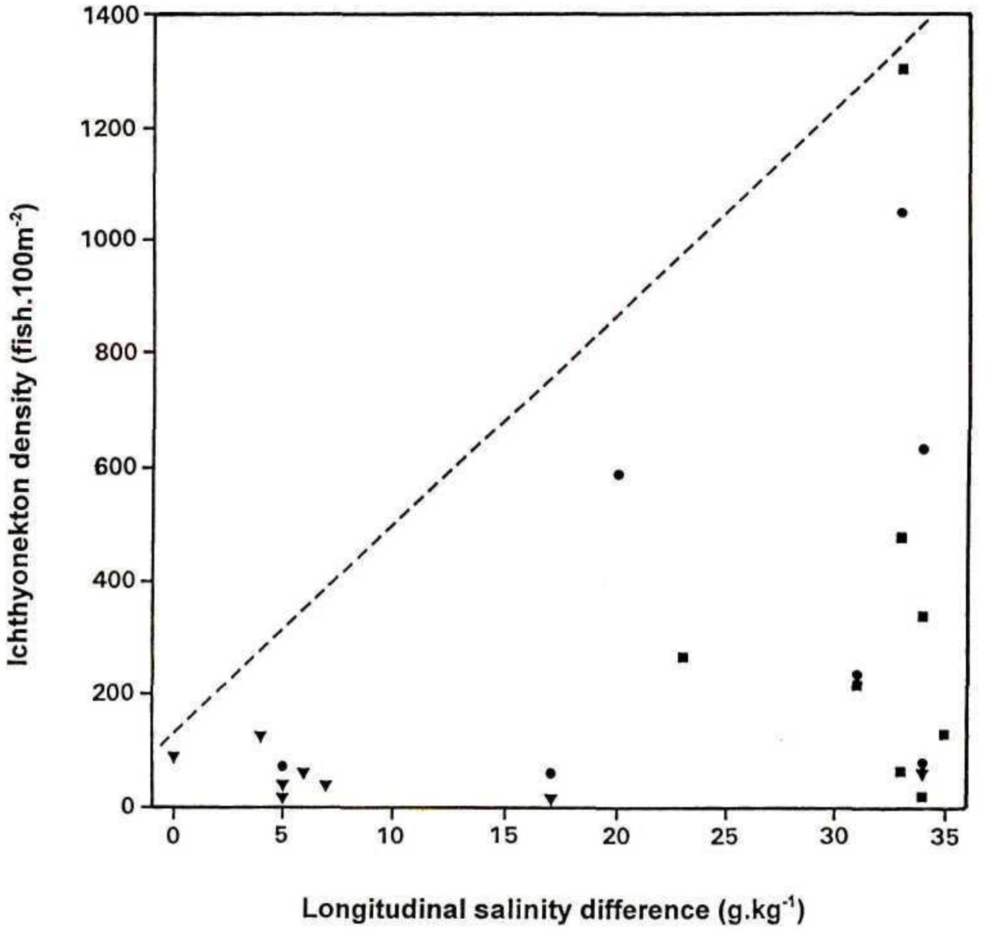


Figure 3.1 : Ichthyonekton densities versus longitudinal salinity difference (difference in salinity between head and mouth) in each of three South African estuaries on the eastern Cape coast (■ Sundays; ● Great Fish; ▼ Kariega) (Whitfield 1994b). The dashed line indicates maximum recorded fish densities.

Figure 3.1 shows that the ichthyonekton densities of the Great Fish and Sundays estuaries, characterised by strong longitudinal salinity differences were more than 5 times greater than those recorded in the Kariega system, with a poorly developed longitudinal salinity difference. Whitfield (1994b) thus suggests that recruitment is only likely to occur to any significant degree if the axial salinity difference exceeds 20 $g.kg^{-1}$.

Although this trend appears to be well supported (Whitfield 1996 *pers. comm.*), the linear regression correlation coefficient of the data shown in Figure 3.1 is a relatively poor fit ($r = 0.39$) (Whitfield 1994b). As a well defined relationship between longitudinal salinity difference and recruitment has not been established the data shown in Figure 3.1 have been rescaled to define three alternative assumptions regarding this relationship, each providing a multiplier dependent on the prevailing longitudinal salinity difference (Figure 3.2).

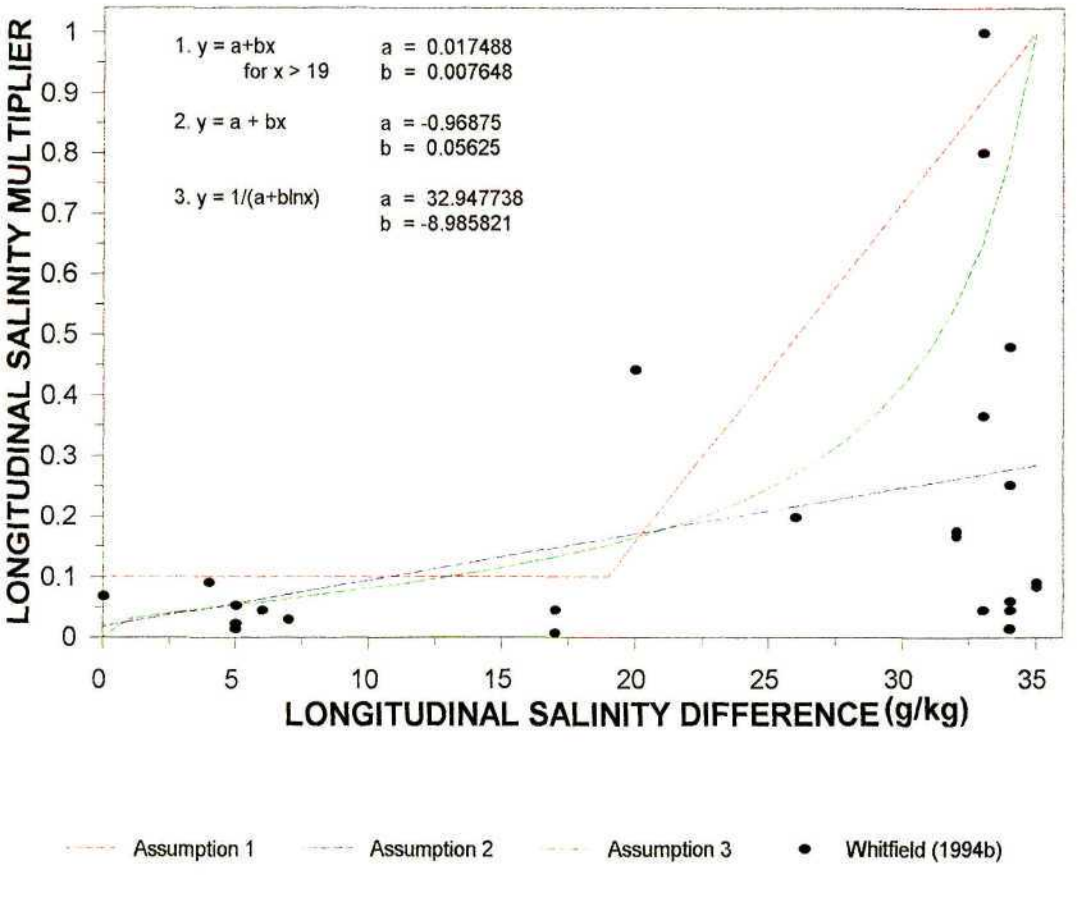


Figure 3.2 : Three assumptions regarding the relationship between recruitment and longitudinal salinity difference, based on data provided by Whitfield (1994b).

Assumption 1 is based on the supposition that recruitment will occur only at 10% of maximum when the axial salinity difference was less than 20 $g.kg^{-1}$, increasing linearly to 100% as the axial salinity difference reaches 35 $g.kg^{-1}$, as defined by the linear equation in Figure 3.2. The second alternative assumes a direct linear relationship between longitudinal salinity difference and recruitment, whilst the final assumption comprises a reciprocal logarithmic function, thereby giving greater emphasis to higher values of axial salinity difference.

3.2.4 Flow rate multiplier (*FRM_t*)

Although Whitfield (1994b) has shown a relationship between axial salinity difference and recruitment, it is likely that fish are responding to riverine-based olfactory cues entering the marine environment, and not salinity or salinity gradients *per se* (Whitfield 1996). The latter suggests that the actual volume of water entering the marine environment may also be relevant, supported in part by recently obtained data (Whitfield 1997, *pers. comm.*). A longitudinal salinity difference of 35 $g.kg^{-1}$ is thus indicative of a high freshwater input and therefore large amounts of land-based olfactory cues entering the sea.

Nevertheless, Whitfield (1997, *pers. comm.*) has also found evidence which suggests that very high flows limit juvenile fish recruitment. Figure 3.3 overleaf shows a decrease in catch per unit effort (CPUE) for three sampling events in the Thukela estuary, KwaZulu-Natal. Whitfield (1997, *pers. comm.*) consequently speculates that for a given estuary, there is a range of outflow rates which correspond to an optimum of volume of land based cues in the marine environment without compromising the ability of juveniles to enter and locate suitable habitat within the estuary. Successively higher flow rates are thus likely to limit the extent of recruitment to an increasing extent by either making access to the estuary difficult or by eliminating suitable habitat in the estuary.

In order to account for the attenuating effects of high flows on juvenile recruitment, the function depicted in Figure 3.4 was derived in consultation with Whitfield (1997, *pers. comm.*), specifically for the Great Brak and similar estuaries. Peak recruitment is assumed to occur at flow rates of between 2 and 8 cumecs, whereas lower flows do not result in peak recruitment due to lower volumes of olfactory cues reaching the sea. Higher flows result in lower recruitment due to loss of habitat as described above.

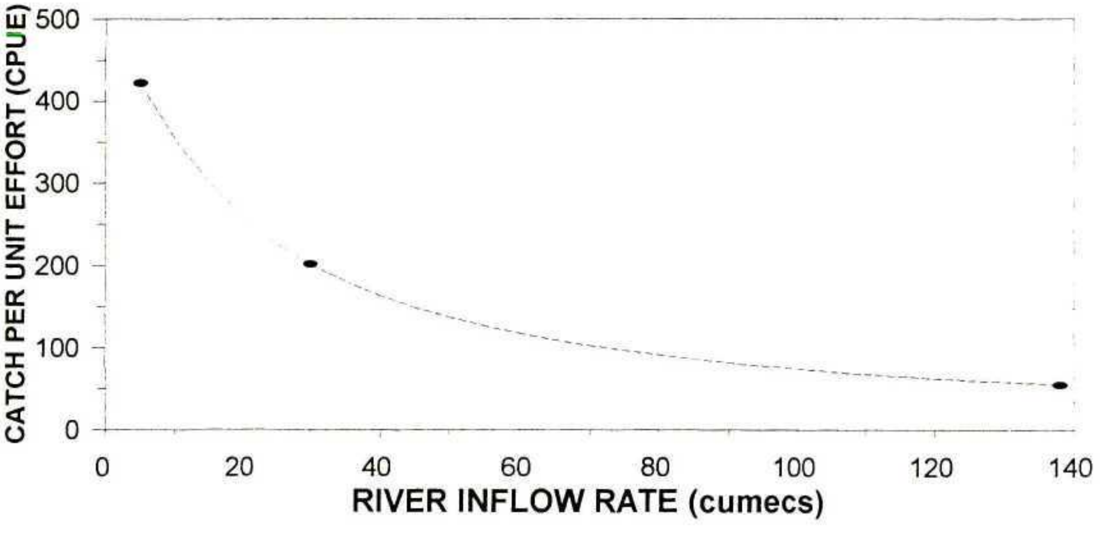


Figure 3.3 : Data provided by Whitfield (1997, *pers. comm.*), showing reduced catch per unit effort (CPUE) at the Thukela estuary at successively higher flow rates.

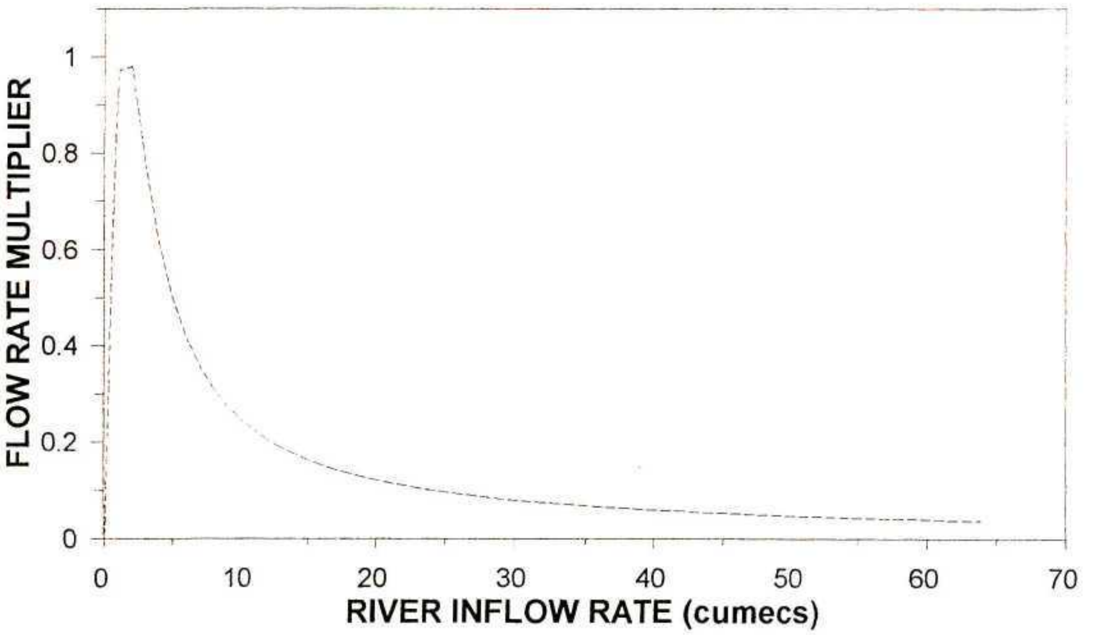


Figure 3.4 : Flow rate multiplier for a given flow rate, based on and assumed relationship between flow rate and magnitude of recruitment (Whitfield 1997, *pers. comm.*).

3.2.4 Mouth condition multiplier (MCM_t)

A further, although more obvious, limiting condition is that the mouth must be open for recruitment to occur. Thus the mouth condition multiplier is set to zero if the mouth of the estuary is closed, or to one while the mouth remains open.

3.3 FORMULATION OF THE FISH RECRUITMENT INDEX (FRI_t)

There are two primary components to be considered in the formulation of the juvenile marine fish recruitment index, firstly the aspect which describes the strength of recruitment of a particular species at a given time (optimal recruitment score), and secondly the measure which provides a value judgment relating to the importance or significance of the species (dependency score), based on the extent to which the species is dependant on estuaries and whether it is endemic or not.

As is evident from the preceding discussion the maximum value for each of the scores has been set at 5. The rationale for this is that firstly both components are considered to be equally important and should therefore receive equal weighting, and secondly scoring over a greater range (e.g. between 1 and 10) may imply a level of resolution in the input data (e.g. recruitment preference) which is currently not available. Furthermore as the index is intended to provide a relative rather than absolute measure, the maxima of the two principal scores are less important than their relative weighting. Formulating scores in such a manner also facilitates the introduction of a weighting at some later point, if required ,where the weighting for one component would be x (where $x = 0$ to 1), and the weighting for the other would be $1-x$. Although this has been raised as a possible modification to the index it will not be considered further as there is currently no reason why either of the two primary components should be given a greater weighting.

At least two alternative formulations of the fish recruitment index are possible, namely an additive and a multiplicative formulation. In the case of the former, the two component scores could be added together to yield a total (maximum would therefore be 10) for the species under consideration at that point in time, or in the latter case could be multiplied to provide a total (maximum would therefore be 25) for the species concerned. As the multipliers have been formulated to reduce recruitment under certain hydrodynamic conditions (e.g. to show no recruitment when the mouth is closed), in both cases either the product or the sum of the two primary components would need to be multiplied by the combined product of the various multipliers. This is because if any of the multipliers is zero at any point in time, a zero recruitment index should be determined by the formulation.

The two possible formulations of the index are thus;

$$FRI_t = \sum_{i=1}^{n} (ORS_{it} . DS_i).(MSM_t . LSM_t . FRM_t) \qquad \textbf{(1)}$$

or,

$$FRI_t = \sum_{i=1}^{n} (ORS_{it} + DS_i).(MSM_t . LSM_t . FRM_t) \qquad \textbf{(2)}$$

where,

FRI_t	=	Fish recruitment index at time t
ORS_{it}	=	Optimal recruitment score of species i at time t
DS_i	=	Dependency score of species i
MSM_t	=	Mouth status multiplier at time t
LSM_t	=	Longitudinal salinity difference multiplier at time t
FRM_t	=	Flow rate multiplier at time t

Whitfield (1994a) has shown that the diversity of fish species declines from 133 estuary associated species in subtropical waters to 70 species in warm temperate estuaries and only 25 species in cool temperate systems. This reduction in diversity is attributed to the loss of tropical and subtropical species due to the colder water temperatures in the south (Whitfield 1994a). Thus according to the above formulations, estuaries in the south will generally score less than those in the north as a result of the decline in species number southwards. In order to facilitate comparison between estuaries which may support different numbers of species the fish recruitment index could be normalised relative to the maximum possible score for the particular estuary in question and the associated range of species which occur in the estuary. However, as further testing of the index will be limited to the Great Brak estuary, the normalised form of the index will not be considered further.

The alternative formulations of the juvenile marine fish recruitment index and the associated assumptions within each of the components will be considered further in Chapter 6.

4 MODELLING ESTUARY HABITAT

4.1 INTRODUCTION

As noted by Kozakiewicz (1995), there have been several attempts to define the concept of habitat beyond acknowledgement that it is the conceptual link between organisms and their environment. Some definitions have an organism focus, defining habitat as a place which provides a single individual with it's life needs (food, cover, water, space and mates) (Kozakiewicz 1995). Thus Southwood (1976), relates habitat to the range of an individuals' movements, stating that for any species habitat may be defined as "*the area accessible to the trivial movements of the food harvesting stages*", while others have adopted a species or population focus, defining habitat as " *any part of the earth where the species can live, either temporarily or permanently"* (Krebs 1985). Lomnicki (1988) has suggested that habitat is "*a part of space that is permanent enough and large enough that the population of the species in question may persist within its boundaries for at least one season, or for the duration of its entire life stage*" (Lomnicki 1988).

According to Kozakiewicz (1995), definitions such as that of Krebs (1985) and Lomnicki (1988) extend the concept of habitat as an individual's living space to that of a living space which must fulfill all the requirements of a species, and additionally must be adequate to support a viable population. Other definitions avoid the individual-population debate by simply considering habitat to be "*the space in which an organism, population or species lives*" (Norse 1993). Similarly Stiling (1992) suggests habitat is "*the sum of the environmental conditions where an organism, population or community lives"* or "*the place where an organism normally lives"* or alternatively, habitat is "*the environment in which the life needs of an organism are supplied* ".

All of the above definitions imply that the presence of an organism, if not a viable population, is a fundamental and indeed, necessary attribute of habitat. For the purposes of this thesis, an alternative view of habitat is considered. This view recognises the primary environmental gradients giving rise to the 'space' which ultimately, may be populated by individuals of a species, thereby transforming the 'space' into a 'habitat'. Such an approach therefore places more emphasis on the identification of areas characterised by similarities in environmental gradients; the hypothesis being that areas which are subject to similar environmental gradients will have similar potentials to support the same range of organisms. To distinguish between this notion of habitat and the traditional view of habitat the former will be referred to as *primary* or *potential* habitat. The most significant advantage of conceptualising habitat in this manner is that if sufficient is understood about the processes which maintain the primary habitat, the consequences of any changes in these processes can be translated into consequences for the organism inhabiting the primary habitat at that time. This approach therefore, has obvious application in the management of estuaries.

4.2 DEFINING HABITAT POTENTIAL IN ESTUARIES

4.2.1 Dimensions of habitat potential

Section 1.3 outlined the key processes maintaining estuarine environments, the two primary processes being freshwater inflow and tidal exchange. Figure 4.1 illustrates the relationship between these processes and the potential habitat which they create. When occupied by an organism through recruitment or colonisation, potential habitat for a variety of organisms becomes the temporary habitat for a range of organisms. The colonisation of the environment by vegetation and animals in turn modifies the habitat. For example salt marshes and mangroves produce large amounts of organic matter which enrich the substratum. The presence of vegetation such as submerged and emergent macrophytes reduces water flow velocities thereby permitting the deposition of finer sediment particles, and ultimately providing a greater area for colonisation by the plant. The stabilisation of shorelines and mudbanks by vegetation reduces scour in subsequent floods, thereby gradually altering the estuarine environment.

Extreme flood events scour accumulated sediment, vegetation and organisms, leaving the estuary devoid of all but the most resilient organisms. For example, Hanekom (1989) recorded the effects of a medium sized flood on populations of benthic species in the Swartkops estuary near Port Elizabeth. Populations of *Solen cylindraceus*, *Upogebia africana* and *Callianassa kraussii* in the middle and upper reaches of the estuary were reduced by 93%, 49% and 28% of the pre-flood values. Marais (1982) confirms that during medium to large floods plankton and fish are swept out to sea, and in some channel-like estuaries with narrow intertidal ranges, flushing is described as 'complete'.

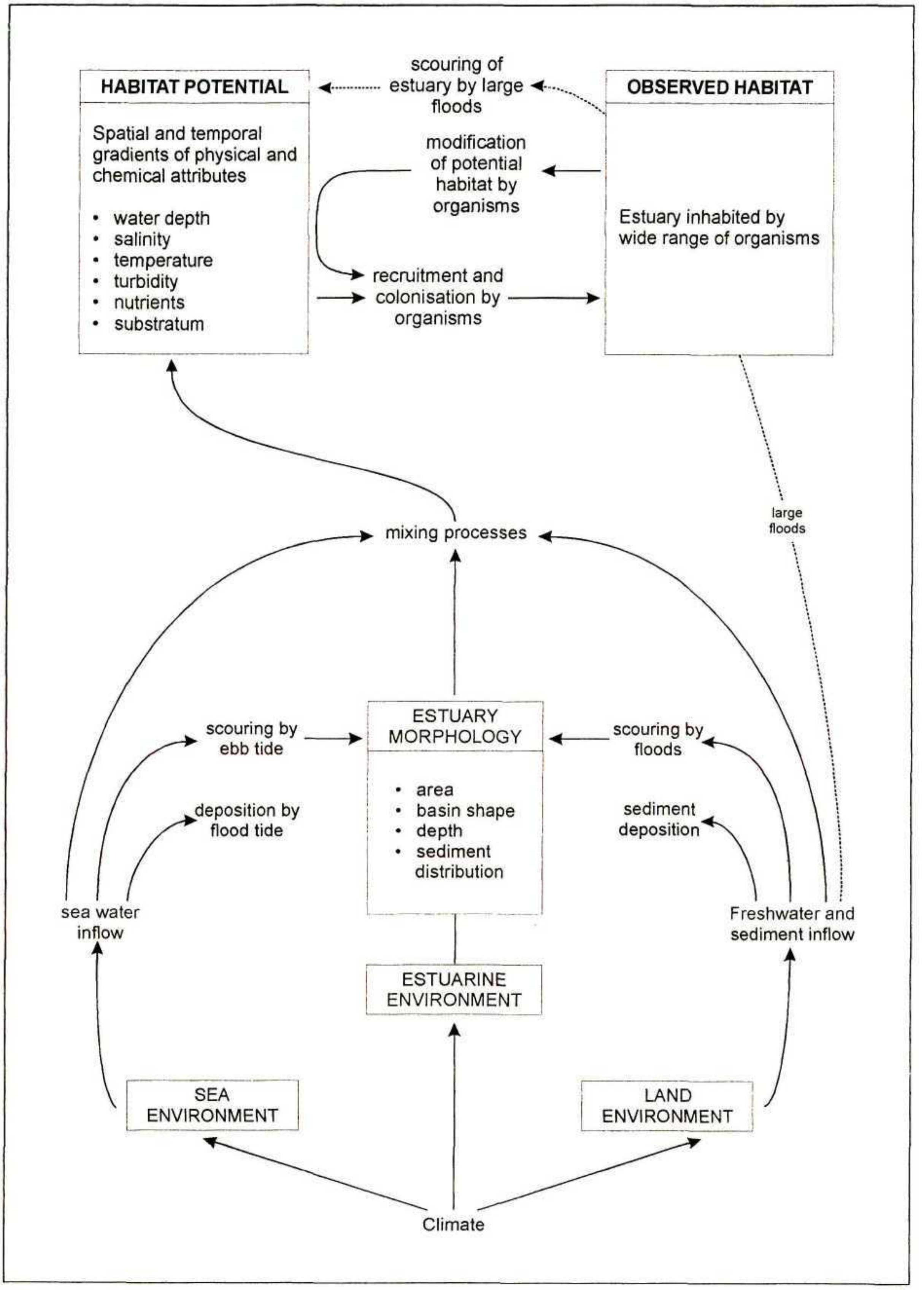


Figure 4.1 : Relationship between potential habitat, observed habitat and the processes which sustain these habitats in the estuarine environment.

The significant role of extreme events has been recognised by Haltiner (1993), acknowledging that in evaluating vegetation dynamics in these environments, traditional ecological approaches based on the concepts of succession and climax are of limited utility. Haltiner (1993) suggests that the inclusion of catastrophic events in evaluation criteria is an essential step in understanding the current status of coastal systems as well as their temporal variability.

Coastal ecosystems reflect a complex geomorphic and biotic evolution, brought about by the interaction of geomorphological and biological processes at a variety of temporal scales (Haltiner 1993). Kempe (1988) provides a synopsis of the range of time scales over which physical processes occur in estuaries, stressing the temporal magnitude of anthropogenic influences over natural processes (Figure 4.2). Both Kempe (1988) and Haltiner (1993) recognise that understanding of present ecosystem conditions, and attempts to predict further evolution of these systems requires a proper understanding of the scales of natural processes and their relationship with anthropogenic influences. Kempe (1988) recommends further that hydrodynamic models need to be coupled to geochemical and ecological models to derive models which are capable of predicting future changes.

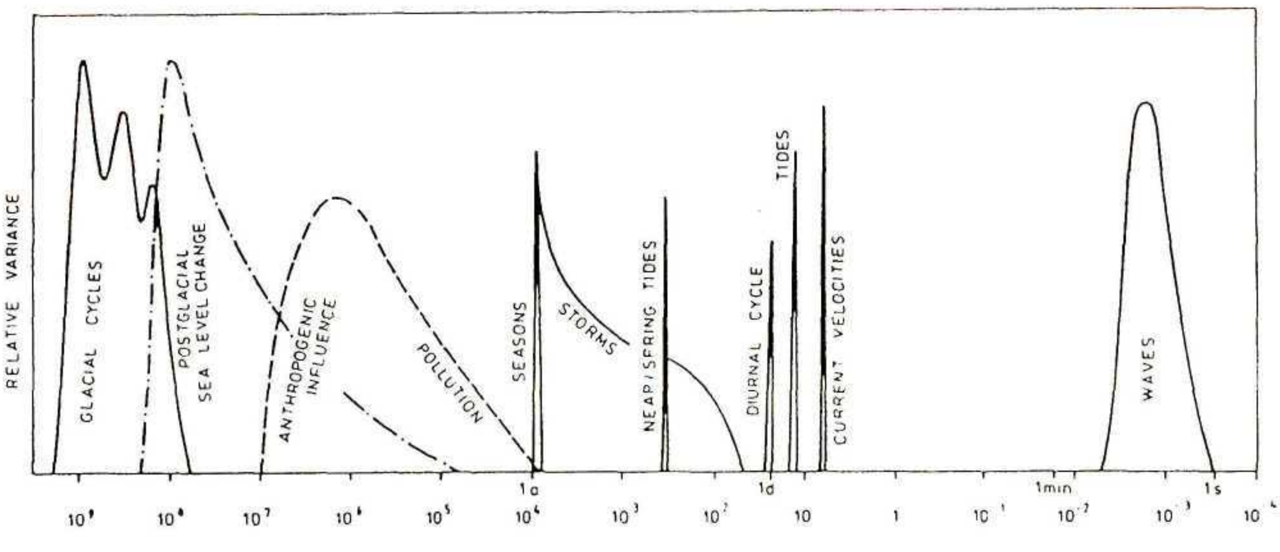


Figure 4.2 : Synopsis of the range and source of temporal change in estuaries, illustrating the temporal effects of anthropogenic influences (Kempe 1988)

4.2.2 Mapping the dimensions of habitat

The recent and rapid development of Geographic Information System (GIS) technology has provided a wealth of techniques which lend themselves to applications in environmental management. For example, Miller (1994) has illustrated the utility of GIS in mapping the distribution of species, identifying possible habitat for certain species as well as in mapping biodiversity. Pereira and Duckstein (1993) have combined GIS and multiple-criteria decision-making (MCDM) techniques to identify suitable habitat for the endangered Mount Graham red squirrel and Haines-Young, Green and Cousins (1993) have presented a variety of applications of GIS in landscape ecology. A similar spread of applications is presented in the proceedings of a recent Conference on 'GIS in Ecological Studies and Environmental Management' (Baranowski & Nagrabecka 1994). Some simulation models now incorporate spatial variation (Costanza, Sklar & White 1990; Maxwell & Costanza 1994; Maxwell & Costanza 1995), and even some individual-based models contain an element of spatial dynamics (DeAngelis & Gross 1992). While three-dimensional approaches have found wide application in mining, hydrology and landform analysis (Raper 1989), it appears that they have yet to be fully utilised in the modelling of ecological processes. This chapter will present a range of techniques, which will use the spatial analytical features of a Geographic Information System, to provide a framework for the assessment and prediction of natural and anthropogenic change in estuaries.

(ii) Estuary morphology

In the past bathymetric information was obtained through surveying of cross-sections at regular intervals, and due to the expense of the exercise was not undertaken for a large number of estuaries. More recently the advent of accurate Global Positioning System (GPS) technology has provided a more rapid means of data collection. Data are obtained by plumbing water depths for a selection of points distributed across the estuary, which in conjunction with a water level recorder, can be used to produce a set of points, each having attributes of latitude, longitude and elevation with respect to a datum water level.

A set of points collected in this manner for the Great Brak estuary was obtained from the Council for Scientific and Industrial Research (Slinger 1995 *pers. comm.*). These points were imported into a Geographic Information System, and together with topographic information obtained from the 1 : 10 000 orthophotograph (Department of Surveys and

Mapping 1989), was used to compile a bathymetric map using a proprietary topographic interpolation algorithm (Environmental Systems Research Institute 1995). Figure 4.3 shows the bathymetry of the estuary as determined by the interpolation procedure. The background shading provides a three dimensional visual perspective, also produced using a proprietary algorithm (Environmental Systems Research Institute 1995).

(i) Sediment distribution

Grain size distribution of the estuary substratum is known to be an important determinant of benthic invertebrate distribution (Kennish 1986). For example, the mudprawn (*Upogebia africana*), as its name implies, is only found in regions of an estuary characterised by fine sand or mud (Hill 1967). The distribution of fish has also been linked to sediment characteristics. In a comparison of the effect of flooding on two South African estuaries Marais (1982) found that the response of the fish population differed in each of the estuaries. The number of mullet increased in the Swartkops estuary after the flood, but decreased to insignificant numbers in the Sundays estuary for several months after the flood. Marais (1982) postulates that the mud and silt deposited in the Swartkops during the flood attracted fish by providing a food source for the mullet, whereas in the case of the channel-like Sundays estuary, the rich benthos was scoured due to the effect of the flood. Not only does this example illustrate the importance of the substratum in determining the distribution of organisms, it also provides an indication of the significance of basin morphology in determining the response of an estuary to flooding.

Although distribution of sediment type within an estuary is a key element of estuarine habitat, grain size distribution within South African estuaries is rarely mapped on a routine basis, and even large, internationally important systems such as Lake St Lucia remain unmapped. Nevertheless on the basis of schematic diagrams and descriptive information (Morant 1983; CSIR 1990), a sediment distribution map has been compiled for the Great Brak estuary, and is shown in Figure 4.4 in relation to the estuary morphology. As a consequence of the paucity of information regarding grain size distribution, only three, broad classes are recognised.

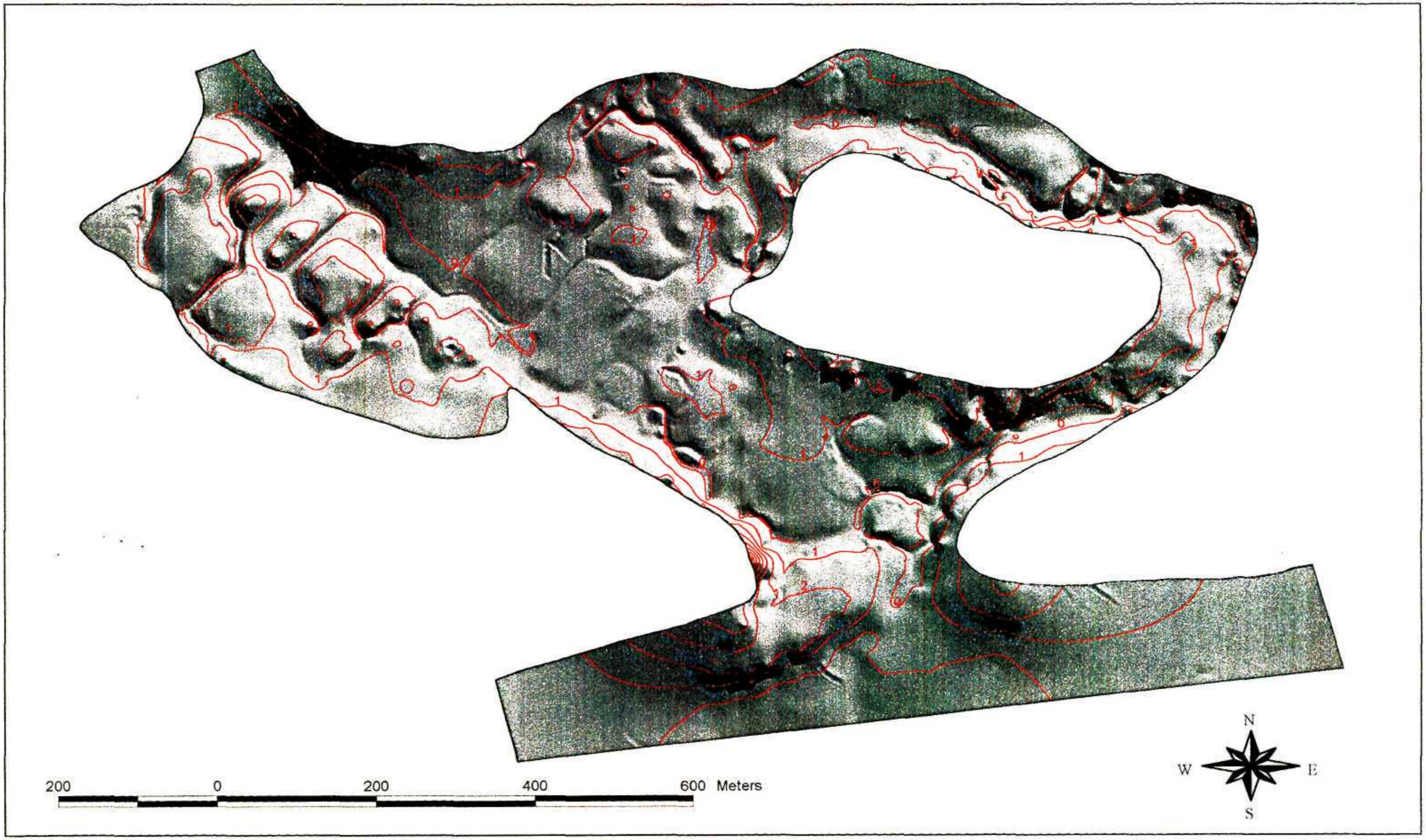


Figure 4.3: Bathymetry of the estuary as determined by ESRI (1995) interpolation procedures

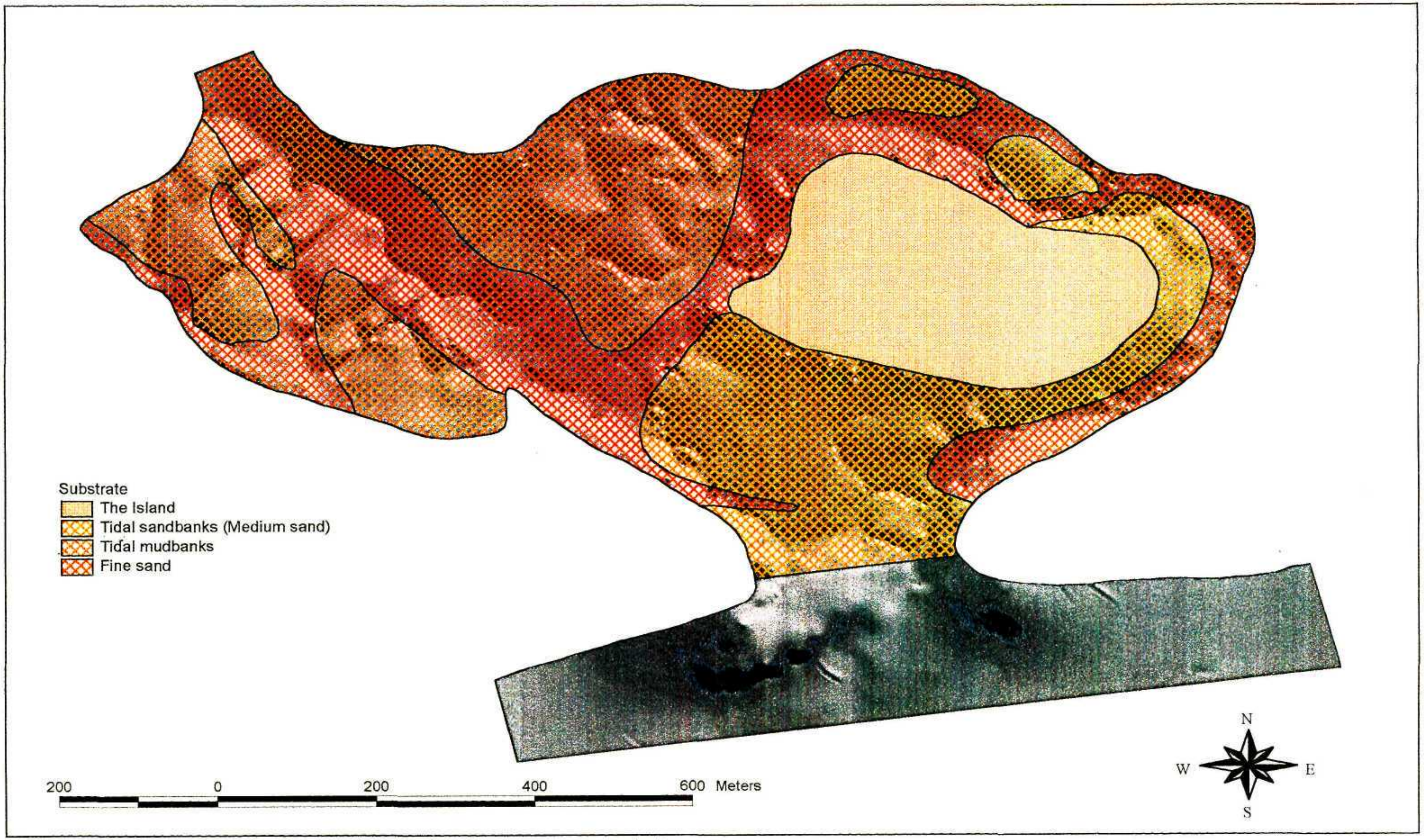


Figure 4.4: Sediment distribution in relation to estuary morphology

4.3 ANALYSING HABITAT POTENTIAL

4.3.1 Inundation and intertidal areas

Using GIS techniques, the three-dimensional model can be discretised into a two dimensional contour map at a specified height interval, whereby the contours define adjacent polygons of height within a certain range. If the contour interval is small enough, accurate surface areas can be determined for a given water elevation by listing the cumulative area of the polygons from the lowest height range to the highest. The volume of each polygon can then be calculated by multiplying the area of the polygon by the contour interval. In this way an accurate surface-area to volume relationship may be defined. This was undertaken for the Great Brak using a 0.01 m contour interval to determine surface area to volume relationships for the estuary (Figure 4.5).

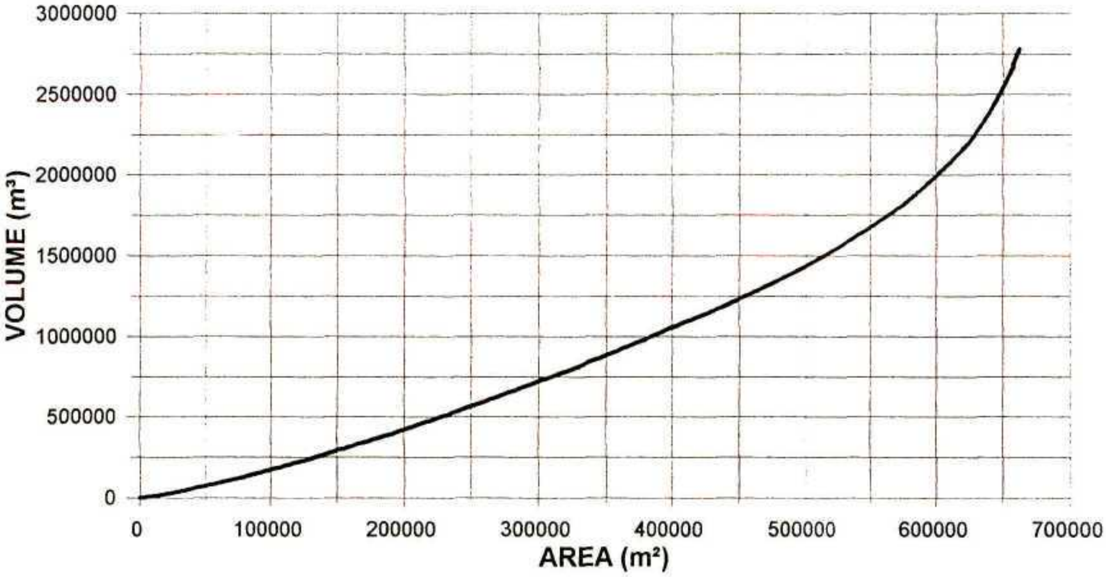


Figure 4.5 : Surface area : volume relationships for the Great Brak estuary, determined at a class interval of 0.01 m.

This approach is useful when considering potential habitat in association with fluctuating water levels. It is thus possible to calculate the area inundated at a given water level as well as the volume of water associated with this extent of inundation. Similarly, the difference between the area of inundation at high tide and low tide provides an indication of the intertidal area at that point in the tidal cycle. By overlaying the sediment distribution and the bathymetry, it is possible to use the approach described above to determine the change in intertidal areas of a given substratum type. Table 4.1 below shows, for a 0.10 m increase in water level, the change in the area of submerged sediment types. Thus at a water level of between 1.0m and 1.1m (MSL) there is almost 50 % less submerged muddy substratum as there is at a water level of between 2.0m and 2.1m (MSL).

Table 4.1 : Area (m^2) of each substratum type inundated for a given water level, and associated estuary water levels.

WATER LEVEL (m) MSL	AREA (m^2)		
	COARSE SANDBANKS	MUDBANKS	FINE SANDBANKS
0.05	10 968	8 647	74 698
0.15	10 968	11 909	86 928
0.25	15 061	15 171	99 158
0.35	22 074	22 121	118 522
0.45	30 475	35 497	138 924
0.55	39 670	45 826	150 349
0.65	49 038	55 766	159 219
0.75	59 125	68 191	167 894
0.85	70 631	80 262	175 414
0.95	81 107	89 494	180 921
1.05	89 103	98 031	186 166
1.15	94 579	108 193	193 018
1.25	100 257	118 614	198 650
1.35	104 939	124 648	201 580
1.45	109 365	129 033	204 088
1.55	114 681	134 140	207 029
1.65	119 250	138 089	208 301
1.75	122 145	140 582	209 009
1.85	124 405	141 917	209 609
1.95	126 447	143 982	210 291
2.05	127 870	145 170	210 839
2.15	128 884	145 852	211 212
2.25	129 838	146 475	211 374
2.35	130 791	147 207	211 438
2.45	131 774	147 923	211 482

Given a time series of water elevation, such as that shown in Figure 4.6, it is possible to produce a corresponding time series representing the intertidal area available to organisms. Thus Figure 4.7 shows the corresponding flux in inundated mudbank areas for minimum and maximum daily water levels. Also shown is the corresponding intertidal area, reaching a maximum during spring tides.

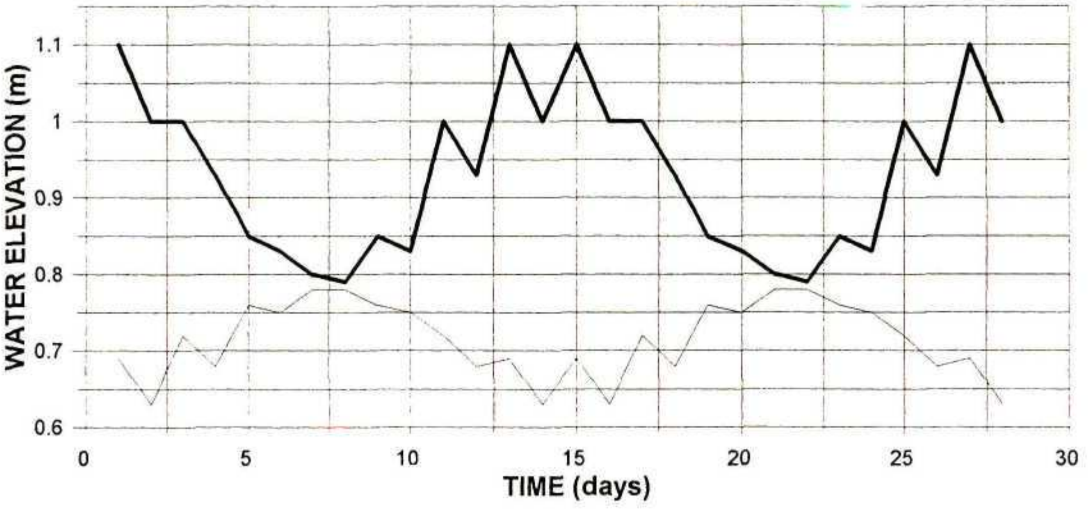


Figure 4.6 : A time series of water elevation over a thirty day period during 1988 (Source: (Slinger 1995))

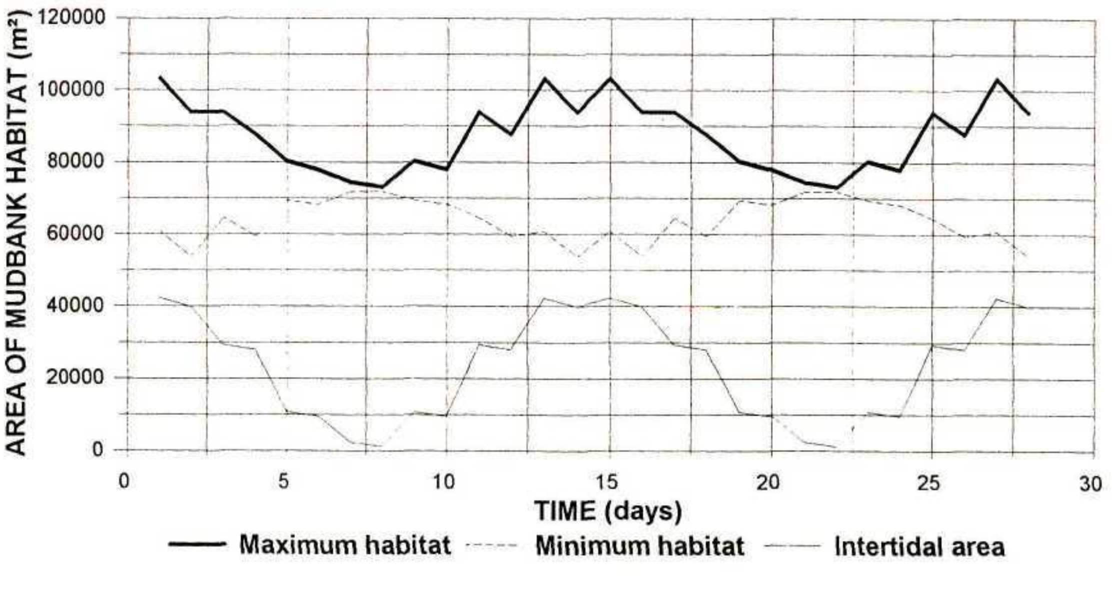


Figure 4.7 : Flux in area of inundated mudbanks for maximum and minimum daily water levels, and corresponding change in intertidal area.

Alternatively, all areas of the estuary can be classified according to their frequency of innundation. Thus Figure 4.8 shows the frequency of inundation of areas of the estuary for the time series provided in Figure 4.6.

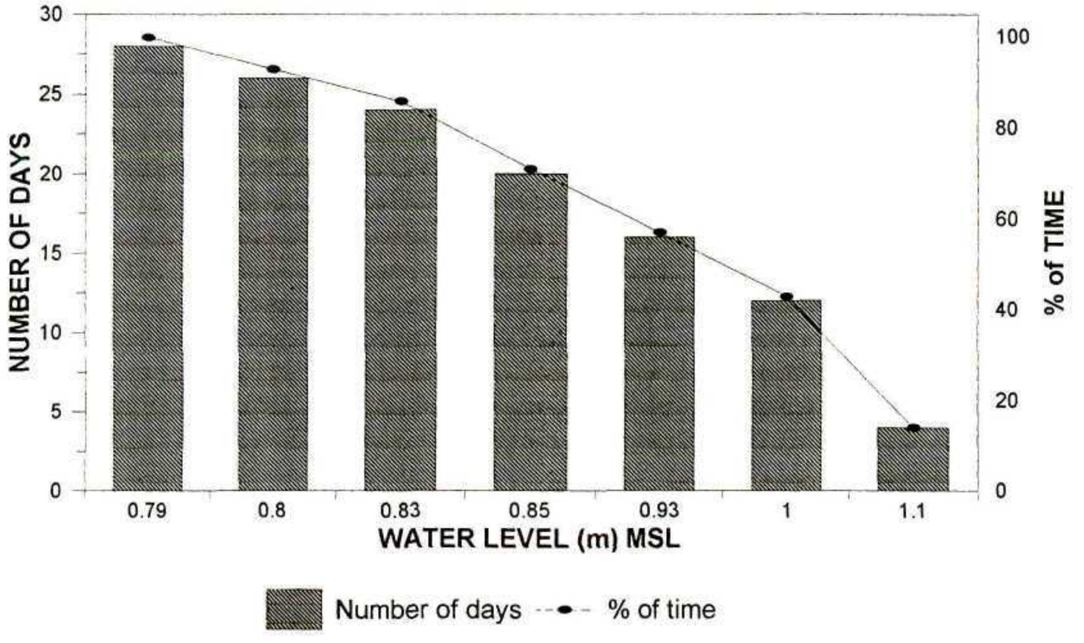


Figure 4.8 : Frequency histogram of water levels in a 30 day period in 1988.

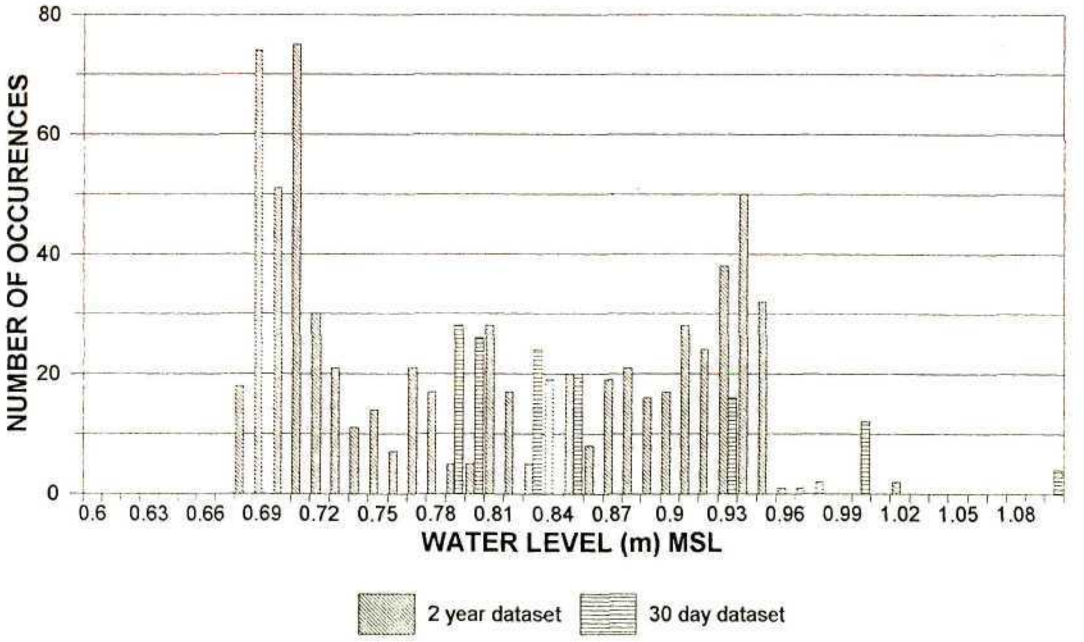


Figure 4.9 : Frequency histogram of water levels for a two year period in comparison with the 30 day period above

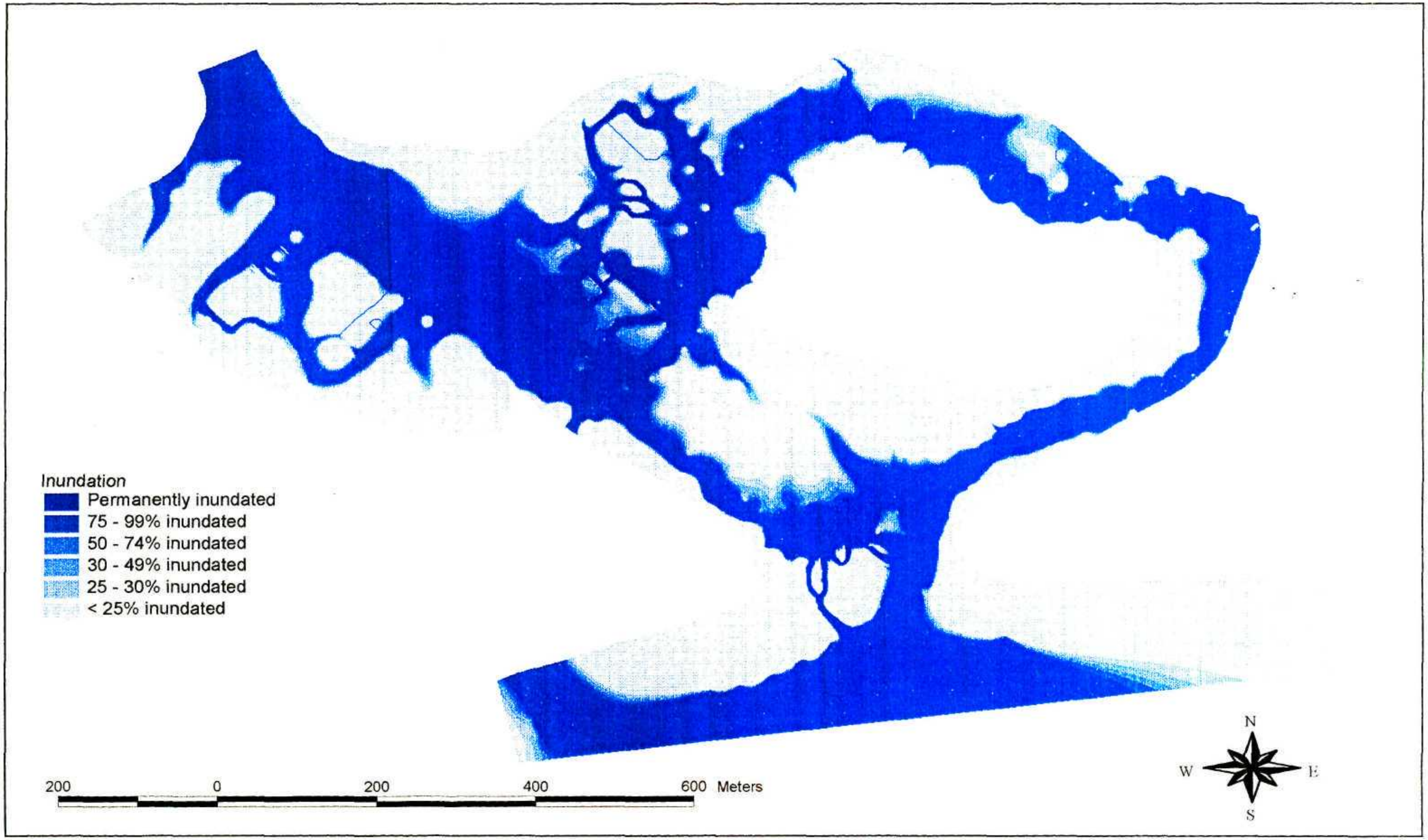


Figure 4.9: Plan view of estuary showing frequency of inundation expressed as % of time inundated

4.4 ANALYSING HABITAT SUITABILITY

4.4.1 Current approaches to habitat suitability analysis

As noted by Kozakiewicz (1995), habitat areas may differ from each other with respect to their suitability for a certain species, as well as their durability (persistence) and changeability over time. Consequently if all the life requirements of the species are identified, "*habitat types within a mosaic could be classified as optimal, sub-optimal, marginal or non-inhabitable*" (Kozakiewicz 1995). This concept is the basis of many of the spatially explicit habitat models reported in the literature. Thus in an approach formulated by Pereira and Duckstein (1993), expert opinion is used to determine scores representing the suitability of certain categories of components of habitat, such as elevation, slope, aspect, canopy cover, tree stem diameter and distance to clearings (Figure 4.11). Predicted habitat and actual habitat were found to be significantly correlated.

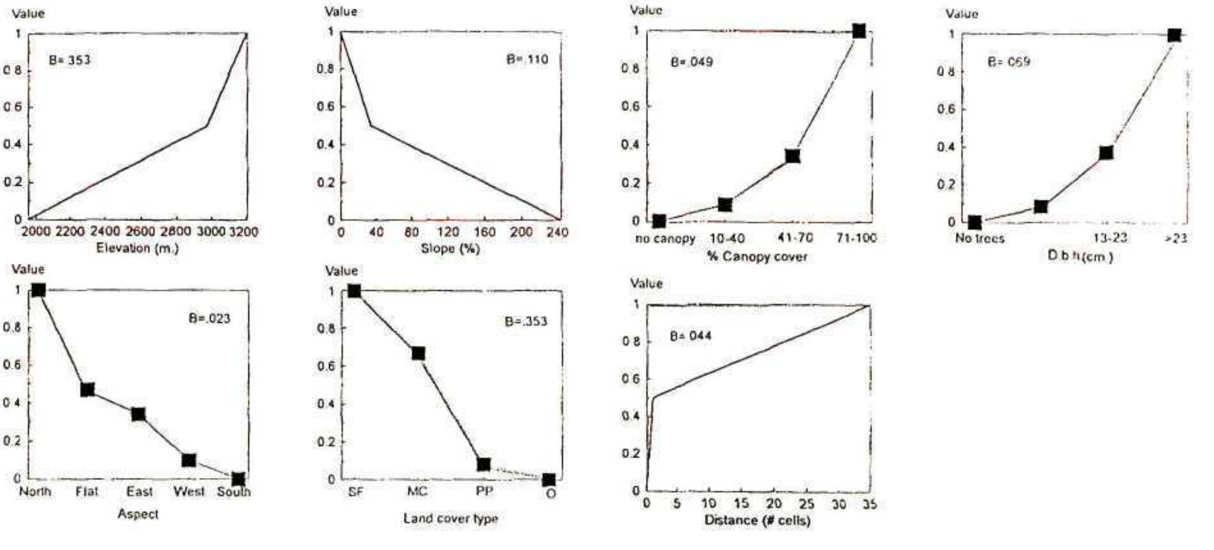


Figure 4.11: Components of the habitat of the endangered Mount Graham Red Squirrel. 'B' values represent weighting of each component to determine overall suitability. 'Distance' refers to distance from clearings (Pereira & Duckstein 1993).

Other approaches to habitat suitability evaluation require considerable data collection, as they are based on statistical analyses to correlate environmental features with population size (Stalnaker *et al.* 1994). For example, Wyoming's Habitat Quality Index (HQI), is determined by regressing several habitat variables against a standing crop of fish to produce a stream-specific minimum flow recommendation (Binns 1982). Latka and Yahnke (1984) utilise an approach somewhere in between that of Binns (1982) and Pereira and Duckstein (1993). Instead of using expert opinion to determine habitat suitability, they correlated the density of roosting Sandhill cranes with flow characteristics of the Platte river (Figure 4.12). A Suitability Index for each component was determined using regression techniques, and a Weighted Usable Area (WUA) was obtained by multiplying the area of each grid and the three suitability indices. This approach was found to produce a significant correlation between observed and predicted habitat in a separately defined test location (Latka and Yahnke 1984).

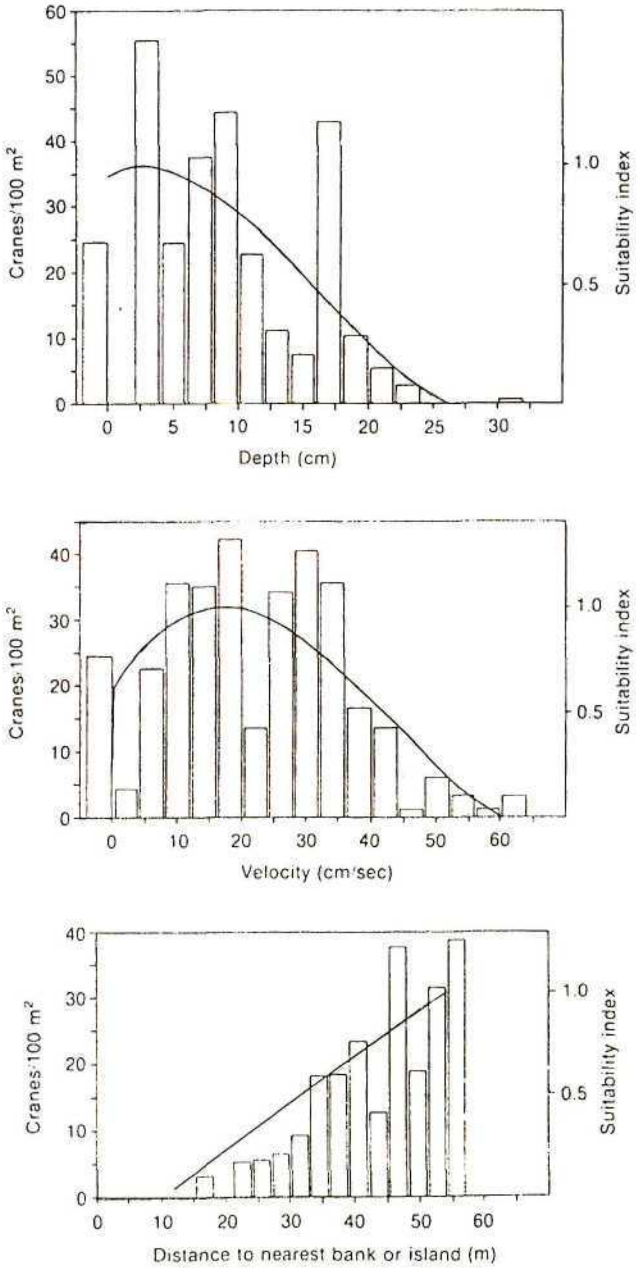


Figure 4.12 : Density and suitability indices for Sandhill Crane roosting habitat characteristics at Platte River, Nebraska (Latka & Yahnke 1984).

Another approach is that developed by Bovee and Milhous (1978), known as the Physical Habitat Simulation System (PHABSIM). Central to PHABSIM is the determination of habitat suitability criteria for species. As habitat requirements may differ during different life stages, the requirements for all life-stages need to be determined (Figure 4.13) (Stalnaker *et al.* 1994). This approach enables the assessment of likely impacts due to changes in flow characteristics, by producing an estimation of useable habitat for a given length of stream for a given species' life-stage and a given set of flow conditions.

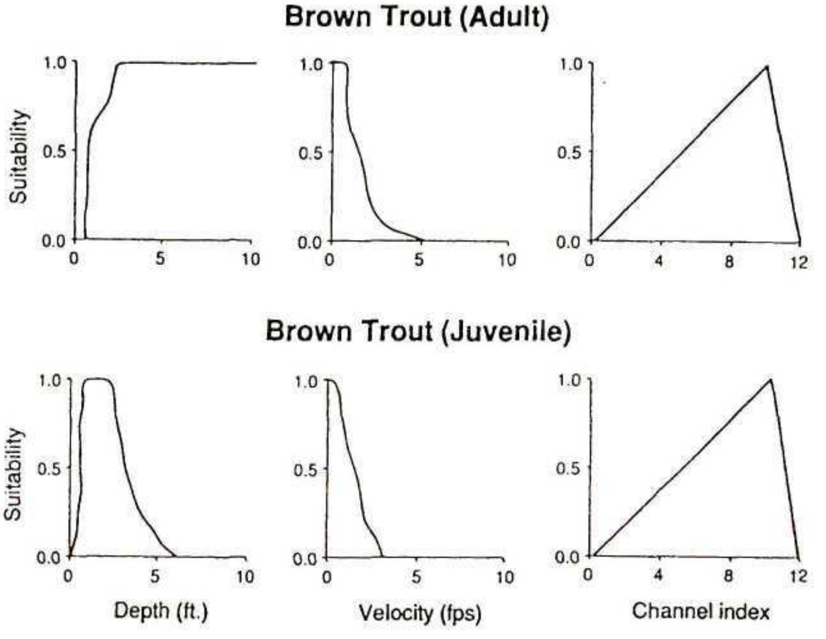


Figure 4.13 : Suitability-of-use curves for adult and juvenile Brown Trout in relation to flow conditions (Stalnaker *et al.* 1994).

More advanced techniques such as the Instream Flow Incremental Methodology (IFIM) build on these approaches by integrating measures of habitat into habitat units which are then related to flow over time resulting into a Habitat Time Series (Figure 4.14) (Stalnaker *et al.* 1994). In these approaches habitat suitability is determined for both the microhabitat (depth, velocity, substrate & cover) and the macrohabitat (water quality, channel morphology, discharge & temperature) (Stalnaker *et al.* 1994). As noted by Stalnaker *et al.* (1994), a criticism of these approaches is the requirement to analyse habitat species-by-species. Not only does this approach not consider inter-species competition, but also requires the determination of habitat suitability curves for a wide range of environmental parameters and all species which may be important (Stalnaker *et al.* 1994).

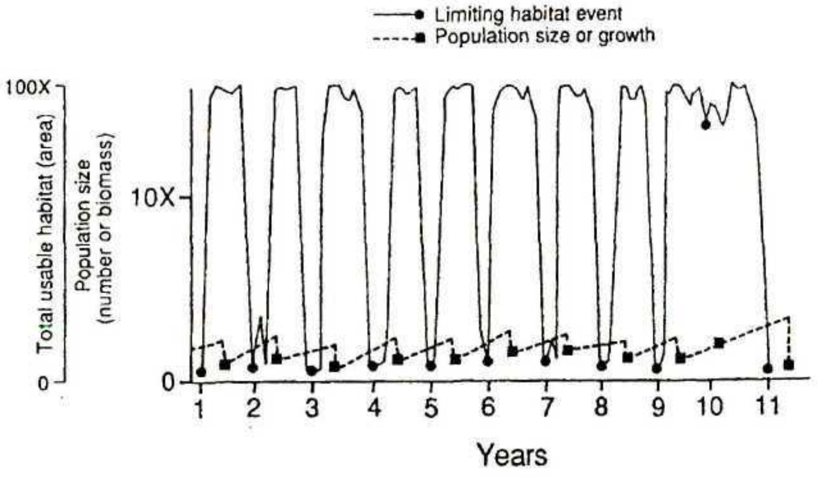


Figure 4.14 : Fish population size related to useable habitat area using the IFIM (Stalnaker *et al.* 1994)

4.4.2 An alternative approach to habitat suitability

As is evident from the discussion in Section 4.2, change in estuaries occurs at a variety of spatial and temporal scales. Thus for example, some components of a benthic invertebrate's habitat, such as substratum characteristics and vegetation cover, may change over a long time scale, whereas others such as inundation or salinity may change over a shorter timescale. Thus at least two categories of change may be evident (Figure 4.15).

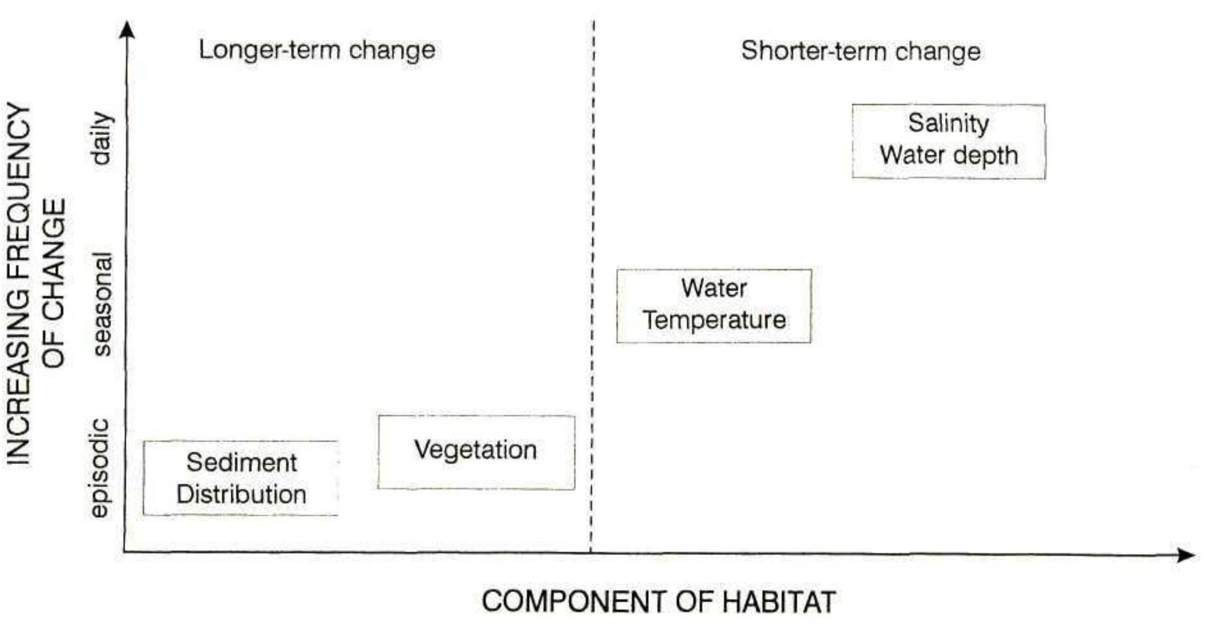


Figure 4.15 : Components of habitat and their rates of change

Because sediment and vegetation undergo change at less frequent intervals in estuaries, they are considered to form the basis of potential habitat. Using the analytical capabilities of a GIS it is possible to overlay the sediment distribution map and a map of the distribution of vegetation types. The intersection of these two data sources produces a single map which defines unique combinations of classes of particular sediment size distribution and vegetation type. Applying this methodology using the available sediment distribution and vegetation maps, yields 21 unique combinations of sediment and vegetation (Figure 4.16).

As each area is defined only by the interaction of two variables it is relatively easy to score the preference of an organism for each of the areas. For example, of the 21 potential habitat areas, the benthic invertebrate *Upogebia africana* could occur in 7. Table 4.2 provides a matrix of potential habitat areas and associated estimates of suitability. As is the case with the methods discussed above, suitability is scored on a scale of 0 (not suitable) to 1 (completely suitable), considered from the perspective of the maximum populations these areas are likely to support.

Table 4.2 : Assumed potential habitat suitability for several unique potential habitat areas. Note : these are provided for illustrative purposes and are not intended to represent the actual habitat preferences of *Upogebia africana*.

POTENTIAL HABITAT AREA (PHA)	SEDIMENT TYPE	VEGETATION	HABITAT SUITABILITY SCORE	REMARKS
PHA 10	mud	*Triglochum bulbosum / Sarcoconia natalensis*	0.8	vegetation limits densities
PHA 12	mud	*Zostera capensis*	0.9	vegetation limits densities
PHA 13	mud	*Ruppia cirrhosa*	0.8	vegetation limits densities
PHA 13	mud	no cover	1	considered ideal habitat
PHA 17	fine sand	*Triglochum bulbosum / Sarcoconia natelensis*	0.5	substratum not ideal
PHA 20	fine sand	*Zostera capensis*	0.6	substratum not ideal
PHA 21	fine sand	no cover	0.7	substratum not ideal

Thus potential habitat areas (PHA's) where maximum densities (PHA x) are likely would be scored as 1, whereas potential habitat areas which are likely to support populations at 30% of maximum density (PHA y), would be scored at 0.3. Assuming a maximum prawn density of 500 individuals per m^2, 50 m^2 of PHA x would support a maximum of 25 000 prawns, whereas PHA y would only support a maximum of 7 500 prawns. Expressed differently, 10 km^2 of

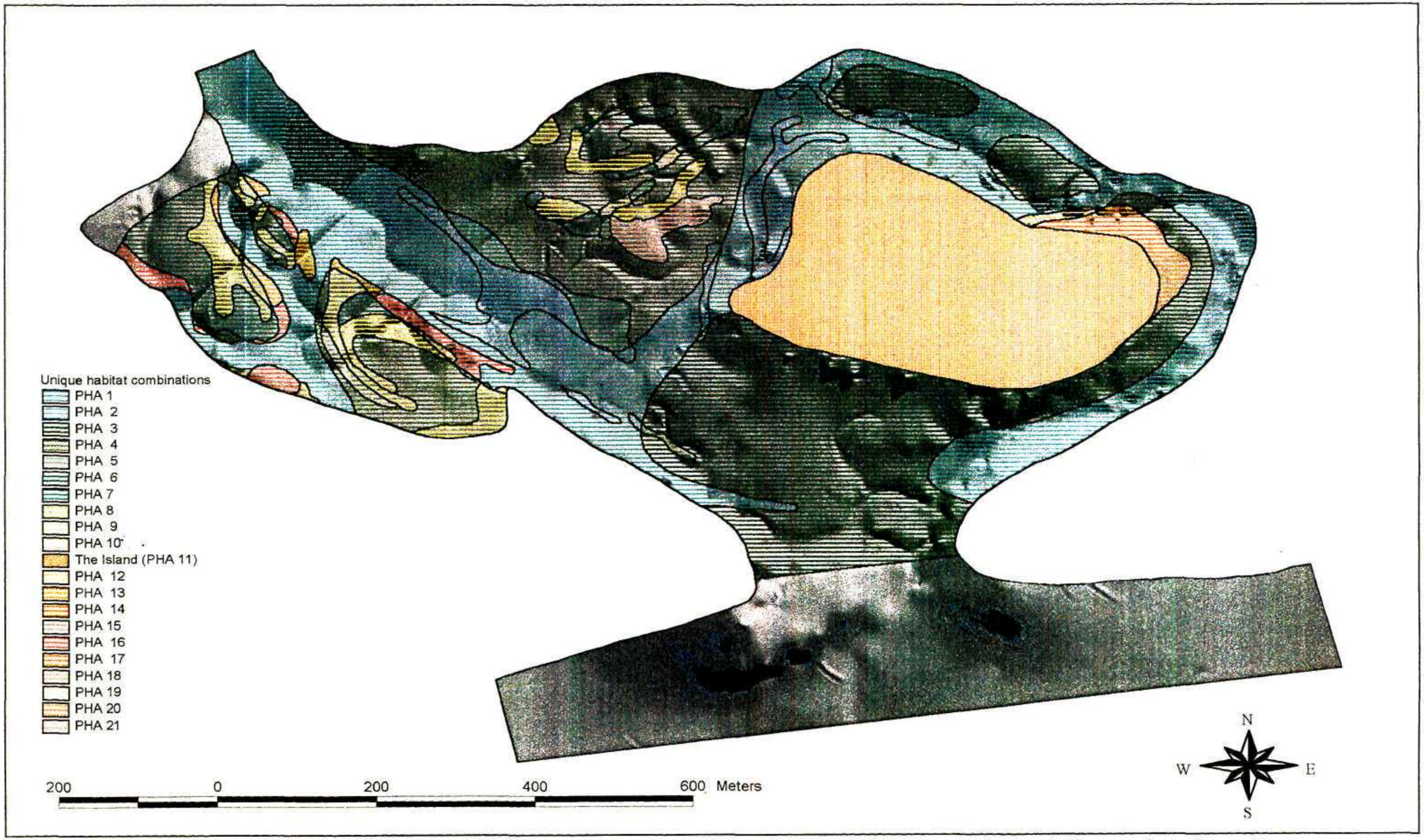


Figure 4.16: Distribution of the 21 potential habitat areas representing unique combinations of vegetation and sediment type

potential habitat at a suitability score of 0.3 would thus support the same maximum prawn population as 3 km^2 of potential habitat at a habitat suitability score of 1. The habitat suitability score thus provides a means of weighting habitat areas such that the resultant value provides an integrated measure of both the quality and quantity of habitat available for a particular species. Stewart, Scott and Iloni (1993) present a variety of techniques for determining scores such as these on the basis of expert consensus. The advantage of these techniques is that the process incorporates a sensitivity analysis and checks for consistency using approaches such as swing weighting and conjoint scaling.

The total *available* potential habitat area can thus be determined by summing the areas of the different potential habitat units weighted by the habitat suitability score, as shown below :

$$TAPHA_t = \sum_{i=1}^{n} AREA_{t_i} . SUITABILITY_{ij} \qquad \textbf{(3)}$$

where,

TAPHA	=	Total available potential habitat area at time *t*
AREA	=	Area of potential habitat unit *i* at time *t*
SUITABILITY	=	Habitat suitability score for potential habitat area *i* for species *j*

As was shown in Table 4.1 it is possible to describe the availability of potential habitat area as the water levels change. The 21 potential habitat areas shown in Figure 4.16 are also subject to fluctuating water levels, and will consequently experience differing periods of innundation. The submerged habitat available to benthic invertebrates such as *Upogebia africana* will therefore vary with water level. Section 4.3.1 described a methodology whereby sediment type distribution and estuary morphology were overlain to generate an estimate of submerged area (or intertidal area) for a given water level (or tidal range) in the estuary. Utilising this approach, Figure 4.17 overleaf represents the change in total potential habitat area (TAPHA) available to *Upogebia africana* over the same 28 day period shown in Figure 4.6, assuming that the habitat suitability indices given in Table 4.2 above are valid.

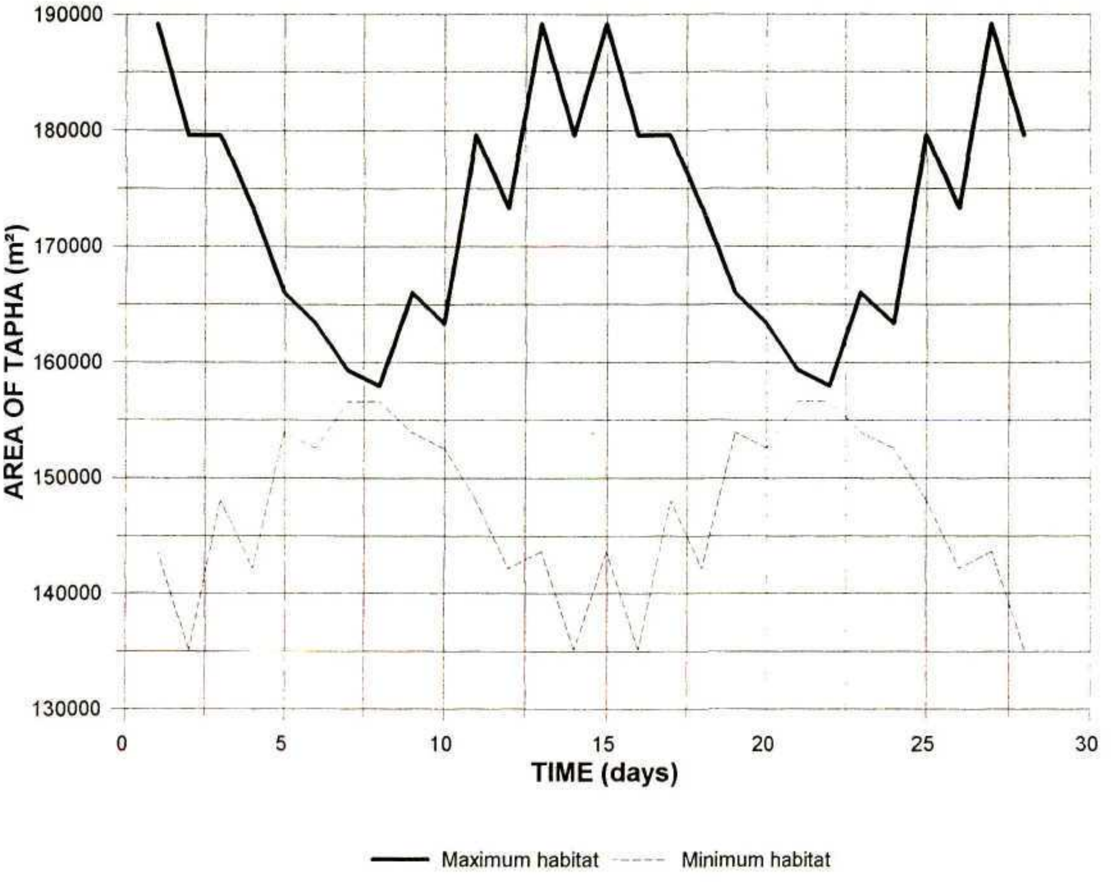


Figure 4.17 : Fluctuation in available potential habitat for *Upogebia africana* over a twenty eight day period

Whether an organism actually utilises the potential habitat or not, is considered to be dependent on other factors which fluctuate in the short term, such as whether or not the organisms' range of physiological tolerances (e.g. temperature and salinity) and other requirements are met in this available potential habitat area within the specified time interval. Chapter 5 presents a modelling approach which addresses this aspect. Chapter 7 discusses the assumptions and limitations of this approach to habitat modelling in greater detail.

5 STRUCTURE OF THE *Upogebia* MODEL

5.1 INTRODUCTION

The anomuran mudprawn, *Upogebia africana* (Ortmann) is widely distributed along the southern African coast, and has been recorded from Langebaan lagoon in the west to Inhambane in the east (Figure 5.1) (Hanekom & Erasmus 1988). Hanekom and Baird (1992) indicate that *Upogebia africana* populations are a major food source for several fish and bird species, while Wooldridge (1991) is of the opinion that populations contribute substantially to epibenthic and benthic macrofaunal biomass in many southern African coastal environments. In addition to their wide distribution and importance in the estuarine food chain, *Upogebia africana* are widely collected as fishing bait by humans (Hill 1967; Hanekom 1980; Martin 1991; Wynberg 1991; Hanekom & Baird 1992). The organism is also relatively well studied, having been the subject of at least two PhD theses (Hill 1967; Hanekom 1980) and several papers (Hill 1971; Hill 1977; Hill 1981; Emmerson 1983; Hanekom & Erasmus 1988; Wooldridge 1991; Hanekom & Baird 1992; Wooldridge 1994; Wooldridge & Loubser 1996).

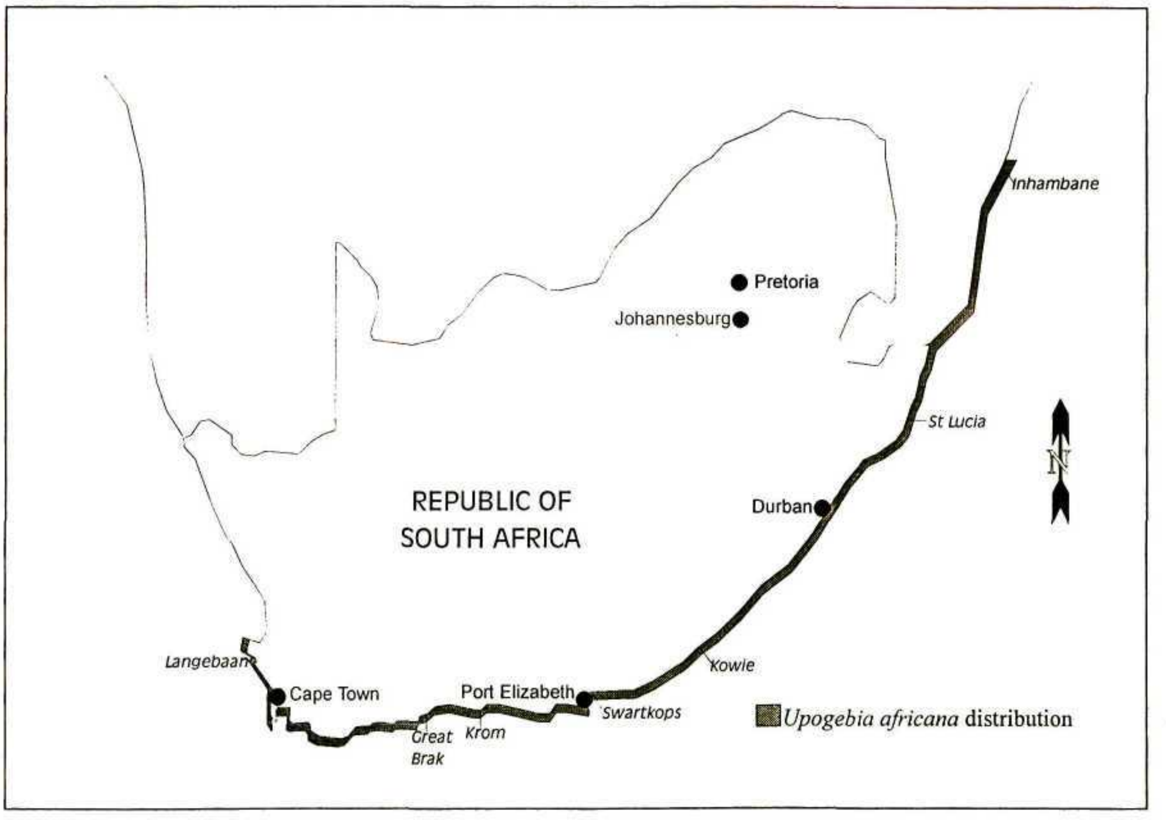


Figure 5.1 : Distribution of *Upogebia africana* in southern Africa (*after* Hanekom & Baird 1992)

As a potential indicator of estuarine functioning therefore, *Upogebia africana* has several advantages. Notwithstanding these, the most important reason for selecting *Upogebia africana* as an indicator organism, relates to the life history of the species.

5.2 THE LIFE HISTORY OF U*pogebia africana*

Upogebia africana is a burrowing mudprawn, typically found in the subtidal and intertidal mudflats of estuaries (Hill 1967), living in burrows between 20 cm and 60 cm deep. The organism feeds by pumping water through the burrow and out through a second vertical shaft, removing detritus from the passing water. Breeding occurs throughout most of the year with the onset of breeding being associated with warmer post mid-winter temperatures (Hill 1977; Hanekom & Erasmus 1989). Females produce several broods per year with generally more than 1000 eggs being produced per brood (Hill 1977; Hanekom & Erasmus 1989). Eggs develop over a period of approximately one month depending on water temperature, and hatch nocturnally within the estuary to yield Stage I zoea. In order to develop further these zoea need to be exported into the marine environment where within approximately 24 hours these develop to Stage II zoea (Figure 5.2). In the marine environment the zoea develop through to Stage IV zoea and finally to a Postlarval Stage before recruiting back into estuaries and subsequent growth to maturity. Growth of the zoea in the marine environment occurs over approximately 14 days in summer and 28 days in winter (Wooldridge 1995, *pers. comm.*).

According to Wooldridge (1991, 1994) the marine phase of development is obligatory for this species, and Stage I zoea trapped in a closed estuary will not develop further. This hypothesis explains why, as noted by Day (1981a), *Upogebia africana* are not found in estuaries which are known to have been closed to the sea for a long period of time, and also explains the discontinuous size classes of adult populations observed in the Great Brak by Wooldridge (1994). In addition to this obligate marine phase in larval development, Wooldridge (1991, 1994) and Wooldridge and Loubser (1996) have shown that the timing of the maximum release of zoea is linked to the spring tide phase of the lunar cycle. Conversely maximum incursion of postlarvae into estuaries was found to be linked to the neap tidal phase. The implication of this is that not only must the estuary mouth be open for a sufficient time to permit export of zoea and recruitment of postlarvae, but furthermore that this should ideally occur in association with spring and neap tides.

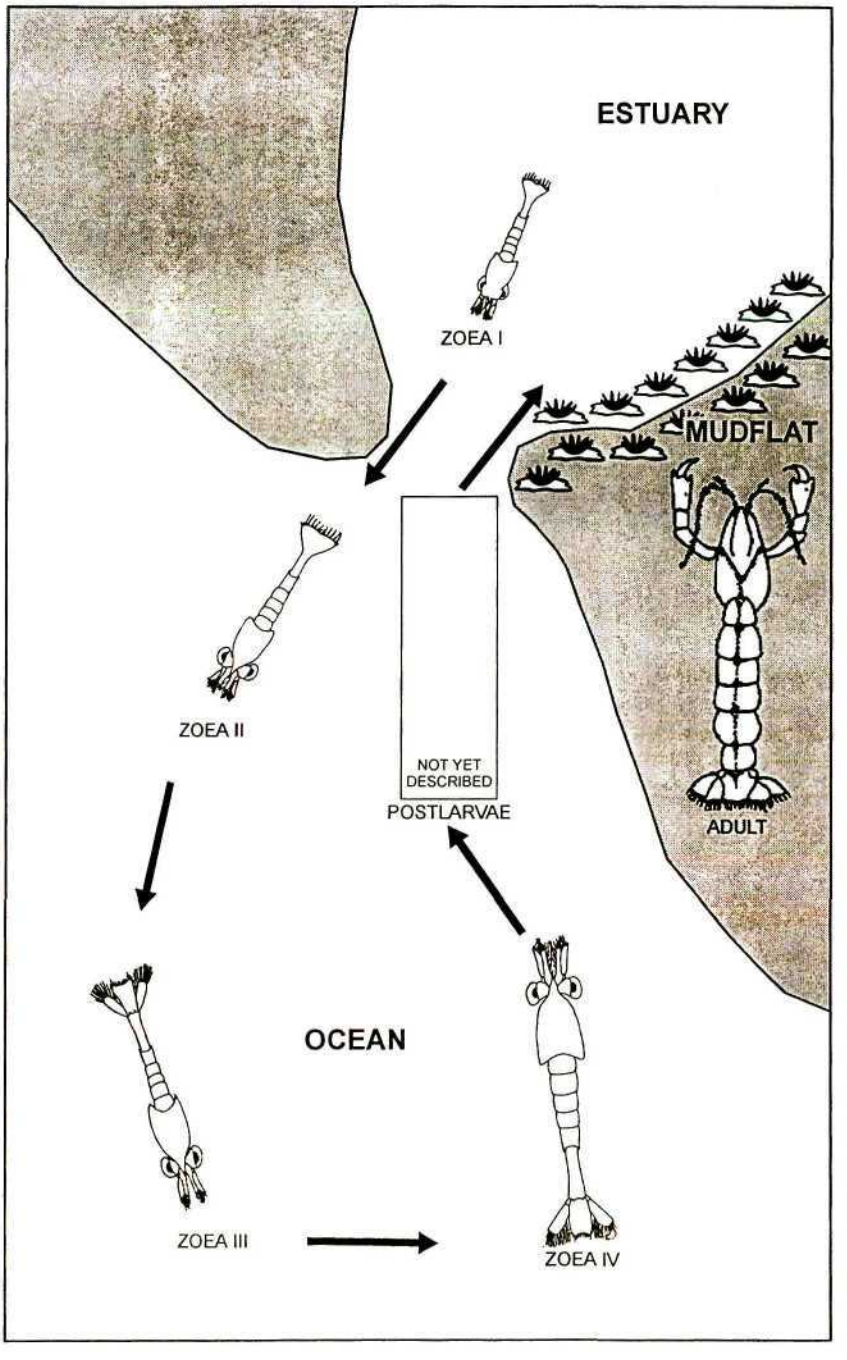


Figure 5.2 : The life history of *Upogebia africana* (Wooldridge 1995, *pers. comm.*)

In summary, the complex dependence of populations of *Upogebia africana* in South African estuaries on the availability of opportunities for export of zoea and recruitment of postlarvae through the estuary mouth renders the organism a sensitive indicator of the extent to which an estuary is providing opportunity for recruitment. The remainder of this chapter focuses on different stages of the organism's life history, and presents the formulation and structure of the model, drawing from the existing literature and also from expert opinion.

5.3 GROWTH OF *Upogebia africana*

Hill (1967), Hanekom (1980) and Hanekom and Baird (1992) have investigated the growth of *Upogebia africana* in the Kowie and Swartkops estuaries respectively. Figure 5.3 shows the estimates of growth provided by Hill (1967) and those determined by Hanekom and Baird (1992) in a more comprehensive study. Hanekom and Baird (1992) grouped prawns into 1 mm size classes and constructed size class frequency histograms for each sampling date. Modal means were fitted to the von Bertalanffy growth model to provide, for two sites, the following:

$$L_t = 19.0(1 - e^{-0.0454(t+4.0726)}) \qquad \textbf{(4)}$$

and

$$L_t = 24.4(1 - e^{-0.0583(t+1.8446)}) \qquad \textbf{(5)}$$

where,

L_t = carapace length at age *t* (mm)

As can be seen from Figure 5.3 the *Upogebia africana* sampled at site 2, appeared to have had a slower growth rate than those sampled at site 1, and in addition, those at site 2 reached a smaller final size, leading Hanekom (1980) to refer to the two populations as 'stunted prawns' and 'large prawns' respectively. Having examined differences in salinity, temperature, substratum and tidal exposure between the two sites Hanekom (1980) concluded that the differences in the growth rates at the two points was due to the greater degree of exposure to marine water borne detritus at site 1. In support of this hypothesis, data also obtained by Hanekom (1980) for the Krom estuary, suggested the same pattern. Prawns from the upper reaches of the Krom estuary were considerably smaller than those sampled adjacent to the mouth. Hanekom (1980) also reports that Christie and Moldan (1977) encountered a similar distribution of macrobenthos in the Langebaan lagoon.

For the purposes of this model, the growth rate of prawns at site 1 was considered to reflect growth under conditions of ideal food supply, not withstanding the effects of temperature and salinity on growth. Hanekom (1980) has noted that temperature could affect the growth of prawns in two ways. In the first instance, Kinne (1970a) has shown that several marine invertebrates grow faster and reach sexual maturity at a smaller size in the warmer parts of their distribution, relative to the cooler parts of their distribution. Secondly, Hanekom (1980) cites Hill (1967), who showed that if the water column temperature exceeded 27 °C, then *Upogebia africana* responded by reducing the circulation rate of water through its burrow, consequently resulting in a lower net import of suspended food particles and potentially retarding the growth rate. Similarly, Kinne (1970b) has also shown that low salinities frequently bring about a reduction in body size. Growth under conditions of ideal food supply as well as ideal temperature and salinity is likely to be enhanced, and consequently a maximum size for a given age was assumed to take the form :

$$L_t = \frac{30t}{t+11} \tag{6}$$

Figure 5.3 shows a comparison of the data provided by Hill (1967) and Hanekom and Baird (1992), with the maximum size-age relationship assumed in Equation 6 above. The longevity of individual prawns is estimated to be at least 4 years, with a maximum carapace length of approximately 26 mm being reached.

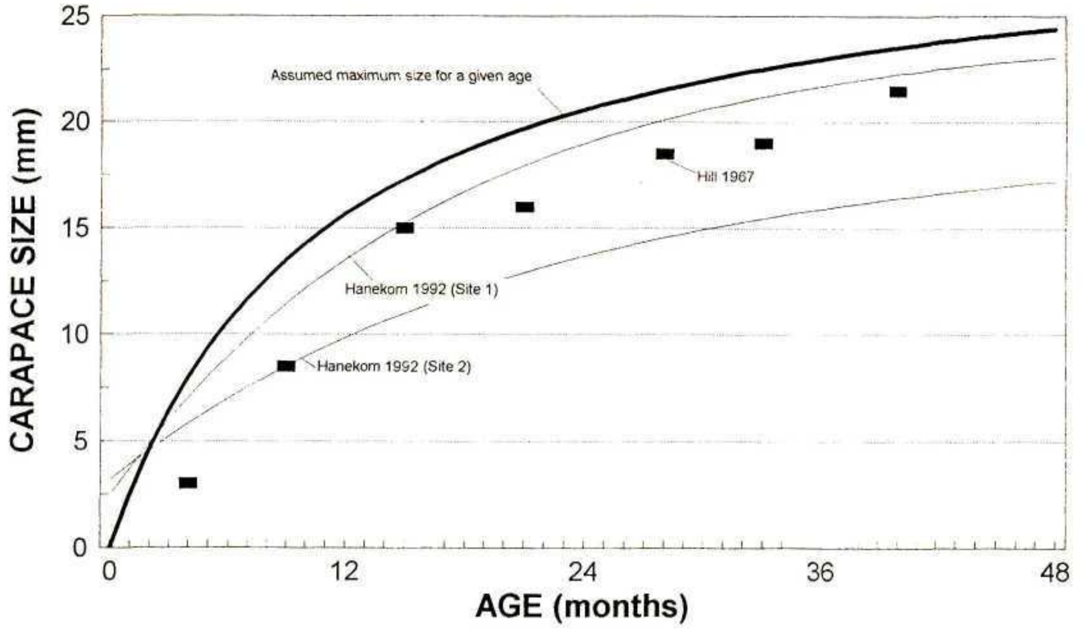


Figure 5.3 : Growth rates of *Upogebia africana* as determined by Hill (1967) and Hanekom and Baird (1992). The assumed maximum size for a given age is also shown.

5.3.1 Effects of temperature on the mortality and growth of *Upogebia africana*

Vernberg (1975) has provided a descriptive account of the response of an organism to an environmental gradient such as temperature or salinity. The middle region (biokinetic zone) is the range of an environmental gradient, to which the organism may remain exposed for a prolonged period (Figure 5.4). The extreme portions of the gradient are known as the lethal zone or zone of resistance adaptation. Exposure of an organism to these extremes will result in stress and possibly death.

However, as Vernberg (1975) points out, characterisation of these extreme zones is difficult since both the duration and intensity of exposure to the environmental factor need to be considered. In addition, the response of a species to an environmental factor may differ during different stages of the life cycle of the organism. Further factors such as body size, sex, moulting, previous exposure to environmental conditions and the season may also affect the response of the organism to prevailing environmental gradients (Vernberg 1975). Kinne (1970a) highlights the fact that aquatic invertebrates are subjected to simultaneous and varied environmental changes, and consequently respond to the total resulting stress rather than to single constituent factors. In fact, as Kinne (1970a), Vernberg (1975) and other researchers have shown, interpreting the response of an organism to environmental gradients is a complex problem; very often characterised by a paucity of data due to the difficulties in designing meaningful multivariable experimental conditions.

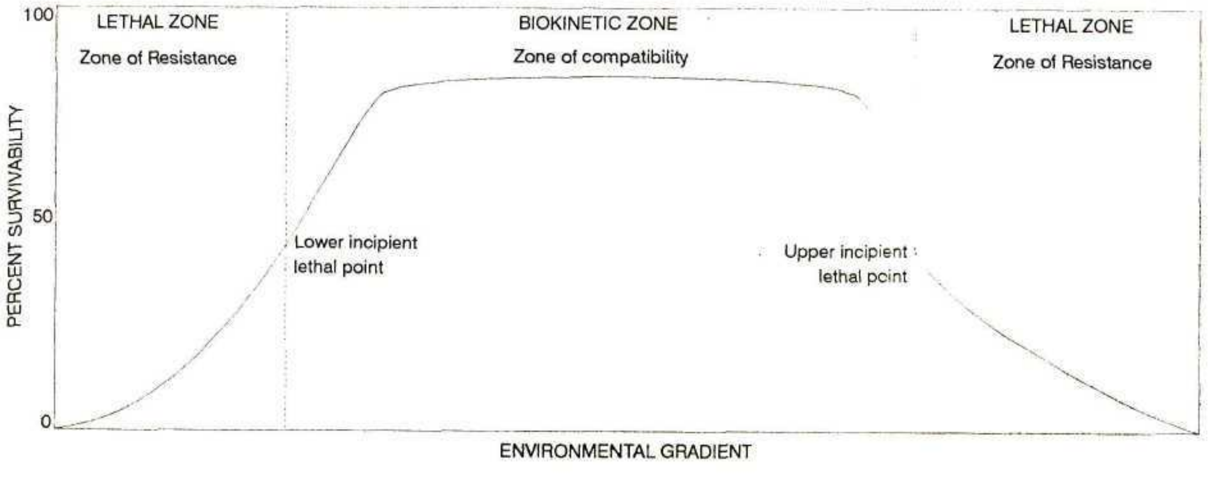


Figure 5.4 : The response of an estuarine organism to an environmental gradient (*after* Vernberg 1975)

In attempting to establish the response of *Upogebia africana* to high temperatures, Hill (1967) observed an interesting behavioural adaptation of the organism. *Upogebia africana* constructs burrows in fine mud, from which it circulates water (and therefore food and oxygen) from the overlying body of water through two or more entrances. When circulating water of 24°C above the burrows of aquarium *Upogebia africana*, the temperature of the surface water and that of the burrow were the same. If the temperature of the circulating water was increased to 27.5°C, the organism pumped the water at a slower rate, resulting in burrow temperatures which were slightly cooler than the overlying water. When overlying water temperatures were increased to 32°C, pumping became intermittent, lasting for three to five seconds after which it ceased for several minutes. As a consequence, only a small amount of water entered the burrow and the burrow temperature continued to remain below that of the surface water. After half an hour most of the aquarium prawns began to close down one of the entrances of the burrow by plastering mud around the holes. After an hour, the temperature in the burrows reached 25.5°C. Thus although *Upogebia africana* is capable of behavioural adaptations which prevent the organism from experiencing conditions outside of the biokinetic zone, these responses have a physiological cost. During these periods pumping through the burrow is restricted and consequently the organism's access to food and oxygen is limited, both of which have a direct effect on the metabolism and growth of the organism.

Hill (1967) thus showed that *Upogebia africana* were capable of buffering the effect of short term high temperatures. To investigate the effects of longer term exposure to higher temperatures, Hill (1967) exposed a group of prawns to a temperature of 31°C. Survival times varied from 2 hours to 108 hours and were found to be independent of sex or size of prawn. To eliminate the bias introduced by extremely resistant individuals, the median time of survival was used to describe the response of the sample in a further set of experiments. Prawns were exposed to temperatures ranging from 28°C to 40°C. Furthermore prawns which were collected in summer may exhibit greater resistance to the effects of higher temperatures due to acclimation, and consequently the experiments were repeated for prawns caught in both summer and winter. These data, including the 30%, 70% and median survival time are presented in Figure 5.5, and show that *Upogebia africana* will not survive at temperatures above 34°C, while at slightly lower temperatures there was an increase in survival time and an associated slower dying time (Figure 5.5).

Hill (1967) highlights a zone of transition between 28°C and 30°C, below which prawns were not killed by temperature. The upper lethal temperature of *Upogebia africana* was determined to be 29±1°C. Hill (1967) found that it was not possible to distinguish between winter and summer acclimatised prawns on the basis of their upper lethal temperatures.

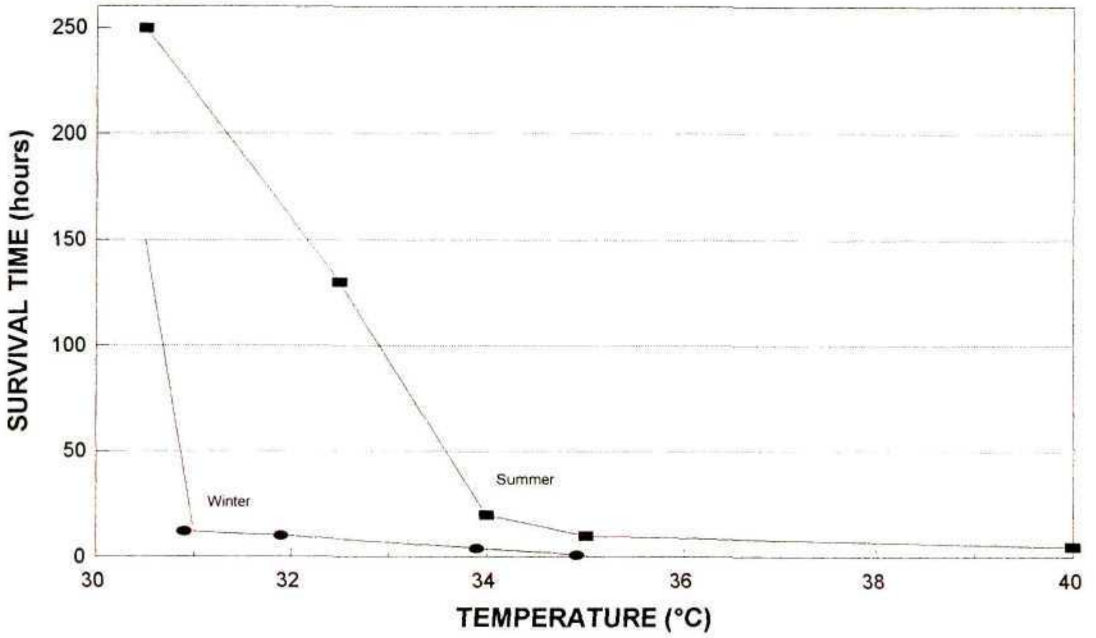


Figure 5.5 : Survival times of winter and summer *Upogebia africana* at a series of temperatures (*data from* Hill (1967)).

Hill (1967) also investigated the prawns living in the substratum of the power station cooling ponds at Knysna. These prawns are exposed to consistently high temperatures, and since the substratum is also heated, they cannot regulate burrow temperature by the mechanisms described above. Hill (1967) found that prawns collected from the warmer part of the pond were significantly smaller than those collected from the cooler parts of the pond. In addition the prawns from the heated pond showed a narrower zone of thermal resistance than those collected in summer from the Kowie estuary. Hill (1967) has argued that both observations are due to the '*deleterious effects of frequent exposure to high temperatures*'.

In support of the hypothesis that high temperatures result in smaller body size, Hill(1967) cites Day (1964), who found that prawns collected from Inhambane in Mozambique were only one third to one half of the size of those found at Langebaan, suggesting that temperatures towards the higher end of the organisms tolerance result in smaller adult prawns. Kinne (1970a) has observed that many marine invertebrates grow faster and reach sexual maturity at a smaller size in the warmer parts of their distribution than the cooler parts of their distribution. Hill (1967) has provided evidence to support this, showing that female prawns from the Kowie estuary breed at a slightly smaller carapace size than female prawns from the cooler Uilenkraal estuary located to the east of the Kowie estuary.

Hill (1967) established that the lower lethal temperature of *Upogebia africana* to be 5°C, but remarked that since this was well below the occasional occurrences of cold (10°C) water in Cape estuaries, temperature could not be considered a factor in the survival of adult prawns. As outlined above, temperature nevertheless affects the rate of growth of marine and estuarine invertebrates and it is therefore important to establish the possible consequences of lower temperatures for *Upogebia africana*.

Unfortunately, apart from the more qualitative observations outlined above, no data exist which relate temperature to the growth rate of *Upogebia africana.* However Hill (1967) did obtain data which describe the relationship between temperature and the respiratory rate of different size classes of prawns (Figure 5.6). Since the respiratory rate is an index of metabolic rate, is reasonable to assume that respiratory rates under different temperatures give an indication of how growth may be affected as a consequence of these temperatures. Hill (1967) reports that the prawns did not show full activity at a temperature of 8.3°C. All size groups however, did show a rapid increase in respiration between 12°C and 16°C, and a more gradual increase in respiratory rate between 16°C and 29°C. A sudden decrease in respiration rate occurs above 29°C. This conforms to Kinne's (1970a) observation that '*in general rates of metabolism and activity increase with increasing temperature over most of the species-specific temperature range tolerated and decrease suddenly near the upper lethal limit.*'

Based on the qualitative data described above, and the findings of Hill (1967) the effect of temperature on the maximum size age relationship as described by Equation 6, was considered to be defined by the function shown in Figure 5.7.

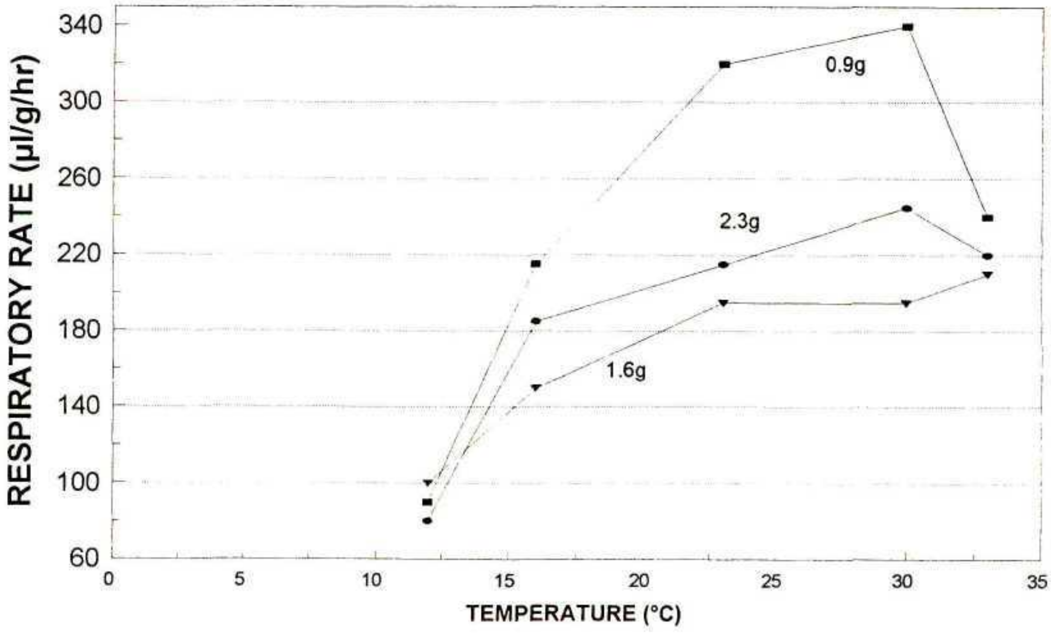


Figure 5.6 : Respiratory rates of three weight classes of *Upogebia africana*, determined at a series of temperatures (*after* Hill (1967)).

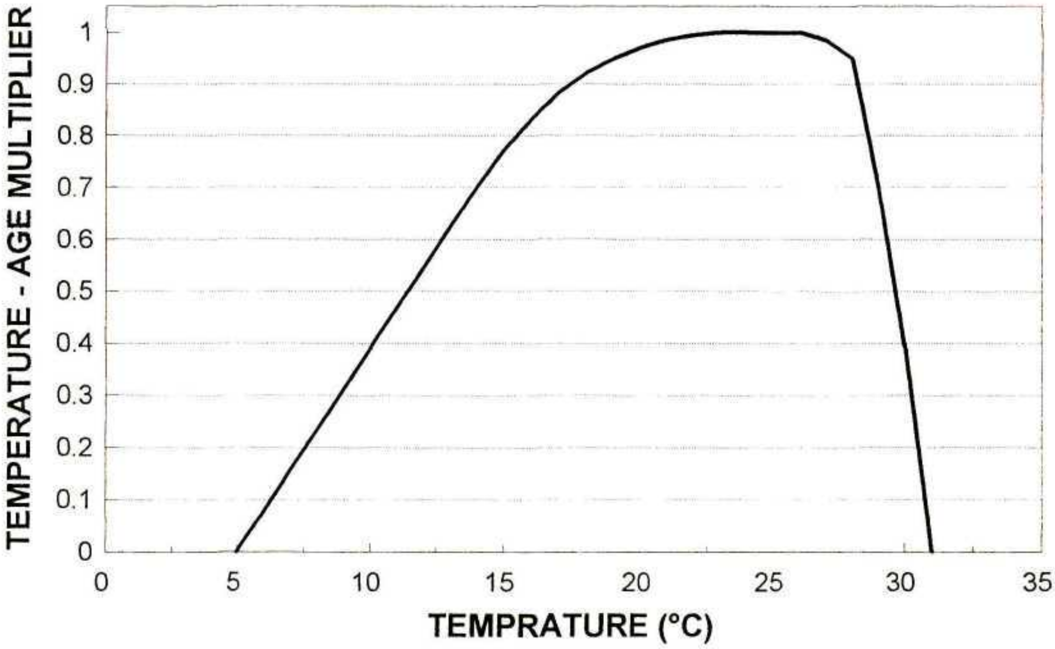


Figure 5.7 : Function describing the relationship between temperature and a multiplier which reduces the growth rate of *Upogebia africana* in non-ideal temperatures.

5.3.2 Effects of salinity on the mortality and growth of *Upogebia africana*

In comparison with the known effects of temperature on growth rates, relatively little is known about salinity effects, although in general it is thought that in most euryhaline invertebrates growth is restricted to significantly narrower salinity ranges than is survival (Kinne 1970b). In addition, Kinne (1970b) maintains that aquatic invertebrates generally respire most efficiently in salinities to which they are genetically adjusted.

In a series of experiments, Hill (1967) established that *Upogebia africana* could survive for extended periods at salinities as low as 1.7 $g.kg^{-1}$ After exposure of several groups of 10 prawns to different salinity conditions, Hill (1967) found that after 200 hours only one death at a salinity of 1.7 $g.kg^{-1}$. However, at exposure to 0.68 $g.kg^{-1}$, 5 prawns died; and at salinities of 0.34 $g.kg^{-1}$ all 10 prawns died in the same time period. Exposure to tap water resulted in rapid mortality, with all prawns dying within 50 hours. In a further series of experiments, mortality due to low salinities appeared to be independent of the sex or age of the prawn. Low salinities were found to delay moulting, although prawns could still successfully moult at salinities of 3.4 $g.kg^{-1}$. Deaths in salinities of 1.7 $g.kg^{-1}$ were directly subsequent to moulting.

Hill (1967) also examined the effects of salinity on respiratory rates. Respiratory rates of prawns were measured after exposure to various salinities for 24 hours, the results of which are shown in Figure 5.8. Lowering of the salinity below 35 $g.kg^{-1}$ thus resulted in an increase in the respiratory rate to a maximum at between salinities of 6 $g.kg^{-1}$ and 8 $g.kg^{-1}$. These results are supported by the observations of Kinne (1970b) who commented that "*in euryhaline invertebrates, rates of oxygen consumption and of related metabolic processes have been reported to be subject to changes, particularly in the salinity range 5 $g.kg^{-1}$ to 8 $g.kg^{-1}$*". Kinne (1970b) has noted that reduced salinity tolerances are often related to maximum metabolic rates, and in addition that changes in salinity may not only affect metabolic rates but also metabolic efficiency. This has been shown for euryhaline fish, but is also thought to hold for all aquatic invertebrates. For example Reeve (1963) showed that very high and very low salinities represented a physiological burden on developing *Artemia salina* embryos, manifesting in increased energy expenditure and therefore less growth. Based the above evidence, the relationship between salinity and growth was assumed to take the form of the function plotted in Figure 5.9 overleaf.

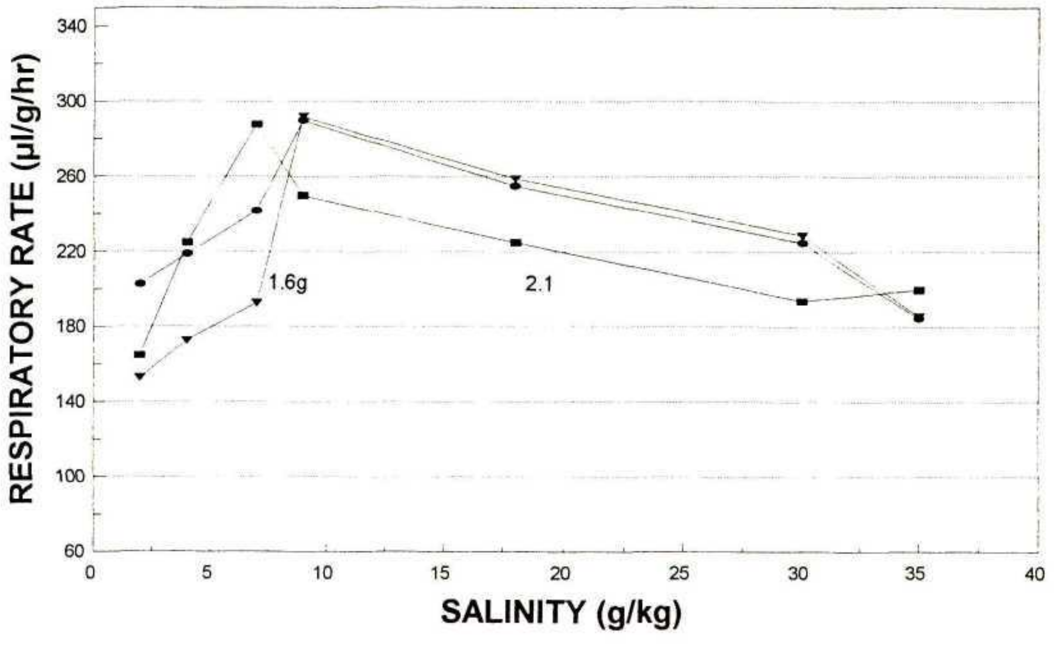


Figure 5.8 : Relationship between salinity and respiratory rate for 3 size classes of *Upogebia africana* (*after* Hill 1967)

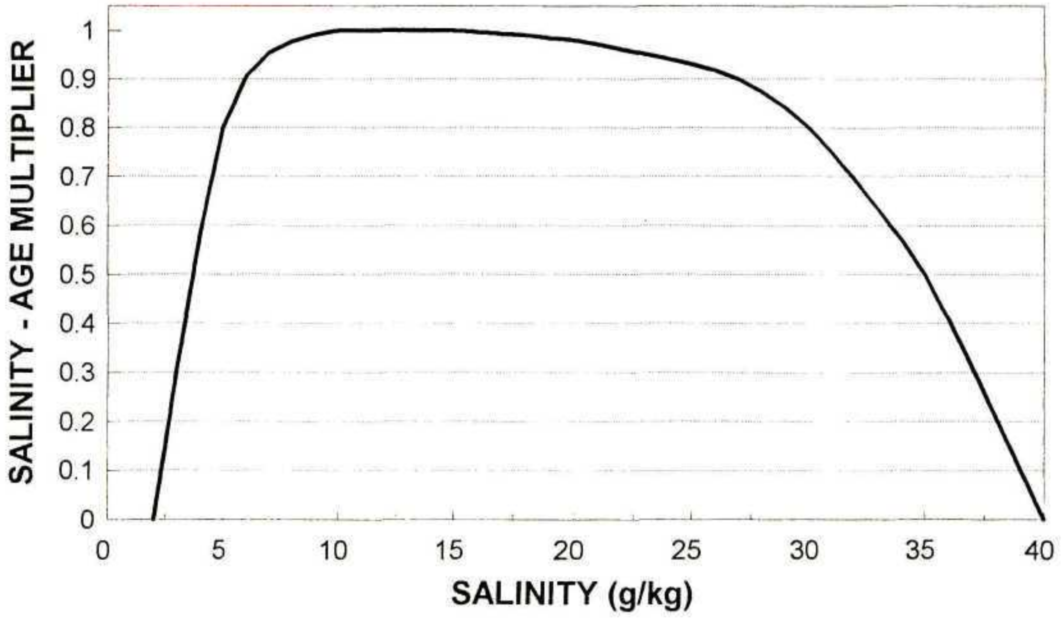


Figure 5.9 : Function describing the assumed relationship between salinity and a multiplier to account for the limiting effects of unsuitable salinities on the growth of *Upogebia africana*

5.3.3 The effects of an interaction of temperature and salinity on the growth and mortality of *Upogebia africana*

Hill (1967) showed that both temperature and salinity can act as separate and independent limiting factors affecting the growth rate and mortality of *Upogebia africana*. To establish the consequences of possible interactions between the two environmental factors, prawns were exposed to a range of salinities at a lethal temperature, and in addition, prawns were kept at a low salinity over a range of temperatures.

The results of these experiments indicated that survival in salinities above 1.7 $g.kg^{-1}$ was temperature independent, while at salinities below this there was a detectable decrease in survival time with increasing temperature. For example at 1.7 $g.kg^{-1}$ the upper lethal temperature appeared to be approximately 26°C, while at 0.5 $g.kg^{-1}$ the upper lethal temperature had dropped to 19°C, and at 0.17 $g.kg^{-1}$ it appeared to be even lower (Hill 1967).

When prawns were exposed to lethal temperatures at different salinities, prawns survived longer in water at 26 $g.kg^{-1}$ than in 35 $g.kg^{-1}$ seawater; supporting the assumption shown in Figure 5.9 suggesting that growth rates decrease in water with salinities in excess of 26 $g.kg^{-1}$. Furthermore Hill (1967) found that the blood oxygen concentration of prawns was constant in the range 7 $g.kg^{-1}$ to 26 $g.kg^{-1}$, suggesting that this range of salinity resulted in an optimum internal oxygen concentration. Hill (1967) thus concludes that although temperature has very little effect on the survival of prawns in non-lethal salinities (>1.7 $g.kg^{-1}$), salinity can have a large effect on survival at high temperatures. Thus, if *Upogebia africana* are subjected to very low salinities (e.g. for up to a week after a flood event), the upper lethal temperature limit is reduced from 29°C to possibly 19°C depending on how low the salinity dropped.

Ecophysiological studies of estuarine organisms (Blaber 1973) often represent the interaction of two environmental variables such as temperature and salinity as a series of curves defining 100 percent mortality and zero mortality. Combinations of temperature and salinity which lie within the envelope of zero mortality will result in no mortality whilst those lying in the outer envelope will experience mortality. Unfortunately such studies have yet to be undertaken for *Upogebia africana* and consequently the curves shown in Figure 5.10 overleaf have been inferred from the preceding discussion.

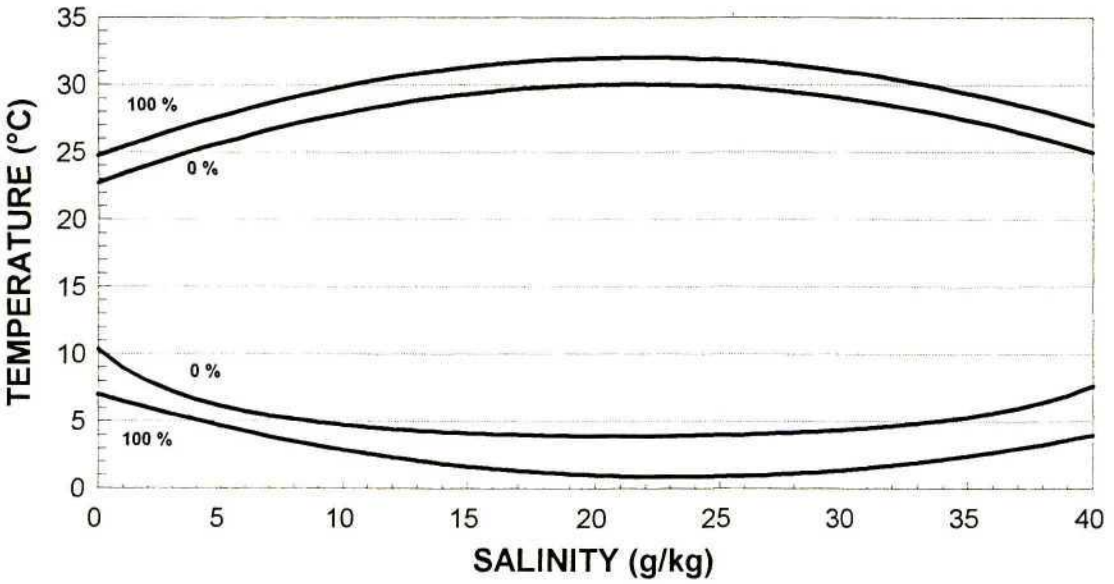


Figure 5.10 : Assumed mortality curves for *Upogebia africana,* showing the effects of different combinations of temperature and salinity on mortality.

Thus for the purposes of the model, the effects of temperature and salinity on the growth rate are assumed to be defined by the curves shown in Figures 5.7 and 5.9 respectively. These curves were determined by quadratic interpolation of selected points, from which two tables of multipliers corresponding to respective temperature and salinities were generated.

Mortalities arising due to lethal salinities, temperatures or lethal combinations of temperature and salinity are determined from the curves presented in Figure 5.10. Mortality arising from a given temperature and salinity is calculated as shown below :

$$PERMORT_t = \frac{(U100M - TEMP_t)}{(U100M - U000M)} \text{ If } TEMP_t \geq 15\,°C \tag{7}$$

$$PERMORT_t = \frac{(L000M - TEMP_t)}{(L000M - L100M)} \text{ If } TEMP_t < 15\,°C \tag{8}$$

where,

$PERMORT_t$ = Percentage of the population which dies at time t

$TEMP_t$ = Temperature at time t (°C)

$U100M$ = Upper temperature at which 100% mortality occurs (°C)

$U000M$ = Highest temperature at which no mortality occurs (°C)

$L000M$ = Lowest temperature at which no mortality occurs (°C)

$L000M$ = Lower temperature at which 100% mortality occurs (°C)

The upper and lower temperatures referred to above are determined for a given salinity, as shown in Figure 5.10 above, and expressed in the quadratic functions below :

$$U100M = 24.719298 + (0.67105263 \times SALIN_t) - (0.015350877 \times SALIN_t^2) \qquad \textbf{(9)}$$

$$U000M = 22.719298 + (0.67105263 \times SALIN_t) - (0.015350877 \times SALIN_t^2) \qquad \textbf{(10)}$$

$$L000M = 9.005848 - (0.5254386 \times SALIN_t) + (0.01125731 \times SALIN_t^2) \qquad \textbf{(11)}$$

$$L100M = 7.005848 - (0.5254386 \times SALIN_t) + (0.01125731 \times SALIN_t^2) \qquad \textbf{(12)}$$

where,

$SALIN_t$ = Salinity at time t ($g.kg^{-1}$)

5.4 REPRODUCTION AND FECUNDITY

5.4.1 Size at sexual maturity

The size at which *Upogebia africana* reaches sexual maturity has been reported by several authors, and varies from location to location. For example Siegfried (1962) indicated that *Upogebia africana* in the Uilenkraal estuary reached sexual maturity at a carapace length of 11 mm, Hill (1967) reported a carapace length of 10 mm for the Kowie estuary, and Hanekom (1980) provides a size of 12 mm and 17 mm for two locations within the Swartkops estuary. In a subsequent review of these data, and based on the von Bertalanffy growth equations formulated by Hanekom (1980), Hanekom and Baird (1992) suggest that sexual maturity is attained after 18 to 20 months. In the formulation of the model sexual maturity is assumed to occur at a physiological[2] age of 18 months. As established above, the actual size of the prawn at this point will depend on several factors including the temperature and salinity regime the prawn has been subject to, as well as the quality and quantity of marine detrital input available as food.

5.4.2 Size and fecundity

Hanekom (1980) has established a positive correlation between the carapace size of female prawns and the number of eggs produced. This relationship is described by Equation 12 below, and is shown below in Figure 5.11:

$$EGGS_t = 0.0224 L_t^{3.8584} \tag{13}$$

where,

$EGGS_t$ = number of eggs produced at time t

L_t = carapace length at time t (mm).

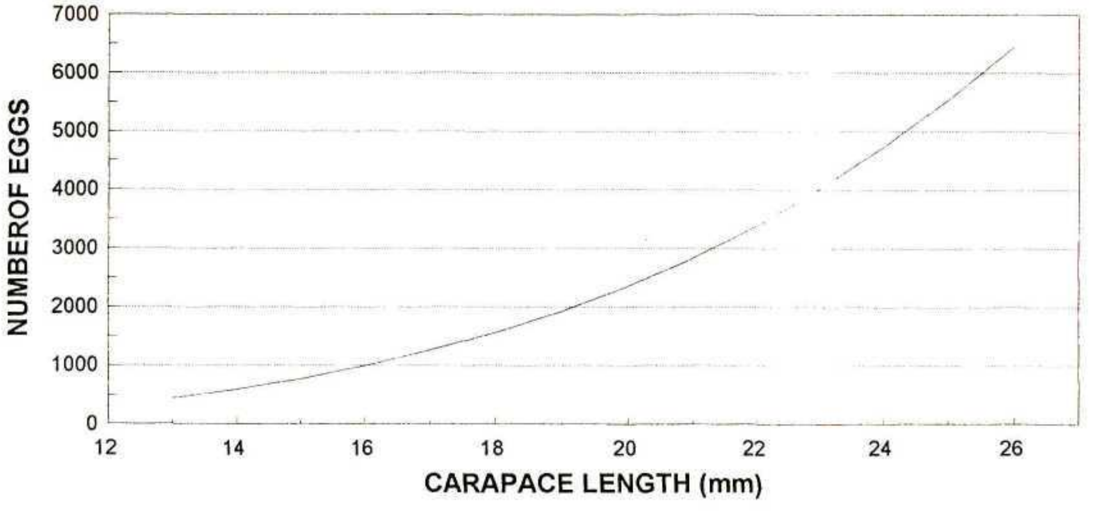


Figure 5.11 : Relationship between number of eggs carried and the size of the female *Upogebia africana* (*after* Hanekom 1980)

2 Two concepts of age are incorporated in this model. Firstly *chronological age* refers to the actual number of days the organism has lived. Secondly, the concept of *physiological age* incorporates the retarding effects of undesirable temperatures and salinities. Further details are presented in the final section of this Chapter.

5.4.3 Effect of temperature on egg development

Hill (1967) found that female prawns in the Kowie river carried eggs for 24 to 30 days, while Siegfried (1962) reports eggs being carried for 40 days, albeit in lower salinity. The rate of embryonic development is thought to be strongly temperature dependent (Kinne 1970a), and as a consequence Hill (1977) investigated the effect of temperature on egg development. The regression of incubation time on temperature as established by Hill (1977) is given below in equation 12.

$$log\ DTIME = 4.037 - 1.881\ log\ TEMP_t \qquad \textbf{(14)}$$

where,

$DTIME$ = estimated development time (days)

$TEMP_t$ = temperature at time t (°C).

Although salinity is also thought to affect the rate of egg development in other crustaceans (Hill 1967), in the case of *Upogebia africana*, no supporting data are available and consequently the possible effects of salinity on egg development were excluded.

5.4.4 Breeding season

According to Wooldridge and Loubser (1996), *Upogebia africana* is capable of reproduction during most months of the year, although the onset of the breeding cycle is usually associated with post mid-winter rising water temperatures (Hill 1977; Hanekom & Erasmus 1989). Peaks in the breeding season have been noted, although these vary between estuaries, and possibly even within an estuary (Wooldridge & Loubser 1996). For example data from the Swartkops (Hanekom 1980) and Kowie (Hill 1977) estuaries suggest both spring and summer peaks, whereas the Knysna and Uilenkraal estuaries show a single summer maximum (Hill 1977). Data obtained from the Great Brak estuary (Wooldridge 1996, *pers. comm.*), suggests a unimodal peak in December - February, decreasing during the winter months.

In the absence of any known relationship between temperature and the onset of breeding, the proportion of ovigerous females in the estuary at any one time was estimated from the sine function shown in Figure 5.12.

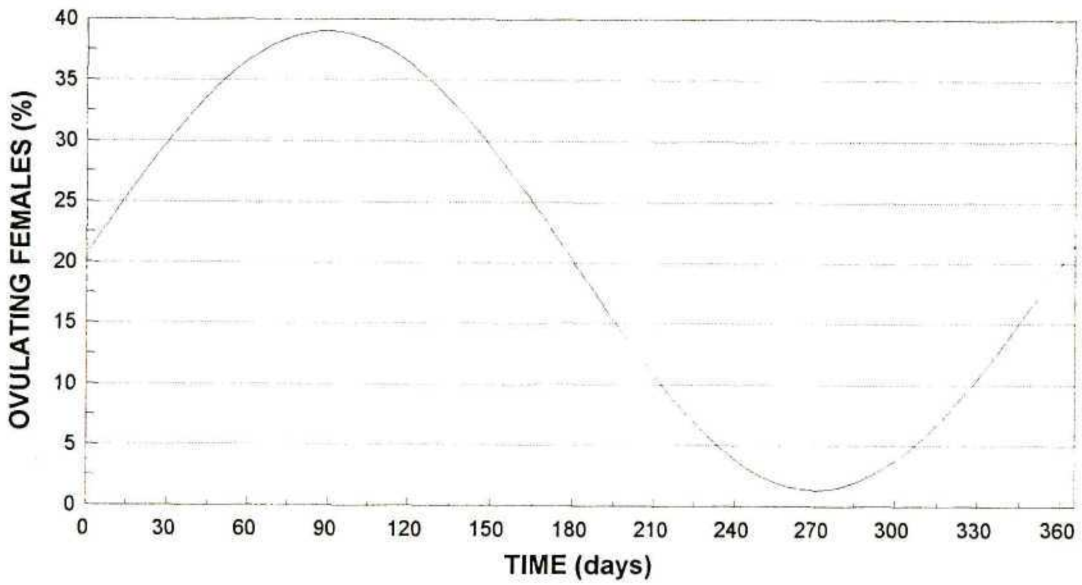


Figure 5.12 : Proportion of adult females assumed to be ovigerous for a given time of year. Year commences on 1 October.

5.5 HATCHING AND LARVAL DEVELOPMENT

Hatching of eggs is abrupt, occurring *en masse* at night within the estuary, in response to environmental cues which are currently being determined (Wooldridge 1996, *pers. comm.*). Sensitivity of newly hatched larvae to temperature and salinity conditions within the estuary are not known, and for the purposes of the model, mortality is assumed to occur in relation to adult tolerances. The larval development of *Upogebia africana*, has been summarised in Section 5.2, and makes reference to the obligate marine phase of larval development. Consequently, for the purposes of the model, larvae which are not exported to the marine environment die within 24 hours (Wooldridge 1996, *pers. comm.*).

Although the association between the lunar tidal cycle and the export of zoea and recruitment of postlarvae has been known for some time (Wooldridge 1991, 1994), recent research provides evidence of more complex behaviour. Wooldridge and Loubser (1996) have established that maximum export of zoea is synchronized to crepuscular high water in the estuary, rather than the maximum tidal amplitude associated with spring tide itself, as previously thought. Similarly, recruitment back into the estuary occurs after neaps when low water at sea is crepuscular. Thus as a result of the asynchronicity between the occurrence of spring and neap tides at dusk, maximum release of larvae would occur at spring tides at the beginning of winter, whereas in summer maximum release of larvae will occur three to four days after spring tides, when the high tide occurs at dusk (Wooldridge 1996, *pers. comm.*). Maximum incursion of postlarvae, on the other hand, will occur nearer neaps in winter and three to four days after neaps in summer. Rather than attempting to include this asynchronicity into the model, maximum recruitment and export was assumed to occur at neap and spring tides respectively. For the purposes of estuary management however, it should be noted that artificial mouth breachings should be undertaken with the above in mind.

Wooldridge and Loubser (1996) argue that the evolution of such a life history must be functionally advantageous. For example, relative to estuaries, the marine environment is stable with regard to temperature and salinity such conditions may be better suited to larval development. A further reason, also related to the widely fluctuating conditions in estuaries has been suggested by Wooldridge and Loubser (1996). They argue that the repopulation of estuaries after extreme events such as floods, or the occurrence of hypersalinities is "*likely to be expedited from a natural reservoir of larvae present in the marine environment, particularly on an open coastline where open inlets are often separated by relatively long distances*."

The fate of larvae at sea is not well understood. It is not known, for example, whether larvae remain in the nearshore environment near the estuary they originated from, or wether they are more widely dispersed. For this reason, and the uncertainties relating to predation in the marine environment, the model has not assumed a relationship between the numbers of zoea exported and the numbers of postlarvae recruiting into estuaries. As outlined above, export of zoea to the marine environment may be a survival strategy for the purpose of ensuring that there is an ample reservoir of postlarvae in the nearshore environment to sustain populations in estuaries. For this reason, the number of zoea exported to sea is considered a key output of the model. In considering recruitment, on the other hand, it is assumed that there are sufficient postlarvae in the nearshore marine environment to sustain the needs of an estuary. Recruitment into an estuary is thus unlimited, and occurs continuously when the basic requirements of an open mouth and neap tides coincide.

5.6 HABITAT AVAILABILITY AND DENSITY DEPENDENT MORTALITY

In the formulation of the model, it is assumed that mortality in the estuarine population of *Upogebia africana* occurs in response to exposure to unfavourable salinities and temperatures (Section 5.3.3). However as recruitment is not limited, the possibility of overcrowding in the estuary exists, particularly during periods of favourable temperature and salinity.

The distribution of *Upogebia africana* within an estuary is limited to those areas where the substratum is suitably fine. In areas of fine mud, populations of *Upogebia africana* may approach densities of up to 500 prawns per m^2 (Hanekom & Baird 1992; CSIR 1990). Competition between adult *Upogebia africana* of different size classes, as well as competition between and within size classes of juvenile *Upogebia africana* has not been studied. Nevertheless it is reasonable to assume that populations of *Upogebia africana* are limited by habitat availability, and for the purposes of the model density dependent mortality is assumed to occur in relation to the function shown in Figure 5.13. In the absence of further information regarding competition, density dependent mortality is assumed to occur uniformly across all size classes. The population density at any one time is determined from the habitat modelling approach discussed in the previous chapter, whereby a given water level determines total available potential habitat.

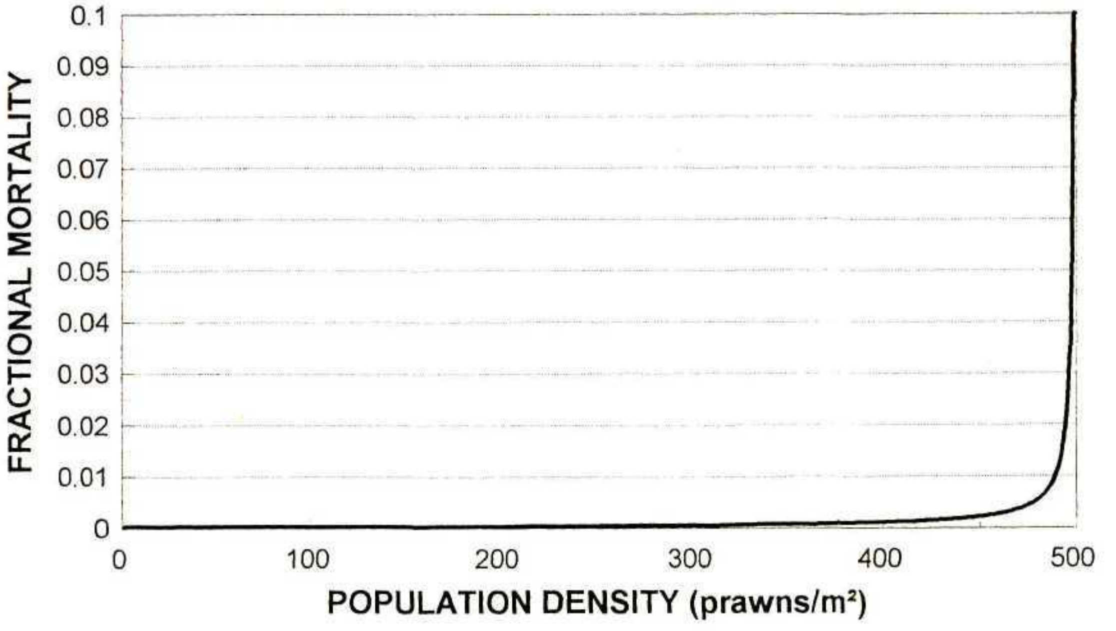


Figure 5.13: Assumed density dependent mortality in populations of *Upogebia africana*

5.7 MODELLING *Upogebia africana* POPULATION DYNAMICS

Sections 5.2 to 5.6 above describe the life history and growth requirements of *Upogebia africana*, presenting both empirical data and simplified assumptions characterising the behaviour of the organism in relation to changing environmental factors. This section attempts to integrate the above by outlining the structure and formulation of a model to predict the population dynamics of *Upogebia africana.*

5.7.1 Basic approach

As noted by Botsford (1992), over the past decade there has been a rapid growth in the number of population models which explicitly account for attributes of population structure such as size and age. Botsford (1992) argues further that our rapidly increasing understanding of population dynamics is primarily due to the fact that these types of models account for the fact that individuals at different ages, sizes, stages and location may respond differently to environmental gradients and hence contribute differently to processes such as reproduction. In addition, at the level of ecosystem management, population attributes such as size class distribution are frequently used as indices of ecosystem health (CSIR 1990, 1992). For this reason, and the fact that the recruitment of a group of *Upogebia africana* occurs as a discrete event, an approach whereby the growth and mortality of individual cohorts comprising the total population are modelled, was selected.

Physiologically based growth models of individual fish (Kitchell, Stewart & Weininger 1977; Stewart & Binkowski 1986) have been widely used and rigorously tested (Rice & Cochran 1984). These models estimate growth from consumption, based on an understanding of the physiology of the fish as a function of their body size and temperature (Crowder *et al.* 1992). In the absence of a detailed understanding of the physiology of dace (*Leucisus leucisus*), and uncertainty relating to feeding and consumption, Hearne (1996, *pers. comm.*) related fish growth to the single environmental variable of temperature, in an approach similar in concept to that of 'degree days' or 'heat units' frequently used in phenological modelling (Waggoner 1974). Under ideal temperature conditions the fish would grow at a maximum rate, and the size of the organism at any time would be determined directly from a relationship relating size and age. Under non-ideal conditions the growth rate is slower, represented in the model by a reduction in the rate of ageing. This concept of differential ageing gives rise to two terms used in this discussion; chronological age and physiological age. The former refers to the actual age of the organism whilst the latter is an expression of the nett effective increase in maturity due to unfavourable conditions.

This model extends the approaches described above by attempting to include the interactive effects of both temperature and salinity on the growth and development of an invertebrate. Unlike the concepts of 'degree days' and 'heat units' which generally attempt to relate the cumulative effects of temperature to the onset or completion of a phase of development, the approach taken in this model incorporates the cumulative effects of unfavourable environmental conditions on the rate of growth and final dimensions of an organism.

5.7.2 Formulation of the model

As discussed in Section 5.2, populations of *Upogebia africana* are maintained by a process of recruitment of postlarvae into the estuary from the sea. Recruitment of a group of individuals, hereafter referred to as a cohort, occurs as a discrete event following spring tides and provided the mouth is open. Cohorts recruiting into the estuary in such an event are numbered sequentially from 1, 2, 3... to i, with i denoting the cohort which recruited during the i th event.

Let

$POP_i\,(t)$ denote the number of individuals in cohort i at time t,

$CAGE_i\,(t)$ denote the average chronological age of individuals in cohort i at time t,

$PAGE_i\,(t)$ denote the average physiological age of individuals in cohort i at time t.

The rate of change in the chronological age of cohort i, is defined simply as :

$$\frac{dCAGE_i}{dt} = 1 \tag{15}$$

The rate of change in the population of cohort i is given by :

$$\frac{dPOP_i}{dt} = -mortality_i \tag{16}$$

where mortality is due to both density effects and lethal combinations of temperature and salinity, given by;

$$mortality_i = stmortality + densmortality \qquad \text{if } CAGE_i < 4 \text{ years} \qquad \textbf{(17)}$$

and where,

$$\begin{aligned} stmortality &= POP_i \times stmort_i \\ densmortality &= POP_i \times densmort_i \end{aligned} \qquad \textbf{(18,19)}$$

Note that cohorts live to a maximum chronological age of four years, so POP_i is set to zero at time $CAGE_i = 4$ years

$stmort_i$ = is a function of temperature and salinity, as defined in Figure 5.10 and equations 7 to 12

$densmort_i$ = density dependent mortality as defined in Figure 5.13

The rate of change in the physiological age of individuals in cohort i is given by :

$$\frac{dPAGE_i}{dt} = sam \times tam \qquad \textbf{(20)}$$

where,

sam = salinity age multiplier, as defined in Figure 5.9, and

tam = temperature age multiplier, as defined in Figure 5.7

The size of individual prawns within a cohort is a function of the physiological age of the cohort and is given by Equation 6, shown in Figure 5.3. Thus under ideal conditions the prawns age one day per day and the rate of change of chronological age (Equation 6) thus equals the rate of change of physiological age (Equation 20). The initial values of these equations are given by :

$$POP_i\ (t_i) = recruitment_i$$

$$PAGE_i\ (t_i) = 0$$

$$CAGE_i\ (t_i) = 0$$

where t_i is the time of the i th recruitment event, and where,

recruitment = number of individuals in a cohort recruiting into the estuary.

The total population of *Upogebia africana* within an estuary at time t is therefore;

$$UPOGEBIA_t = \sum_i POP_i \tag{21}$$

5.7.3 Formulation of management indices

Although the most simple index of estuary functioning might be a measure of total population of *Upogebia africana* expressed over time (e.g. mean monthly or average annual population numbers), a more useful performance index may be the performance of the population over time relative to a reference state. Assuming the total population under specified conditions is known, and denoted by $RUPOGEBIA_t$, then an alternative index would be :

$$INDEX1_t = \frac{UPOGEBIA_t}{RUPOGEBIA_t} . 100 \tag{22}$$

where,

$INDEX1_t$ = performance index 1 at time t

However, this index only provides an indication of the numbers of prawns present at time t relative to the number of prawns which would have been present under the specified conditions. As the number of prawns present at time t is only an indicator of recruitment opportunity in the absence of lethal temperatures/salinities, a normalised, cohort weighted index is suggested. In the calculation of this index, the biomass of the i th cohort is determined by calculating the mass of individuals in the cohort and multiplied by population number of the i th cohort. In other words the biomass of the i th cohort is determined and then summed over the cohorts. The second management index expresses this value relative to the biomass of *Upogebia* under specified reference conditions. Thus,

$$INDEX2_t = \sum \frac{(PMASS_i . POP_i)}{(RMASS_i . RPOP_i)} \quad \textbf{(23)}$$

where,

$PMASS_i$ = mass of an individual in cohort i

POP_i = number in cohort i

$RMASS_i$ = mass of an individual in of cohort i of population under reference conditions

$RPOP_i$ = number in cohort i of in population under reference conditions

and where

$$PMASS_t = 0.0721 L_t^{3.1524} \quad \textbf{(24)}$$

where,

L_t = size of the individual determined from the physiological age

Similarly $RMASS_i$ would be determined from the size of individuals under reference conditions. The above two indices reflect performance of *Upogebia africana* within the estuary. As mentioned in Section 5.5, the export of zoea into the marine environment is also of importance. Figure 5.11 defines a relationship between the carapace size of female prawns within a cohort, and the mass of eggs produced by the female in one brood. Assuming that 50% of the population is female, and the proportion of the female population which is ovigerous at any one time is described by Figure 5.12, then the mass of eggs produced by a population is given as:

$$TOTEGGS_t = \sum_i (POP_i \times 0.5 \times ovig_t \times eggs_i) \qquad \textbf{(25)}$$

where,

$ovig$ = proportion of the adult female population ovigerous at time t

$eggs_t$ = mass of eggs produced at time t by a female in cohort i

The final management index thus takes a similar form to index 1 above, namely;

$$INDEX3_t = \frac{TOTEGGS_t}{RTOTEGGS_t} \qquad \textbf{(26)}$$

where

$TOTEGGS_t$ = total number of eggs produced by population at time t

$RTOTEGGS_t$ = total number of eggs produced by population under reference conditions at time t

6 APPLICATION OF THE MODELLING APPROACH

6.1 INTRODUCTION

All three approaches discussed in the preceding Chapters will be applied in a scenario based case study of the Great Brak estuary. The scenarios address a variety of flow conditions ranging from estimated pristine natural runoff to the long term mean runoff expected as a result of the construction of the Wolwedans dam, as well as disturbance scenarios involving a reduction in winter runoff and a flood. Prior to applying the models in a case study, the data requirements will be discussed and some of the difficulties associated with model testing and verification will be addressed. Where possible, limited sensitivity analyses will be undertaken.

6.2 PREDICTING ESTUARY HYDRODYNAMICS AND MOUTH CONDITION

Section 1.6 makes reference to the need for two levels of prediction, the first of which predicts the consequences of changes in freshwater inflow for the structure and character of the estuary, including the likelihood of mouth closure. Simulation of the physical characteristics of estuaries at the scale of days and months is readily achieved using commercially available, one dimensional hydrodynamic simulation programmes such as Mike 11 (Danish Hydraulic Institute 1992). Such software packages generally require bathymetric cross-sections, water level recordings and river inflow as input data, and are capable of providing time histories of water levels, flow velocities and even salinity through modelling of hydrodynamic and transport-dispersion processes (Huizinga 1994). The computational approach of Mike 11 is based on implicit finite difference techniques to solve the St Venant equations, with simulation timesteps of between 0.5 and 3 minutes.

As a consequence simulation periods of days to months are manageable, while simulation periods of a year or more require major computational effort. In addition ecological processes are seldom modelled at the same time scale as hydrodynamic processes. A water release policy, for example, may extend over a full year, the ecological effects of which may only be evident in the following breeding season. Booij and Sokolewicz (1990) and Holthuijsen and Booij (1990) have emphasised the hiatus between the scale of hydrodynamic and ecological processes, and the consequent difficulties for parallel modelling of these processes.

For the purposes of estuary management, ecologically based models need to be run over a five to ten year simulation period, requiring hydrodynamic information for a similar length of time. In order to meet these needs, Slinger (1994, 1995, 1996) has developed a generalised, mechanistic model of estuarine physical dynamics (Estuarine Systems Model). The purpose of this model is to predict the state of the estuary mouth given freshwater inflow and the nearshore wave condition, and provides as output; average salinity, water level and tidal flux as well as the longitudinal salinity difference.

Although the Estuarine Systems Model only provides salinity for Station R2 (near the estuary mouth), the Mike 11 model can be used to obtain salinity at other locations. Figure 6.1 shows, for various inflow velocities, the relationship between salinity predicted by the Mike 11 model at both Station R2 and R4, the latter being the most appropriate for the region of the estuary in which *Upogebia africana* occurs (Figure 6.2). Salinity predicted by Slinger (1995) was thus adjusted on the basis of this relationship.

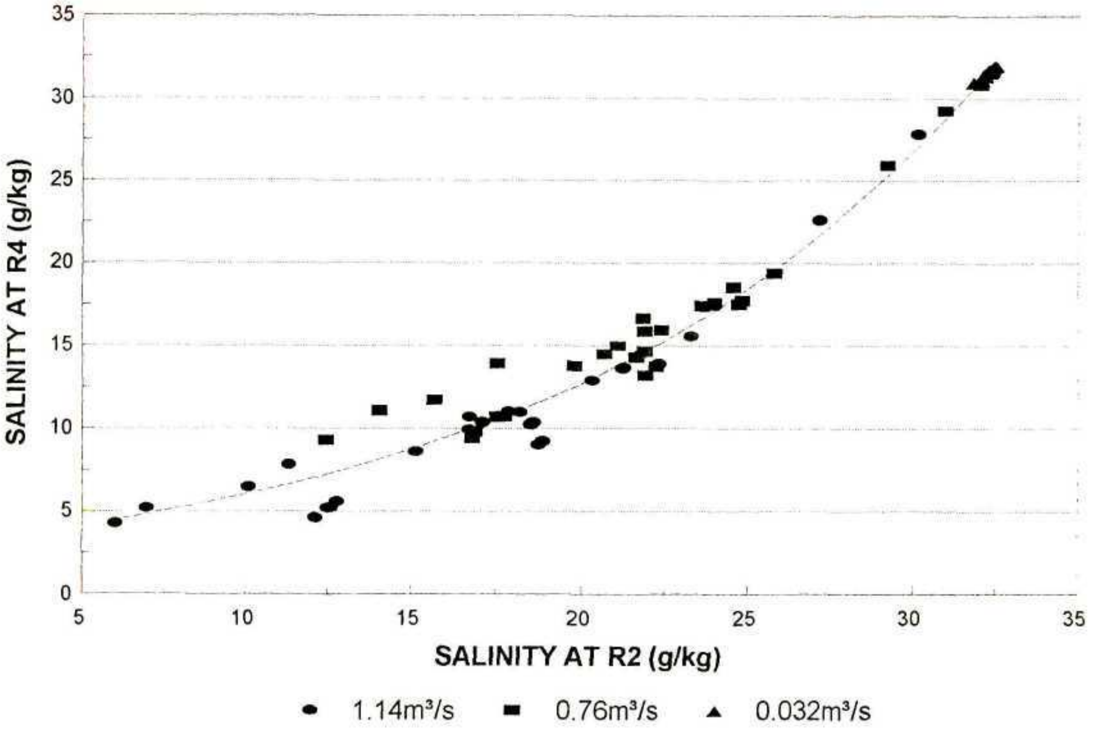


Figure 6.1 : Relationship between salinity predicted by the Mike 11 model at stations R2 and R4 in the Great Brak estuary, for a variety of inflow conditions.

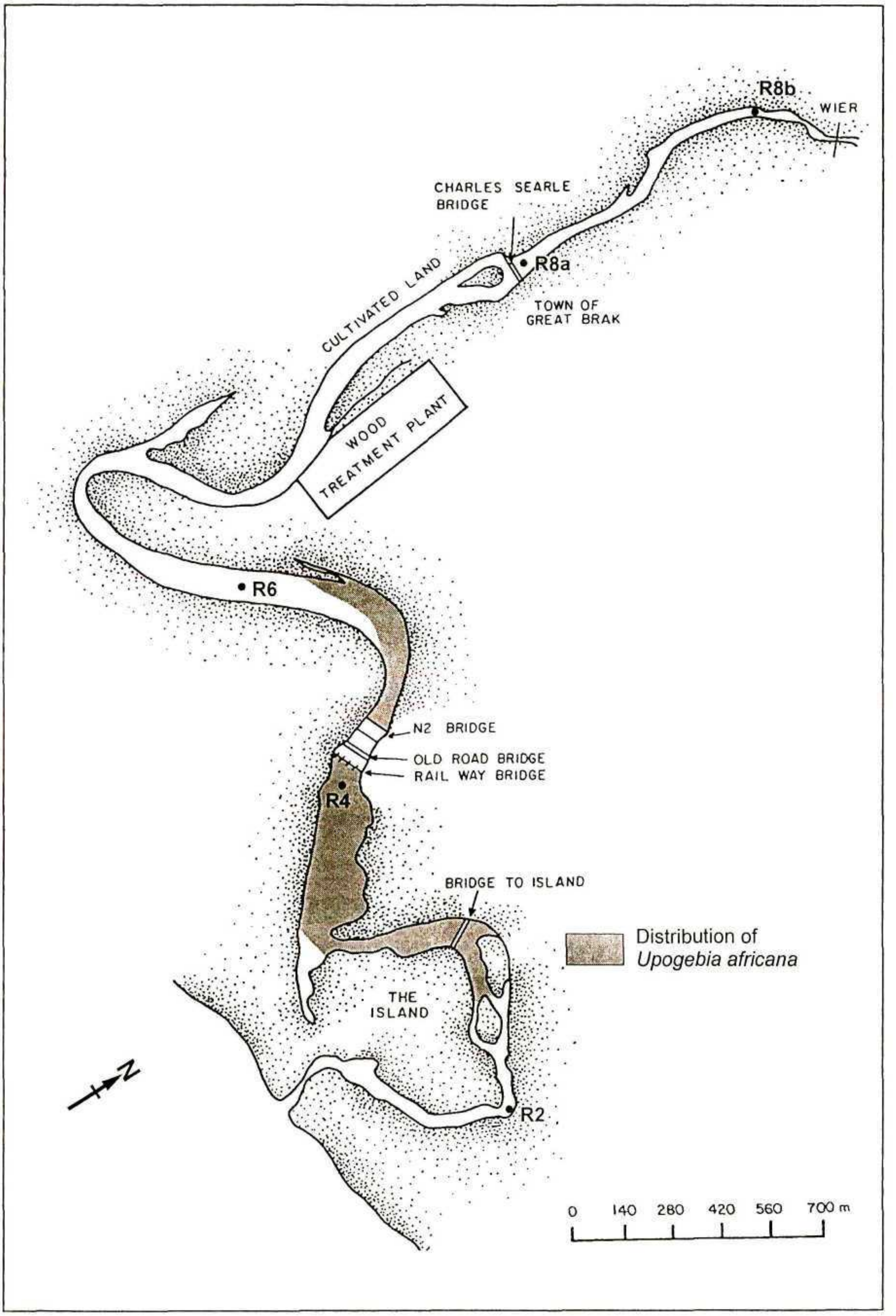


Figure 6.2 : Location of stations in the Great Brak estuary in relation to the distribution of *Upogebia africana* (CSIR 1990).

Unfortunately the Estuarine Systems Model cannot predict estuary water temperature, and although it is possible to obtain simulated water temperature from the Mike 11 model, water temperature has not been regularly recorded at the Great Brak estuary and consequently no calibration of this element of the water quality module of Mike 11 has been undertaken (Slinger 1994). Water temperatures for the estuary are known to vary seasonally between 19°C and 25°C (Slinger 1995 *pers. comm.*), and therefore for the purposes of this study, water temperature is assumed to fluctuate according to the sine function shown in Figure 6.3, with a random daily variation of 2°C on either side of the predicted value.

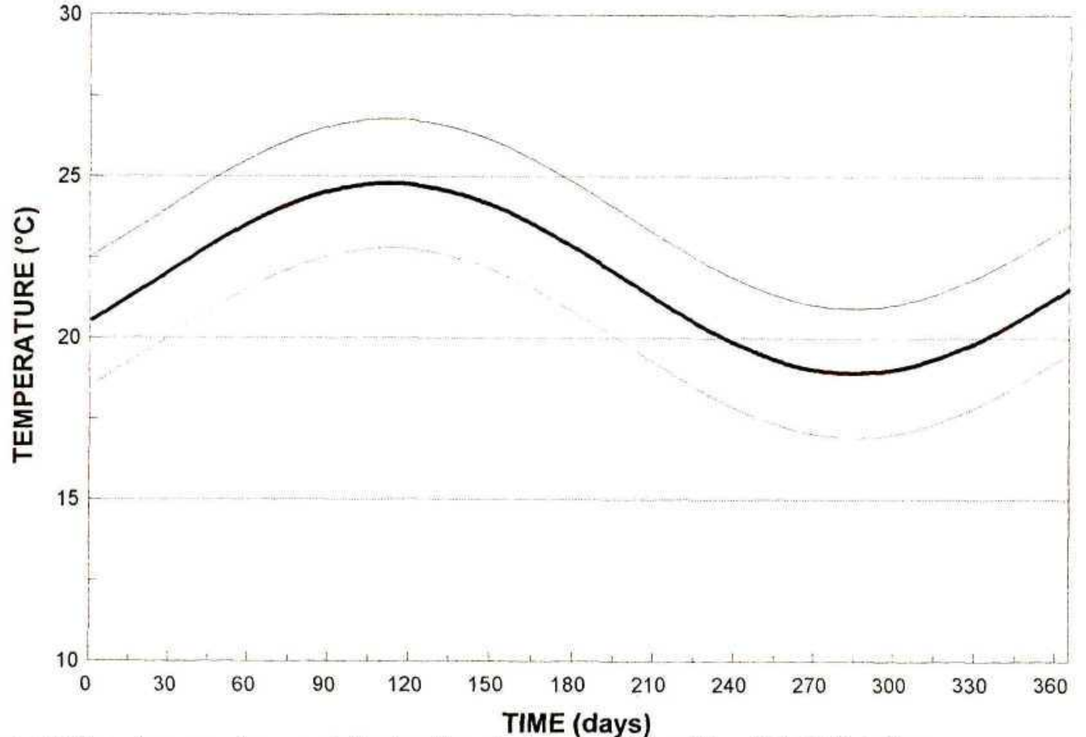


Figure 6.3 : Assumed annual fluctuation in temperature. Day 1 is 1 October.

6.3 FRESHWATER INFLOW SCENARIOS

The Estuarine Systems Model (Slinger 1995) was thus used to generate the input data necessary for the modelling approaches described in this thesis. Four sets of scenarios are described below, representing a progressive reduction in freshwater inflow to the estuary. These scenarios are based on hydrological modelling of the pre- and post- impoundment river flows. In addition, two sets of disturbance scenarios are presented, whereby the four scenarios referred to above are subjected to a flood and drought respectively.

6.3.1 Natural runoff scenario (Scenario 1)

The natural runoff scenario (Figure 6.4) refers to runoff prior to any modification through the construction of any dams or abstraction for irrigation, with a mean annual runoff of 34.7 x 10^6 m^3 per year. According to Slinger (1995) under natural runoff conditions the estuary mouth would be predominantly open, and would have remained open all year between 25 and 50 % of a long term (>15 years). A closed mouth would have occurred on average between one and two months per year, but would have rarely closed for more than one month at a time.

6.3.2 Pre-dam runoff scenario (Scenario 2)

This scenario represents the reduction in runoff attributable to the construction of farm dams and consumptive use by irrigation and afforestation (Figure 6.5). Under the pre-dam scenario, the mouth would have been open between 50 and 75 % of the time, with the mouth closing for 3 to 5 months during dry periods. Approximately once every 20 years the mouth would have been open for less than 40% of a year (Slinger 1995). In this scenario mean annual runoff is considered to be of the order of 24.5 x 10^6 m^3 per year.

6.3.3 Post-dam scenarios (Scenario 3a and 3b)

It is anticipated that the estuary mouth would be open for between 30 and 50% of the time, and once every 25 years the mouth would be open for 75% of the time (Figures 6.6 and 6.7). Once in every 4 years the mouth will be open for between 1 and 3 months. The current assured allocation to the estuary is 2 x 10^6 m^3 per year, and the dam is not expected to overflow. For this scenarios two baseflow conditions were described :

(a) An annual freshwater inflow of 10 x 10^6 m^3 per year, distributed as seasonal baseflow.

(b) An annual freshwater inflow of 2 x 10^6 m^3 per year, distributed as seasonal baseflow.

6.3.4 Disturbance scenarios

In addition to the four scenarios described above, an additional set of two disturbance scenarios were applied to each of the above. The first set of disturbance scenarios consists of a summer flood event equivalent to the 1 in 50 yr flood volume superimposed on each of the runoff scenarios. The second set of disturbance scenarios represents a 50% decrease in freshwater inflow to the estuary over the period March to June. Figures 1 to 8 of Appendix A show the consequences of these disturbance scenarios for the hydrodynamic character of the estuary.

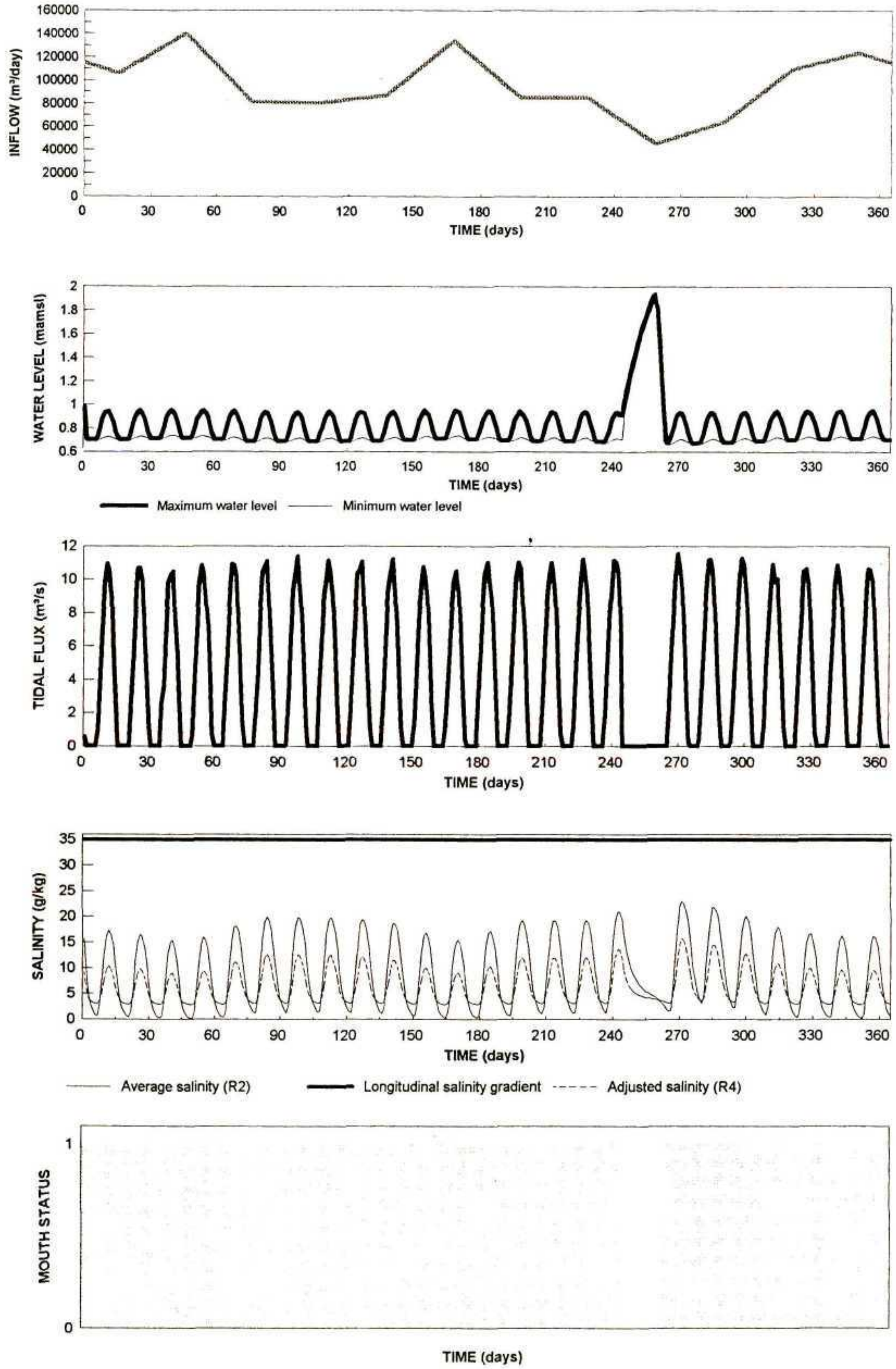


Figure 6.4 : Flux in key characteristics of the estuary environment in relation to freshwater inflow anticipated under natural conditions (Scenario 1)(Slinger 1995)

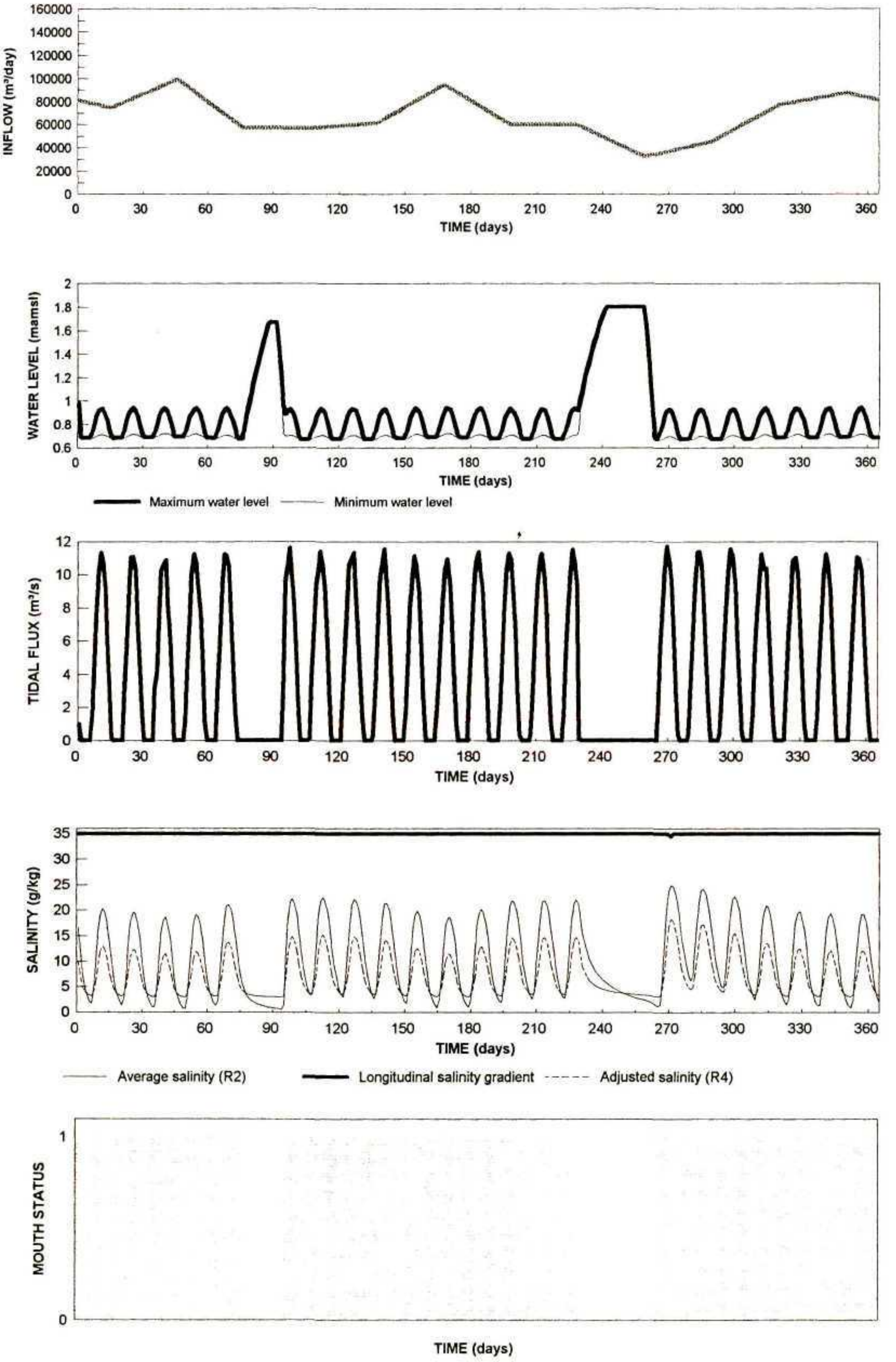


Figure 6.5: Flux in key characteristics of the estuary environment in relation to freshwater inflow anticipated under pre-dam conditions (Scenario 2)(Slinger 1995)

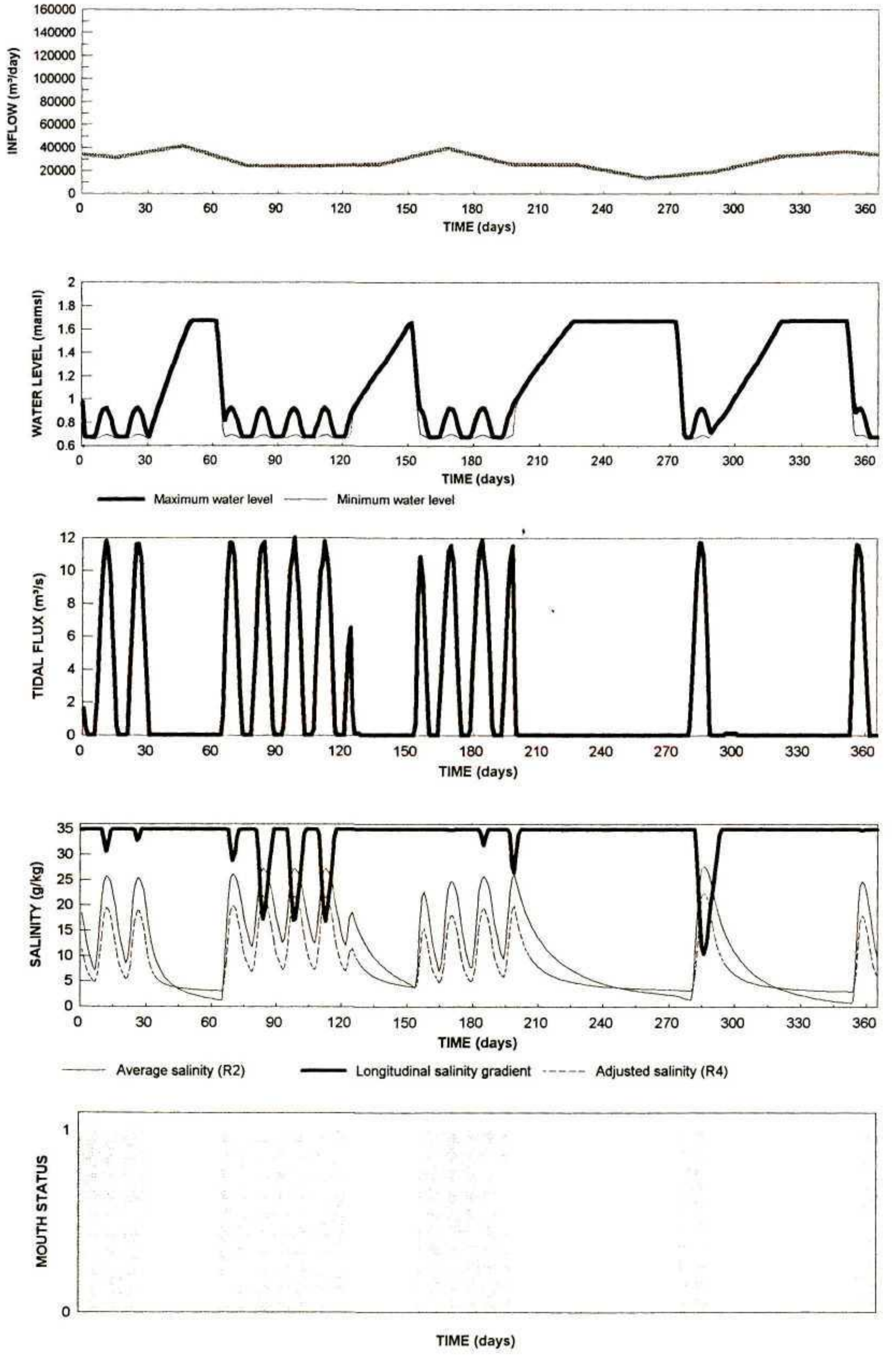


Figure 6.6 : Flux in key characteristics of the estuary environment in relation to freshwater inflow anticipated under post dam conditions, with high baseflow (Scenario 3a) (Slinger 1995)

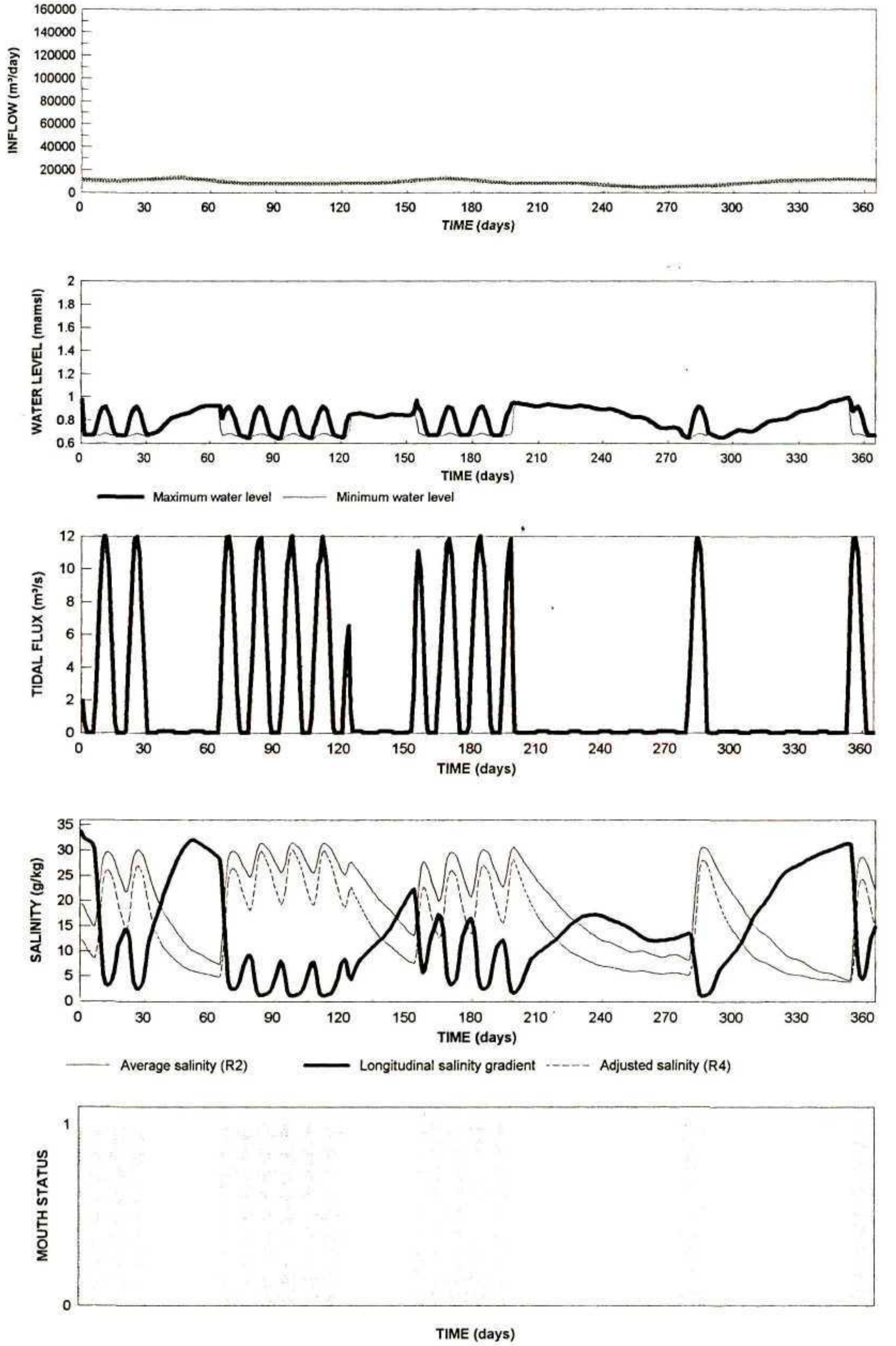


Figure 6.7 : Flux in key characteristics of the estuary environment in relation to freshwater inflow anticipated under post-dam conditions with low baseflow (Scenario 3b)(Slinger 1995)

6.3.5 Summary of scenarios

Key characteristics of the scenarios are summarised below in relation to the freshwater inflows which gave rise to them (Table 6.1).

Table 6.1 : Freshwater release volumes and characteristics of the resultant conditions in the estuary (Slinger 1995)

		AVERAGE CONDITIONS				50% LESS BASEFLOW IN WINTER				1 : 50 YEAR FLOOD			
		1	2	3a	3b	1	2	3a	3b	1	2	3a	3b
FRESHWATER INFLOW ($x10^6$ $m^3.yr^{-1}$)	min	16.7	11.8	4.9	1.55	15.5	10.94	4.56	1.4	16.72	11.8	4.9	1.55
	max	51.2	36.1	15.06	4.75	51.2	36.15	15.06	4.7	721.2	695.1	968.9	962.3
	mean	34.7	24.5	10.2	3.22	30.08	21.24	8.85	2.79	35.64	25.4	11.52	4.53
SALINITY ($g.kg^{-1}$)	min	2.89	2.95	2.96	3.90	2.89	2.98	3.0	3.91	2.86	2.87	2.95	3.11
	max	15.84	18.29	22.69	30.21	17.45	8.13	24.92	32.86	15.84	18.29	22.7	30.21
	mean	6.24	7.18	7.95	14.17	7.09	19.97	9.17	16.37	6.64	7.52	8.09	14.14
AXIAL SALINITY DIFFERENCE	min	34.97	34.41	9.57	1.04	34.97	28.08	4.84	0.38	34.97	34.41	9.57	1.04
	max	35	35	35	33.61	35	35	35	33.07	35	35	35	35
	mean	35	34.99	33.69	14.08	35	34.88	31.29	11.29	35	34.99	33.75	15.16
MOUTH STATUS (Days)	closed	20	54	213	213	20	47	211	211	20	54	211	211
	open	345	311	152	152	345	318	154	154	345	311	154	154
	% open	94.52	85.21	41.64	41.64	94.52	87.12	42.19	42.19	94.52	85.21	42.19	42
WATER LEVEL (m MSL)	min	0.67	0.67	0.67	0.60	0.67	0.29	0.29	0.28	0.67	0.67	0.67	0.54
	max	1.90	1.80	1.67	1.00	1.67	2.18	1.92	1.92	1.72	1.80	167	0.99
	mean	0.79	0.85	1.11	0.77	0.76	0.81	1.10	0.76	0.78	0.83	1.05	0.74

6.4 MODEL TESTING AND VALIDATION

At least in the early stages of model development, the value of models lies more in assisting in the process of information and understanding consolidation than it does in their predictive capability. As expressed by Lee (1993) "*predictions of [ecosystem] behavior are ... incomplete and often incorrect. These facts do not decrease the value of models, but they do make it clear that ecosystem models are not at all like engineering models of bridges or oil refineries. Models of natural systems are rarely that precise or reliable. Their usefulness comes from their ability to pursue the assumptions made by humans - assumptions with qualitative implications that human perception cannot always detect.*" Once sufficient data to test their validity and refine their formulation are available, greater confidence can be placed in their predictive ability. Data to test the approaches described in this thesis have yet to be obtained, and consequently it has not been possible to rigorously test any of the models.

Nevertheless, where possible, preliminary testing and evaluation have been undertaken. In the case of the fish recruitment index, several of the assumptions discussed in Chapter 3 are considered below in Section 6.4.1. In the case of the *Upogebia africana* model, the modelling approach is tested in Section 6.4.2 against the limited data available, with certain parameters being subject to a sensitivity analysis in Section 6.4.3.

6.4.1 Evaluating the assumptions of the fish recruitment index

As indicated in Chapter 3, the two principal components of the fish recruitment index are the dependency score and the optimal recruitment score. The former represents the extent to which a species is dependent on estuaries, with different categories of dependence being allocated various scores, based primarily on a value judgement related to the perceived importance of estuaries for the species under consideration. Several possibilities for score allocation were identified in Chapter 3, and these are repeated below in Table 6.2 for reference.

Table 6.2 : Dependency scores (DS_i) allocated to categories of fish included in the model according to their dependence on estuaries.

CATEGORY	DEPENDENCY SCORE (DS_i)		
	Assumption 1	Assumption 2	Assumption 3
I	not included in model	not included in model	not included in model
IIa	5	5	5
IIb	3	3	2
IIc	2	1	1
III	1	1	1
IV	not included in model	not included in model	not included in model
V	5	5	5
endemic	5	5	5

The second of the two principal model components, the optimal recruitment score, is based on data spanning several years, compiled from a variety of sources (Wallace & van der Elst 1975; Melville-Smith & Baird 1980; Bruton *et al.* 1987; Whitfield 1990; Whitfield & Kok 1992). As there is a greater level of certainty regarding the preferred recruitment periods for fish, and furthermore as data to confirm these assumptions are continually being obtained, this component was not subject to further testing.

To evaluate the effect of various assumptions relating to the dependency score, the dependency score and the optimal recruitment score were combined to form what is referred to as an intermediate score. Thus the intermediate score reflects only the interaction of the two principal components and not the effects of the salinity gradient, mouth condition and flow multipliers. Chapter 3 also referred to the possibility of either an additive or a multiplicative formulation of the index. Thus Figures 6.8 and 6.9 overleaf show, for species occurring in the Great Brak estuary, the intermediate scores obtained by summing and multiplying the dependency score and the optimal recruitment scores respectively, as two different formulations of the model.

The intermediate score in the case of the additive formulation is obviously lower than the corresponding score in the multiplicative formulation as a higher value is obtained by multiplying the maxima (5) of the two component scores than by summing them. In general, there appears to be little difference between the assumptions in either the additive or the multiplicative formulations, and the scores show the same relative trend.

In order to evaluate the effect of increasing the score of endemic species, a fourth assumption was made, whereby species were scored as shown in Assumption 3 of Table 6.2, except that the score of endemic species was not increased to 5. Thus Assumption 4 represents the most conservative assumption in that it is weighted towards entirely dependent species and does not include a weighting for endemic species. As shown in Figures 6.8 and 6.9 in the case of both of the alternative formulations, the scores obtained using this assumption are considerably lower.

An interesting difference between the multiplicative and the additive formulations is evident particularly in the winter months, typified by lower intermediate scores. In the case of the multiplicative formulation scores for the four assumptions converge during the winter months whereas scores in the additive formulation do not. As fewer fish are recruiting in winter one would expect the nett effect of different assumptions (possibly viewed as the potential 'error' in assumption) to diminish, as shown in the multiplicative formulation. For this reason the multiplicative formulation was selected. However it should be noted that as the index provides a relative rather than an absolute measure of recruitment, whether one opts for a multiplicative or additive formulation will not influence policy formulation on the basis of the index.

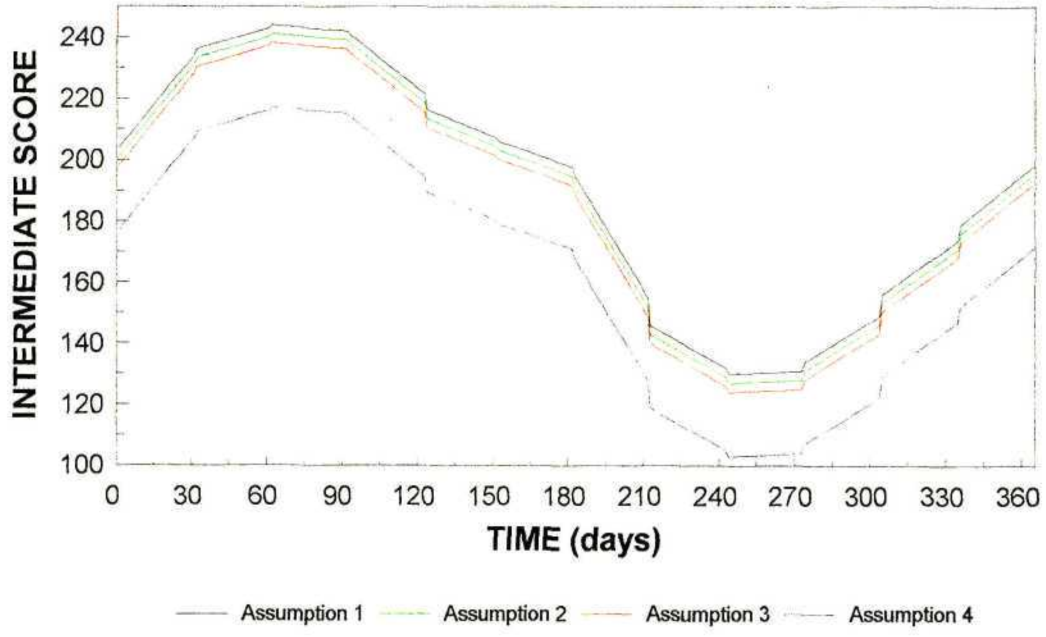


Figure 6.8 : Intermediate scores obtained by adding the dependency score and optimal recruitment score for species occurring at the Great Brak estuary. Different assumptions reflect different score allocations for classes of dependency.

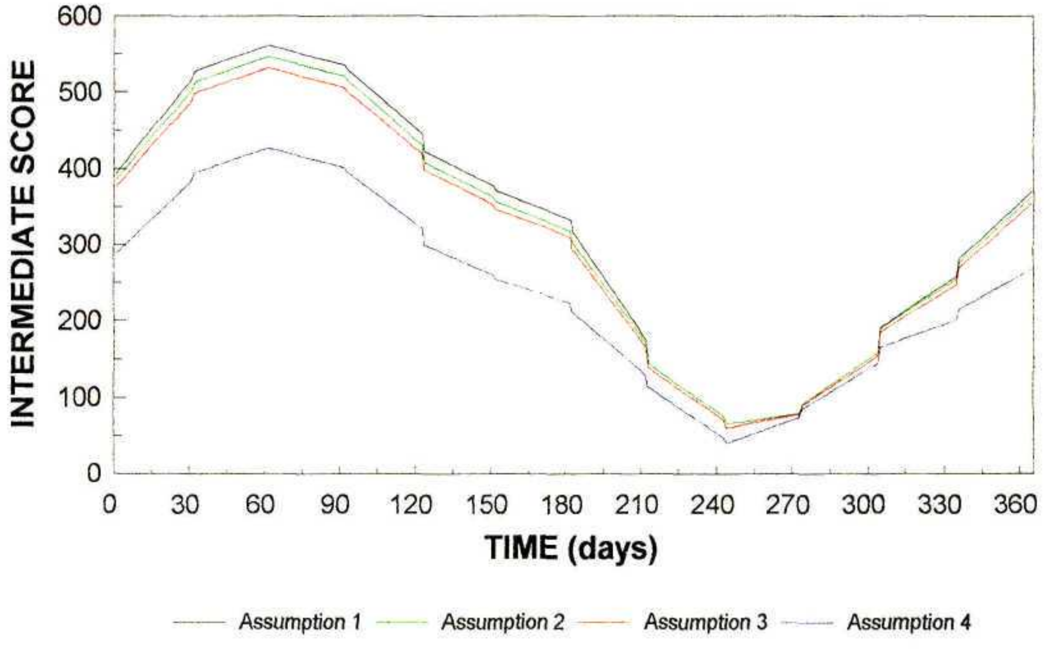


Figure 6.9 : Intermediate scores obtained by multiplying the dependency score and optimal recruitment score for species occurring at the Great Brak estuary. Different assumptions reflect different score allocations for classes of dependency.

A second series of assumptions referred to in Chapter 3 concern the relationship between the prevailing longitudinal salinity difference and the extent of recruitment by juvenile marine fish. Three alternative assumptions were made (Figure 3.2, Page 65), based on the data provided by Whitfield (1994b). Figure 6.10 below illustrates the resultant differences in the longitudinal salinity multiplier under each assumption. Longitudinal salinity difference was assumed to fluctuate between 0 and 35 over the tidal cycle.

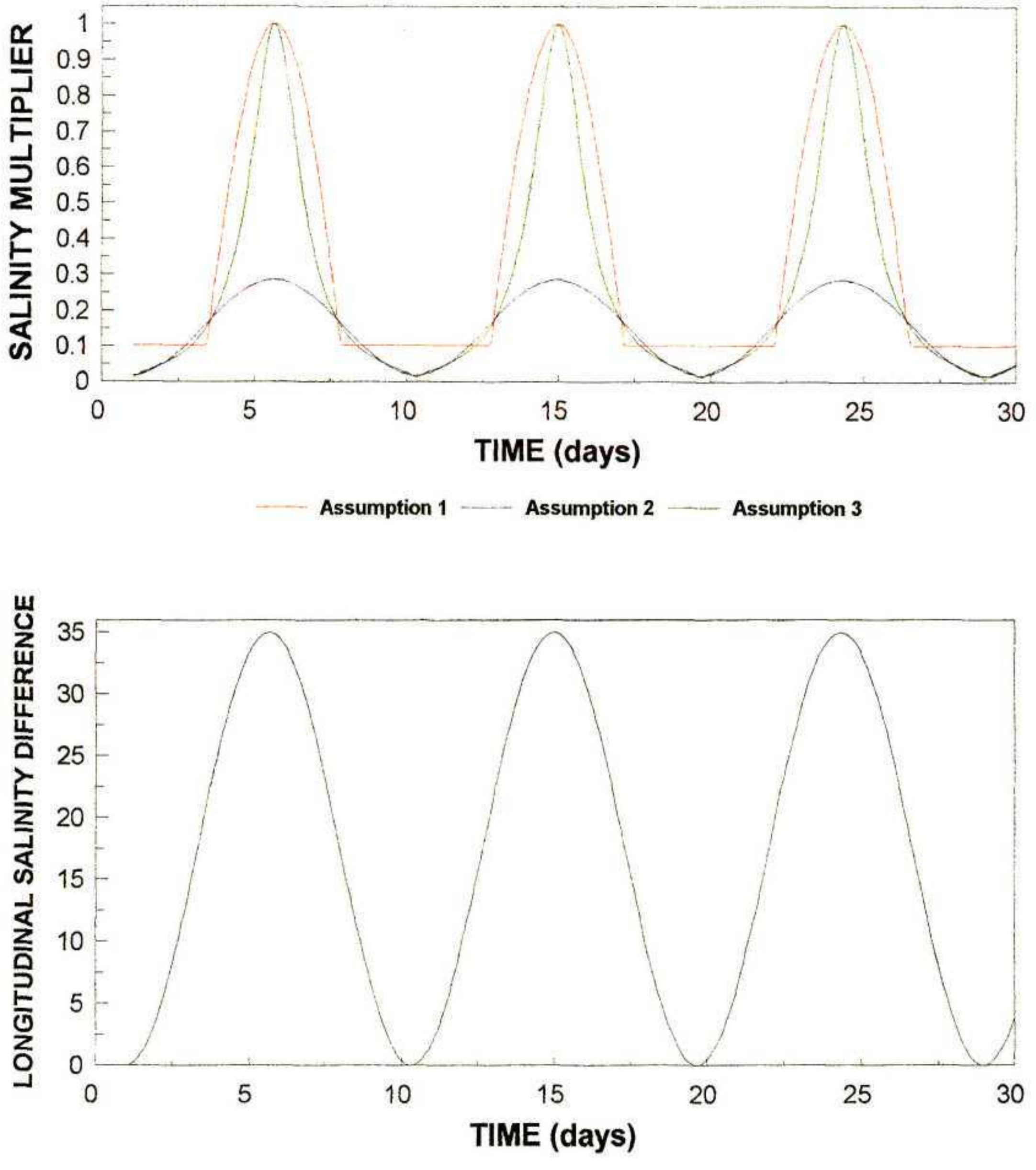


Figure 6.10 : Differences in the resultant longitudinal salinity multiplier over a 30 day period due to alternative assumptions. Longitudinal salinity difference was assumed to fluctuate over the tidal cycle as shown in the lower graph.

As shown in Figure 6.10, Assumption 2 provides the most conservative response, while Assumption 1 returns a greater value for the longitudinal salinity difference multiplier over a greater range of longitudinal salinity difference values. Assumption 3 provides intermediate values, albeit weighted to the higher values of longitudinal salinity difference.

Figure 6.11 below illustrates the effect of these assumptions on the resultant fish recruitment index, as defined by Equation 1 on Page 69, for the freshwater inflow regimes characterising the post dam situation, as shown in Scenario 3b (Figure 6.7, Page 125).

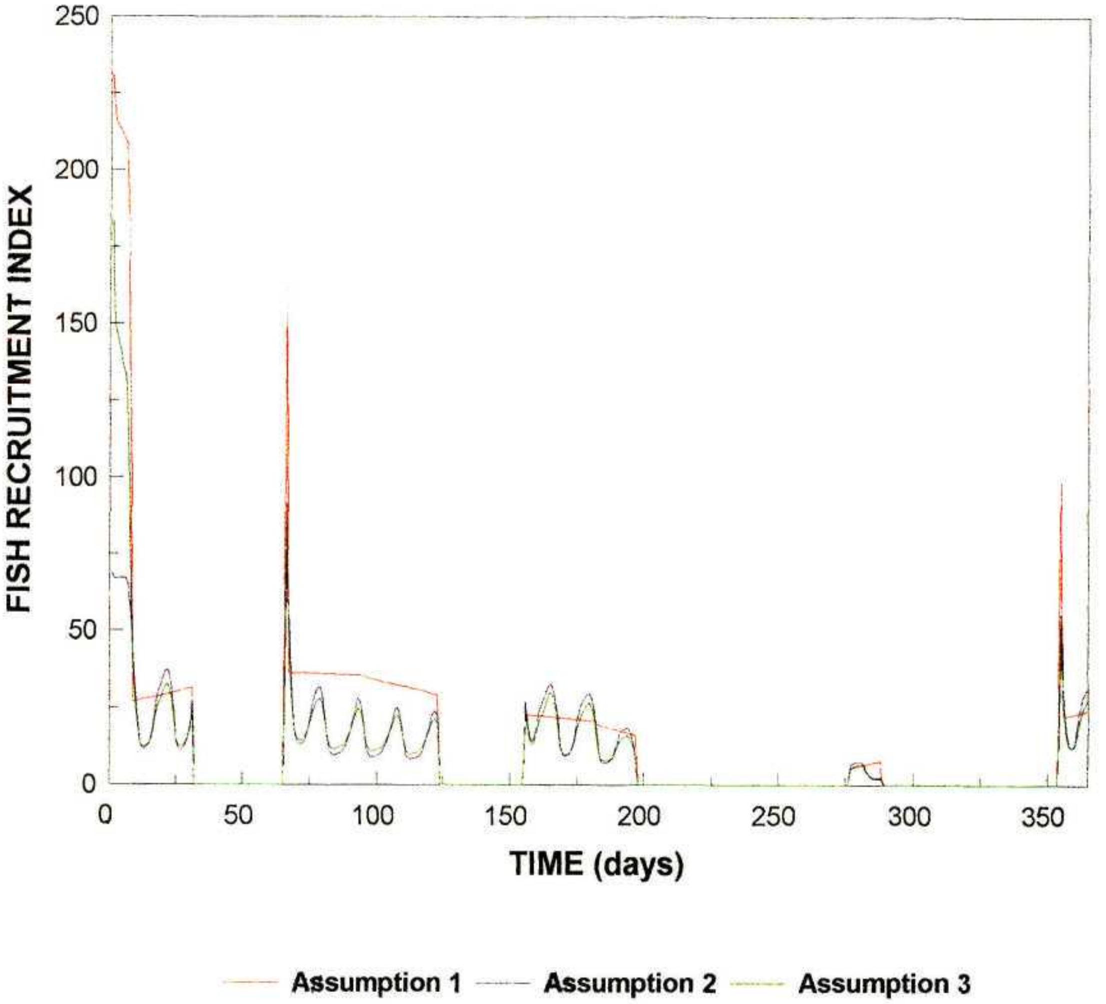


Figure 6.11 : Flux in the fish recruitment index for a given freshwater inflow scenario (Scenario 3b) and for three alternative assumptions regarding the relationship between longitudinal salinity difference and the extent of juvenile fish recruitment.

As can be anticipated from Figure 6.10, the index responds more sharply to an increase in longitudinal salinity difference under Assumptions 1 and 2. For longitudinal salinity differences less than 19 $g.kg^{-1}$, the fish recruitment shows a sinusoidal flux due to tidal effects for Assumptions 2 and 3, but not for Assumption 1. This is due to the fact that, in the case of Assumption 1, recruitment is assumed to occur at a constant, albeit low level below a longitudinal salinity difference of 19 $g.kg^{-1}$. Assumption 1 may reflect the most reasonable assumption, as it is unlikely juvenile fish will respond as closely to the flux in longitudinal salinity gradient as is suggested by Assumptions 2 and 3 in Figure 6.11.

As suggested above, Assumption 2 reflects the most conservative description of the relationship between juvenile fish recruitment and the prevailing longitudinal salinity difference, while Assumption 1 represents the strongest expression of this relationship. Similarly when considering the dependency score, Assumption 1 in Figure 6.9 shows a greater resultant score than Assumption 4, which weights the score to dependent species only and does not account for the presence of endemics. Combining the two most conservative assumptions and the two least conservative assumptions thus results in an envelope of possibility for each freshwater inflow scenario. Figure 6.12, thus shows the maximum and minimum score for the fish recruitment index, under the freshwater inflow regime described by Scenario 3b.

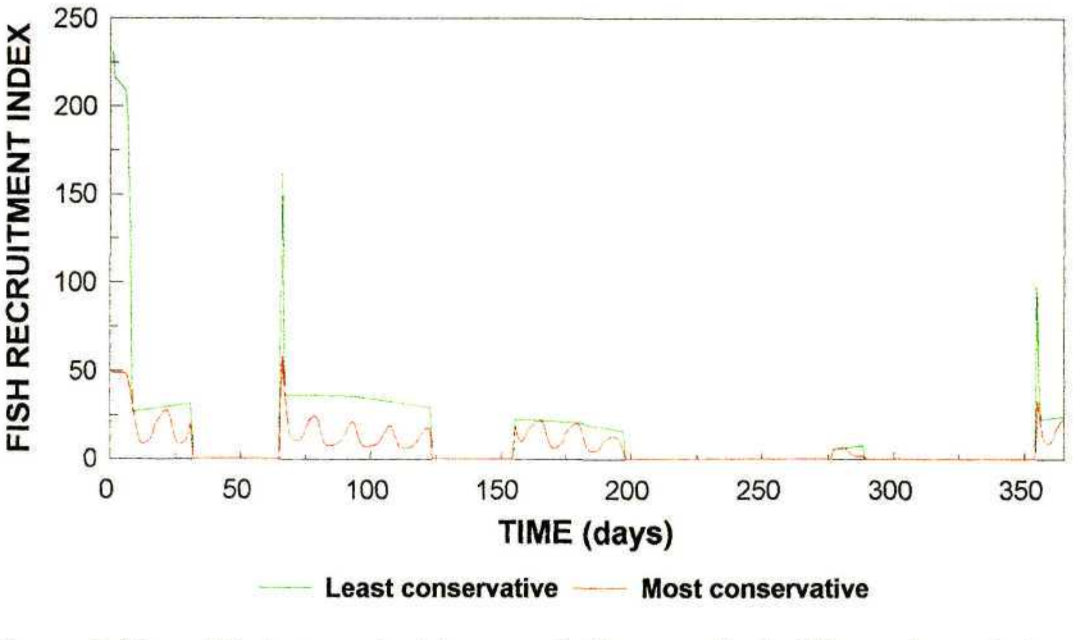
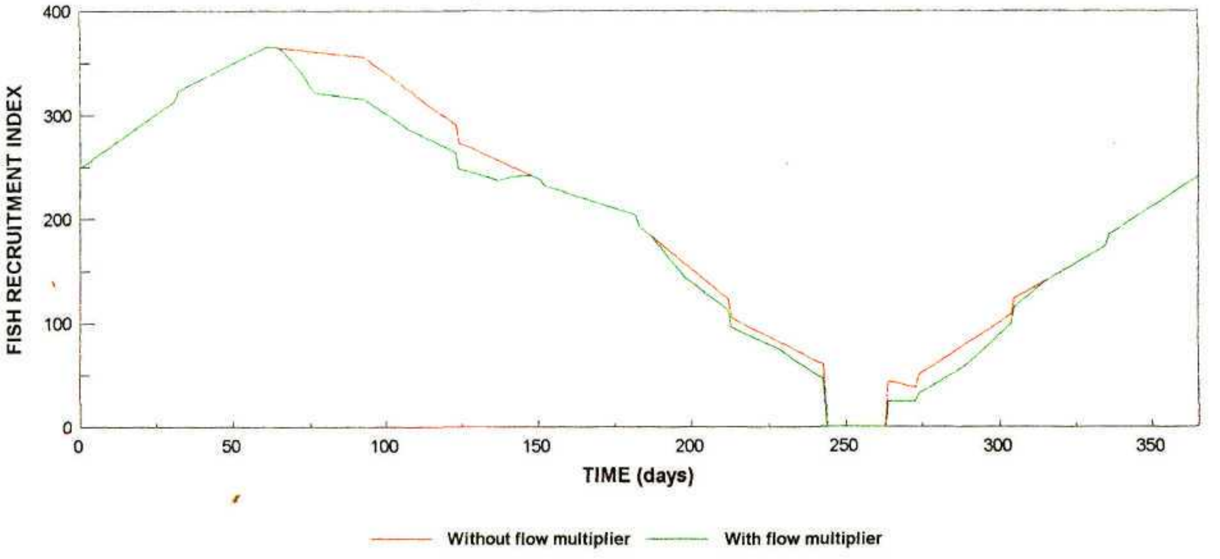


Figure 6.12 : Maximum and minimum probable scores for the fish recruitment index under particular freshwater inflow characteristics (Scenario 3b)

The final component of the fish recruitment index discussed in Chapter3 is the inclusion of a multiplier to account for the deleterious effects of very low or very high flows (Section 3.2.4, Pg 66). The latter occurs due to loss of suitable habitat arising from high flows, whereas the former is as a result of insufficient cues reaching the marine environment to trigger large scale recruitment. To illustrate the effect of including the flow rate multiplier, two scenarios are presented in Figure 6.13 below.



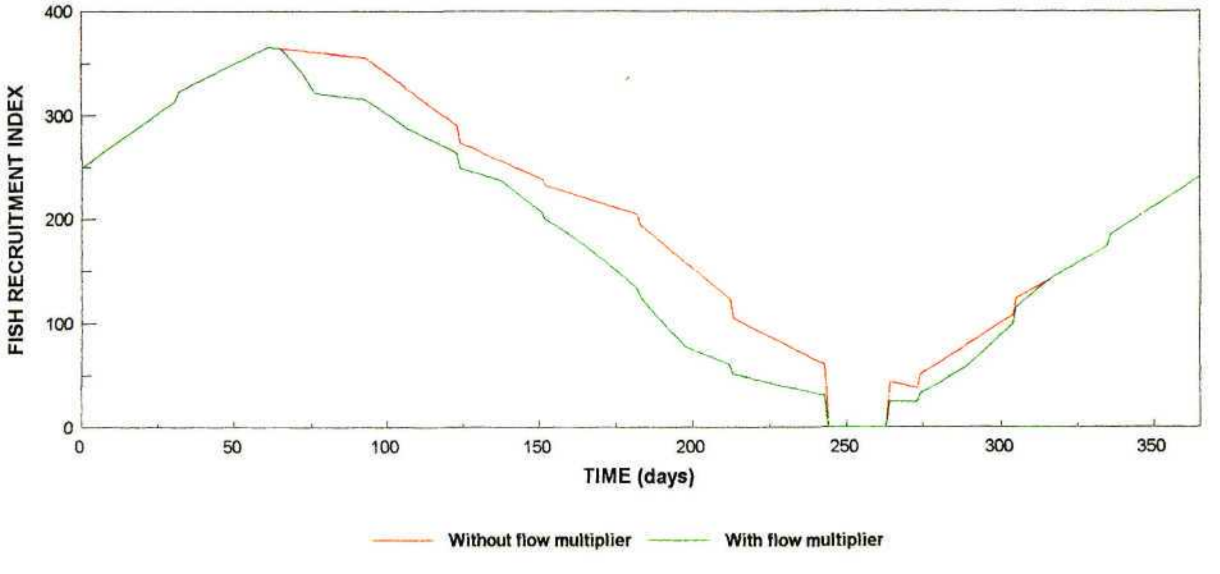


Figure 6.13 : The effect of including a flow rate multiplier on the fish recruitment index. The upper graph corresponds to natural runoff (Scenario 1), whereas the lower graph represents a 50% decrease in flow from March to June.

However, this component of the fish recruitment index is the least developed to a lack of supporting recruitment data, and particularly the flow conditions over which the recruitment was recorded. It is presented here as an opportunity for potential development of the index, subject to further testing and evaluation amongst estuarine biologists.

6.4.2 Preliminary validation of the *Upogebia africana* production model

Wooldridge (1994) surveyed the *Upogebia africana* population in the Great Brak estuary in January and late August 1992, and as the population was limited to a 150m radius in the lower estuary, it was possible to sample the entire area populated by mudprawns on both occasions. The sampling was repeated subsequent to this initial assessment in May and August 1993, and in March and August of every subsequent year. Figure 6.14 shows a frequency histogram of the size class distribution of mudprawns for the two sampling periods in 1992 (Wooldridge 1994). Also shown is a frequency histogram of size class distribution for the permanently open Swartkops estuary, illustrating the extent to which the population in the Great Brak had become skewed towards the adult size classes due to the fact that juveniles could not recruit into the estuary because of extended periods of mouth closure (Wooldridge 1994).

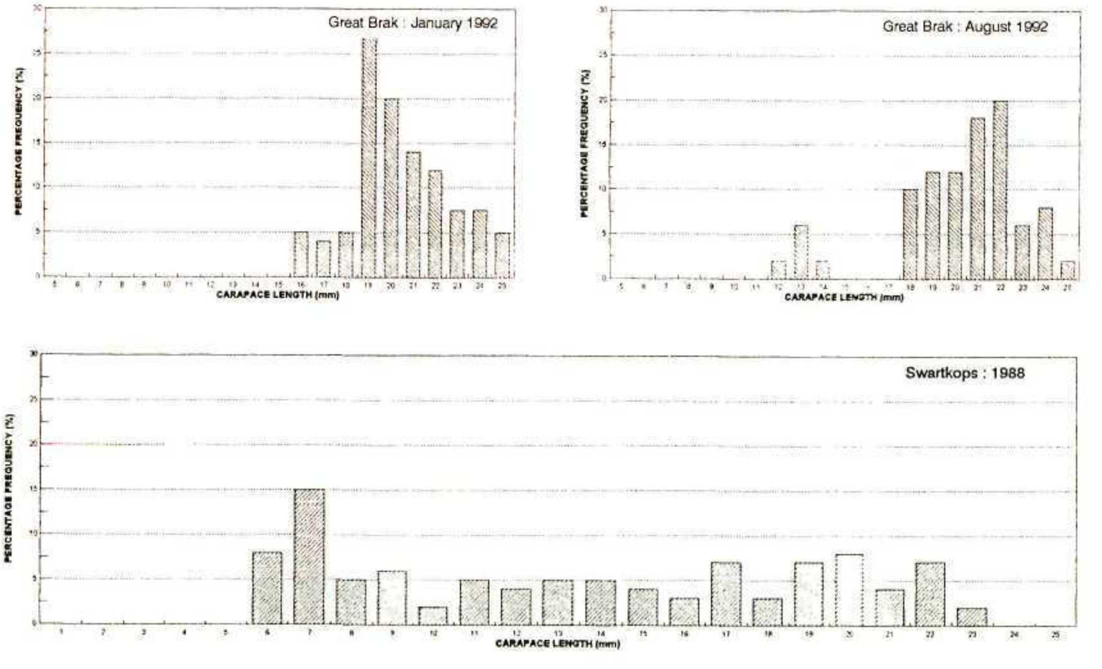


Figure 6.14 :Size class frequency distributions of *Upogebia africana* at the Great Brak estuary for two sampling periods. Also shown is a typical size class distribution for the permanently open Swartkops estuary (*after* Wooldridge 1994).

In order to test the validity of the *Upogebia* model, the model was run to test whether predicted size class distributions were comparable to those found by Wooldridge (1994). Although actual salinity and temperature measurements on a regular basis are not available for the period 1986 to 1992, the condition of the mouth over the period is known. These data, together with temperature determined from the function in Figure 6.3 were used as inputs to the model. Salinity in the estuary was assumed to vary with tidal influence under open mouth conditions (Slinger 1995 *pers. comm*), reducing gradually to approximately 2 to 4 $g.kg^{-1}$ under closed mouth conditions, depending on the length of time the mouth remained closed. Results of the recorded and predicted size class frequency distributions for the first two sampling periods in 1992 as well as the most recent sample in 1996 are shown in Figure 6.15.

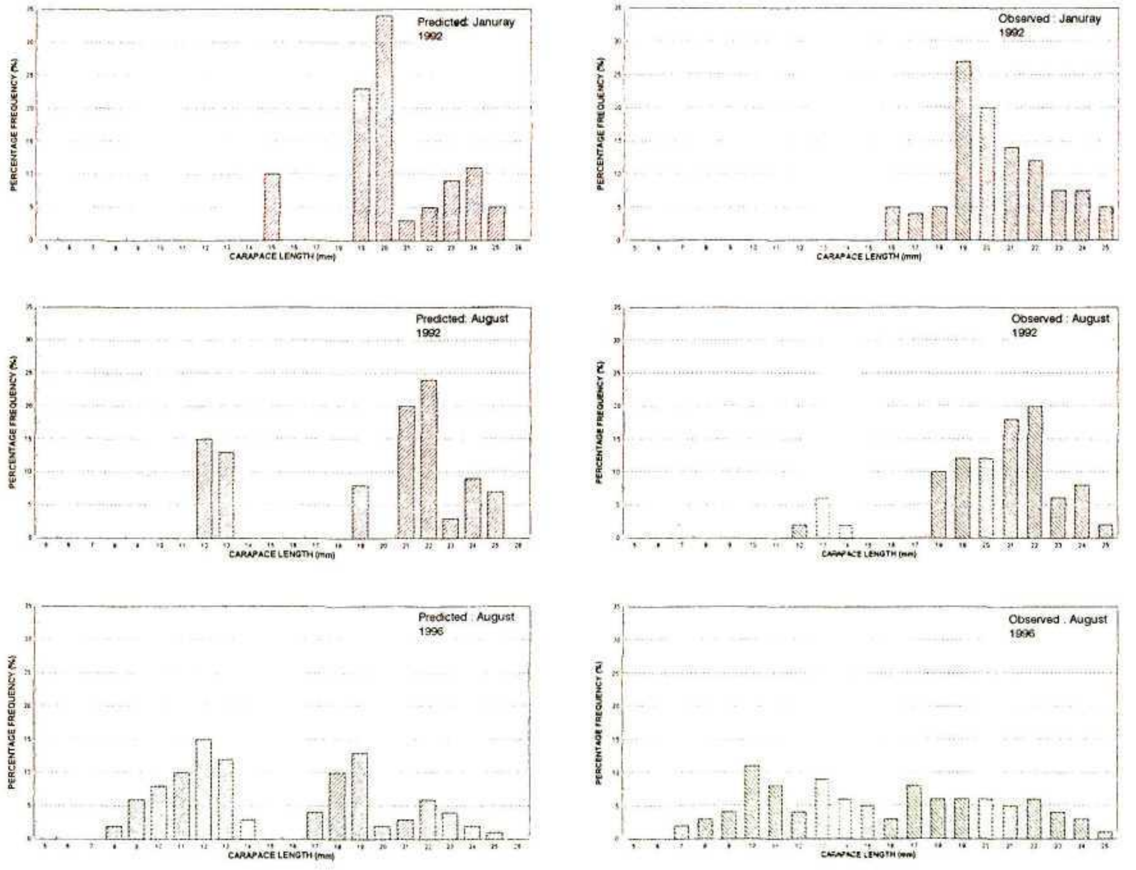


Figure 6.15 : Comparison of observed an predicted Upogebia africana size class frequency distributions for 3 sampling dates, namely January 1994, August 1994 and August 1996 (Wooldridge 1994; Wooldridge 1996, *pers. comm*).

Given that these simulations were undertaken on the basis of assumed salinity and temperature, and with no information regarding patterns of predation or recruitment success, the simulations show a surprising degree of similarity to the findings of Wooldridge (1994). For example during the first two sampling periods, the model results predict the distribution of prawns over a similar range to that of Wooldridge (*pers. comm.*) with comparable peaks in the size class distribution. More significantly the model predicts the appearance of cohorts in the 12 to 13 mm size class in August 1992 which would have recruited due to the open mouth conditions during the previous summer. In the final sampling period the predicted size class distributions occur over a much wider range of sizes, similar to that of the observed data, and closer to that expected from a system with greater opportunity for recruitment, such as that shown for the permanently open Swartkops (Figure 6.14).

The most obvious discrepancy between the observed and predicted size class frequency distributions is that the latter is generally smoother, with some prawns represented in each of the size classes over the range. In contrast the predicted size class distributions show gaps in the distribution, with relatively higher occurrences of prawns in certain size classes. For example in the first sampling period (January 1992), approximately 14 % of the population was recorded in the category 16 to 18 mm, whereas the model predicts the occurrence of approximately 10 % in the 15 mm size class only. A similar pattern is evident in the last sampling period where no prawns are predicted in the 15 and 16 mm size classes, whereas they were recorded in these size classes in the field. Also noteworthy is the fact that the model predicts relatively higher occurrences of prawns in the immediately adjacent size classes (Figure 6.15). This phenomenon may be attributable to a primary assumption of the model, namely that the growth of all individuals within a cohort occurs at the same rate. As has been suggested by others (De Angelis & Gross 1992), growth rates within a cohort are likely to vary and consequently the size class frequency distribution is likely to be smoother than that predicted, as a proportion of prawns in a cohort will grow more slowly and will therefore be recorded in a lower size class while another fraction may grow more quickly and would consequently be recorded in a higher size class. The model therefore preserves the size class distribution anticipated on the basis of recruitment opportunity. For example the predicted gap in 15 and 16 mm size class distributions, corresponding to an age of between 12 and 14 months, occurs because the mouth was closed between 4th June and 6th September, except for two short periods of 7 and 4 days. Recruitment during this period would have been unlikely, or at least very limited and therefore it is reasonable to assume that the prawns recorded in these size classes are represented as a result of differing growth rates of cohorts recruiting before and after the period of mouth closure.

In addition to the available information pertaining to size class frequency distributions, estimates of prawn density are also available (CSIR 1990, CSIR 1992, Wooldridge 1996 *pers. comm.*). Although different sampling methods were used, all surveys of prawn density were based on a count of observed prawn burrow holes. These estimates have been converted to individuals per m^2, using the standard method (CSIR 1990). Figure 6.16 shows a composite of these data sources reflecting the recorded decrease in density due to the construction of the Wolwedans dam and subsequent recovery of the population after the implementation of the mouth management strategy.

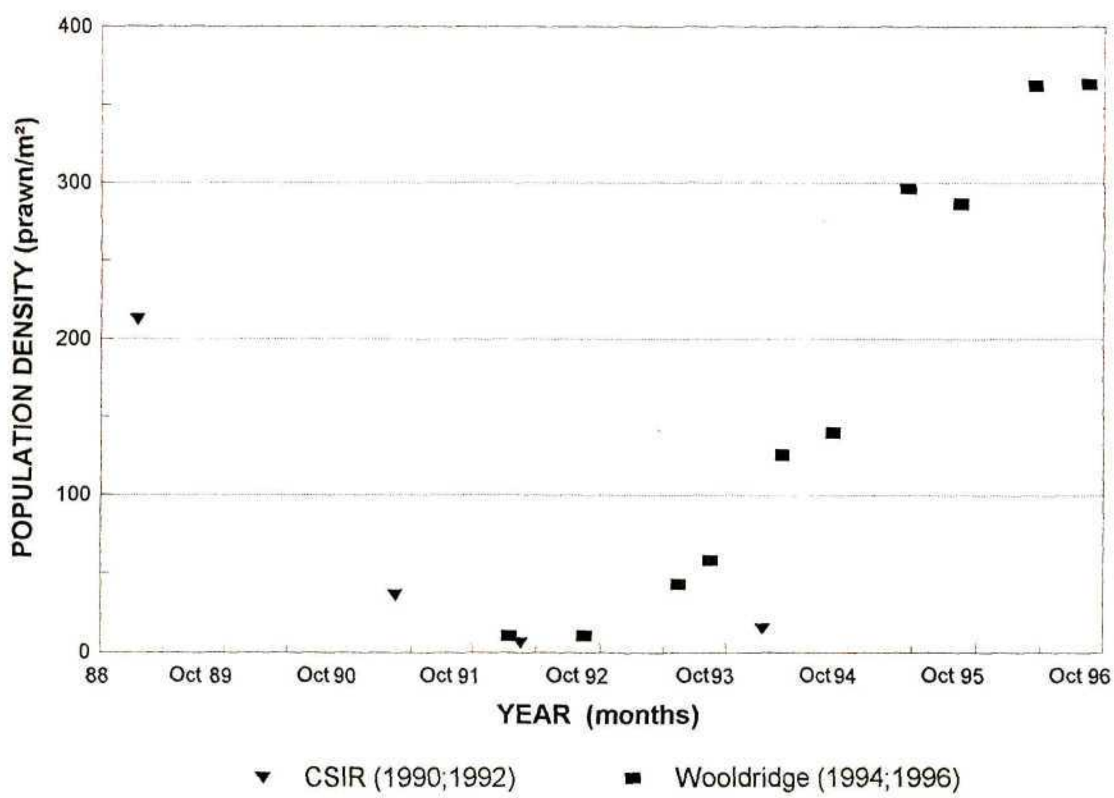


Figure 6.16 : Changes in density of the *Upogebia africana* population (prawns.m^{-2}) over the period 1989 to 1996, as determined from several sources (CSIR 1990; CSIR 1992; Wooldridge 1996, *pers. comm.*).

As is evident from Section 5.5, recruitment of postlarvae was not considered to be dependent on populations within the estuary, and was assumed to occur from an unlimited reservoir in the marine environment. Initial attempts to predict prawn density on the basis of this assumption were unsuccessful, as selection of a fixed recruitment level could not generate the exponential increase in population indicated by the observed data. This suggests that recruitment is strongly related to the current population within the estuary and consequently the model was modified to account for this. Recruitment was thus related to the number of eggs produced by adult female prawns, which is in turn related to the size of these female prawns (Section 5.4). These eggs were assumed to develop into zoea and exported to the marine environment if conditions were correct, and could then return to the estuary between two weeks and a month later if the mouth was open during this period.

When implemented in this manner the model predicted considerably improved estimates relative to the observed prawn densities. Only a small fraction (0.001%) of eggs were required to return as postlarvae in order to sustain the population at observed levels. However, predictions suggested that densities between 1990 and 1993 would have been higher than those which were observed (Figure 6.17).

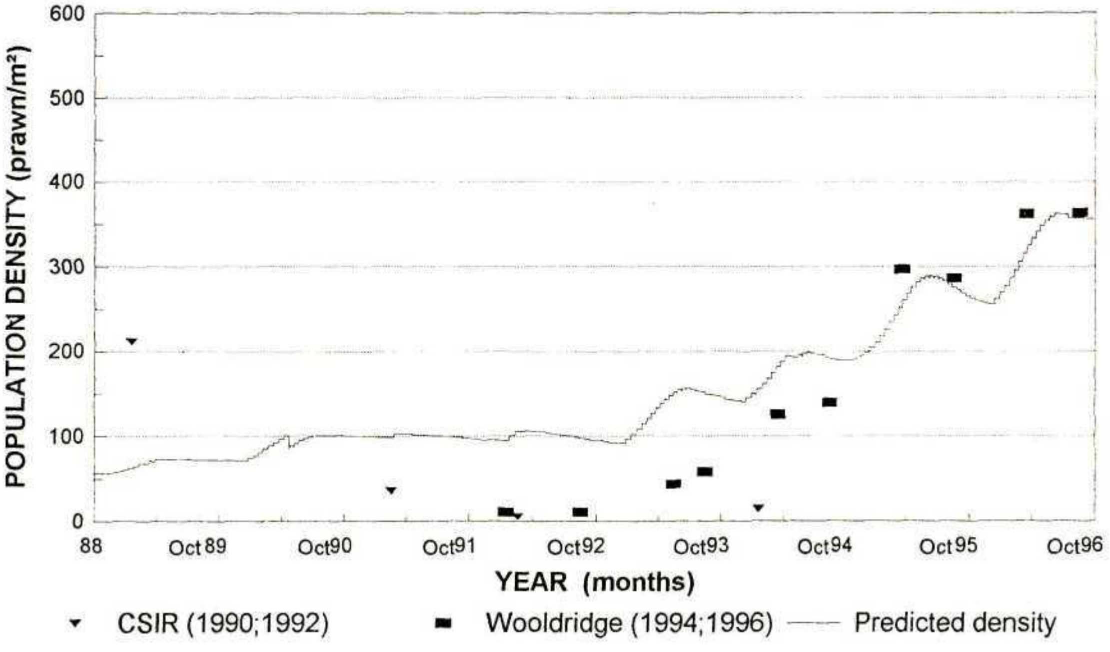


Figure 6.17 : Comparison of observed and predicted prawn density, suggesting densities higher than those actually recorded.

Closer inspection of the size class frequency distributions between August 1992 and May 1993 showed an apparent reduction in the adult population. Whereas in August 1992 the population was almost entirely adult, approximately nine months later the adult population was considerably reduced (Figure 6.18). A reduction of this nature could not occur as a result of natural die-off in the adult population, suggesting that loss of adults occurred by some other means. Inspection of the hydrological record indicated that on 16 October 1992 the dam filled for the first time due to heavy rains in the catchment. As a consequence of dam overtopping the estuary was subject to a flood event approximately equivalent to the 1 in 5 year flood.

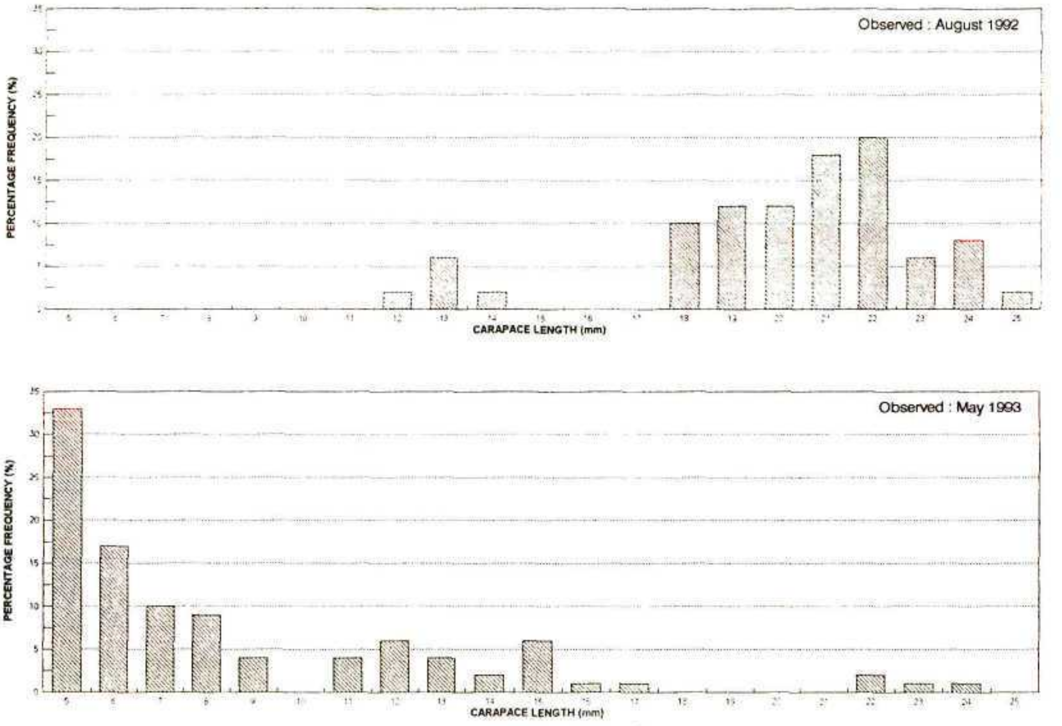


Figure 6.18 : Comparison of observed size class frequency histograms for the periods August 1992 and May 1993 (Wooldridge 1996, *pers. comm*).

Given the effects of a medium flood noted by Hanekom (1989), it is not unreasonable to assume that the reduction in the population between the August 1992 and May 1993 could be attributed to this flood event. Similarly there is no explanation for the decline in the population density recorded between January 1989 and February 1991. Although no observed size class distributions are available for this period to confirm an abrupt decrease in density, a flood was recorded in October 1989. Assuming that these flood events did in fact bring about a decrease in population density, the model was systematically re-run assuming different mortalities due to the floods. Mortalities in the region of 90% for the first flood and 40% for the second flood were found to provide a reasonable match between predicted and observed prawn density for the period under consideration (Figure 6.19).

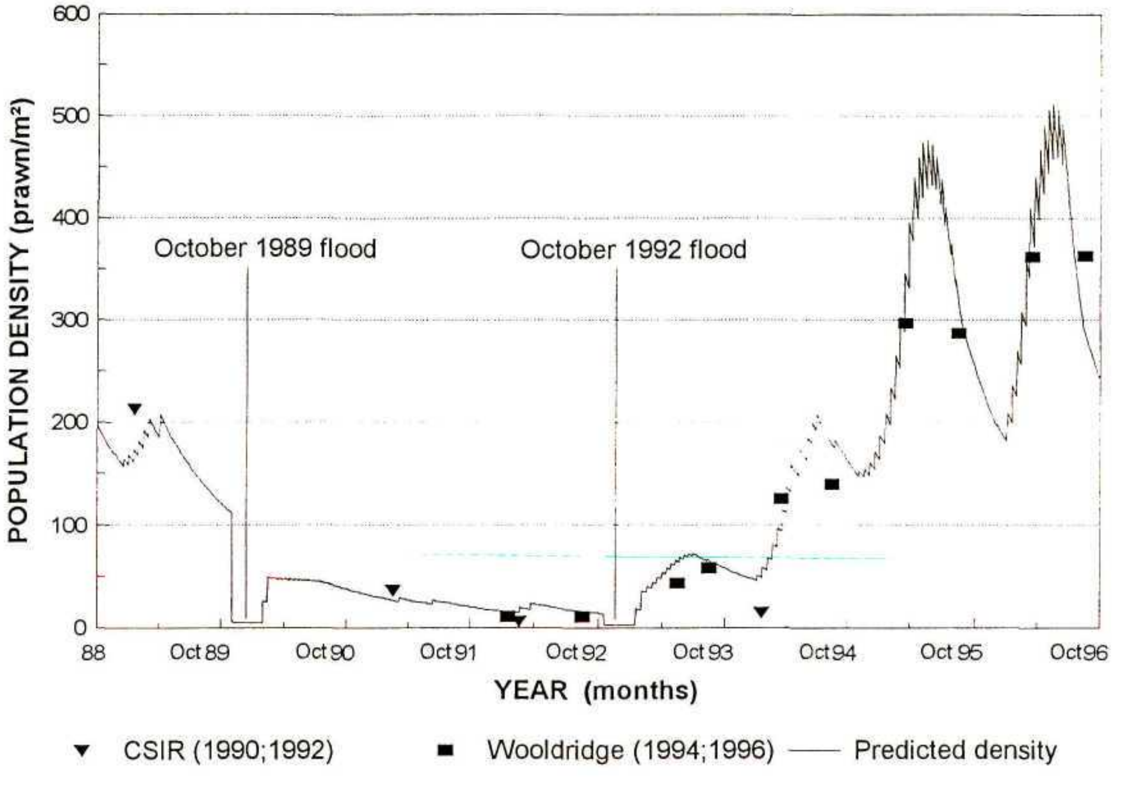


Figure 6.19 : Comparison of predicted and observed prawn densities, assuming that the floods in October 1989 and 1992 reduced the prawn population.

Thus although the *Upogebia africana* model has not been systematically validated, preliminary testing suggests that the model does provide an indication of the response of the population to known changes in the environment, in accordance with expected behaviour and the limited data available. However in order to develop a more comprehensive understanding of the performance of the model, a preliminary sensitivity analysis of the key assumptions and parameters of the model is presented in Section 6.4.3. It is on this basis that parameters were selected for undertaken the scenario simulations in Chapter 7.

6.4.3 Sensitivity analysis of parameters in the *Upogebia africana* model

The *Upogebia africana* production model comprises a number of assumptions, some of which are supported by field and laboratory observation and others which have been determined on the basis of expert opinion. Table 6.3 below provides a summary of the key assumptions and uncertainty in the model, and also provides and indication of the degree of uncertainty associated with each parameter.

Table 6.3: Key assumptions of the model, basis or source of the assumption / parameters and estimate of level uncertainty.

ASSUMPTION / PARAMETER	SOURCE	LEVEL OF UNCERTAINTY
Relationship describing maximum size for a given age (Figure 5.3 Pg. 95)	Expert derived assumption based on three datasets at different locations	LOW
Relationship between temperature and the temperature-age multiplier (Figure 5.7 Pg. 100)	Expert derived assumption based on ecophysiological responses of prawns to a range of temperatures	MODERATE
Relationship between salinity and the salinity-age multiplier (Figure 5.9, Pg. 102)	Expert derived assumption based on ecophysiological responses of different size classes of prawn to a range of salinities	MODERATE
Mortality due to the interaction of unfavourable salinity and temperature (Figure 5.10, Pg. 104)	Limited laboratory data concerning survival times and expert opinion	HIGH
Relationship between carapace size of female prawn and fecundity (Figure 5.11 Pg. 106)	Derived from field data	LOW
Proportion of adult females ovigerous for a given time of year (Figure 5.12 Pg. 108)	Anecdotal evidence, very limited field data	HIGH
Density dependent mortality Figure 5.13 Pg. 110)	Limited data concerning maximum density	VERY HIGH
Fractional mortalities of returning postlarvae	Assumption based on initial model validation and very limited field data	VERY HIGH

Each of the key assumptions indicated above was tested in a rudimentary sensitivity analysis by systematically increasing and decreasing the value of the parameter concerned (by 5% and 15 %) and plotting the resultant effect on the three primary indicators. Figures 6.20 to 6.26 show the results of this exercise.

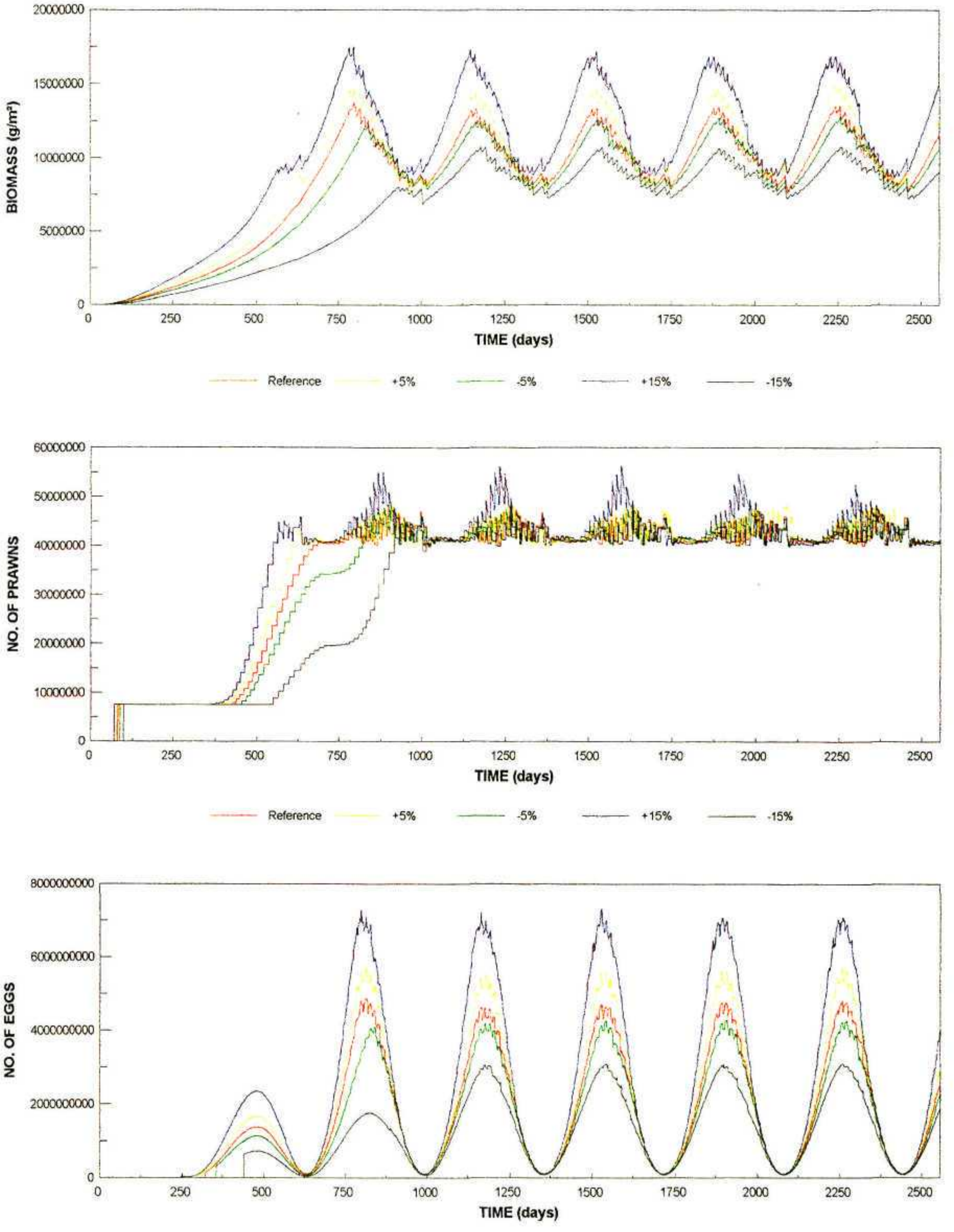


Figure 6.20 : Results of the sensitivity analysis showing the effect of increasing and decreasing the parameter describing maximum size for a given age by a constant percentage over the period of simulation.

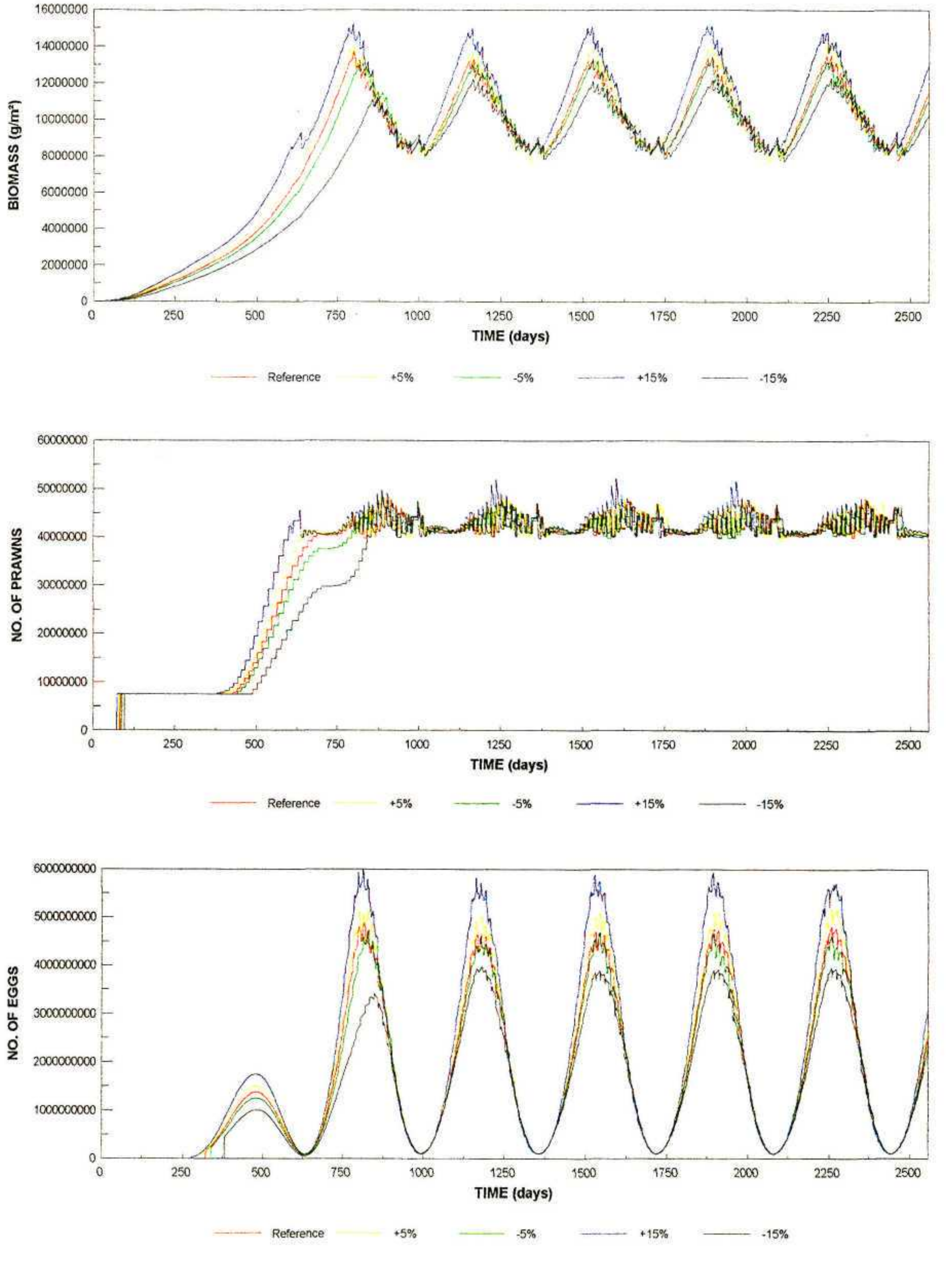


Figure 6.21 : Results of the sensitivity analysis showing the effect of increasing and decreasing the value of the combined temperature and salinity growth multipliers.

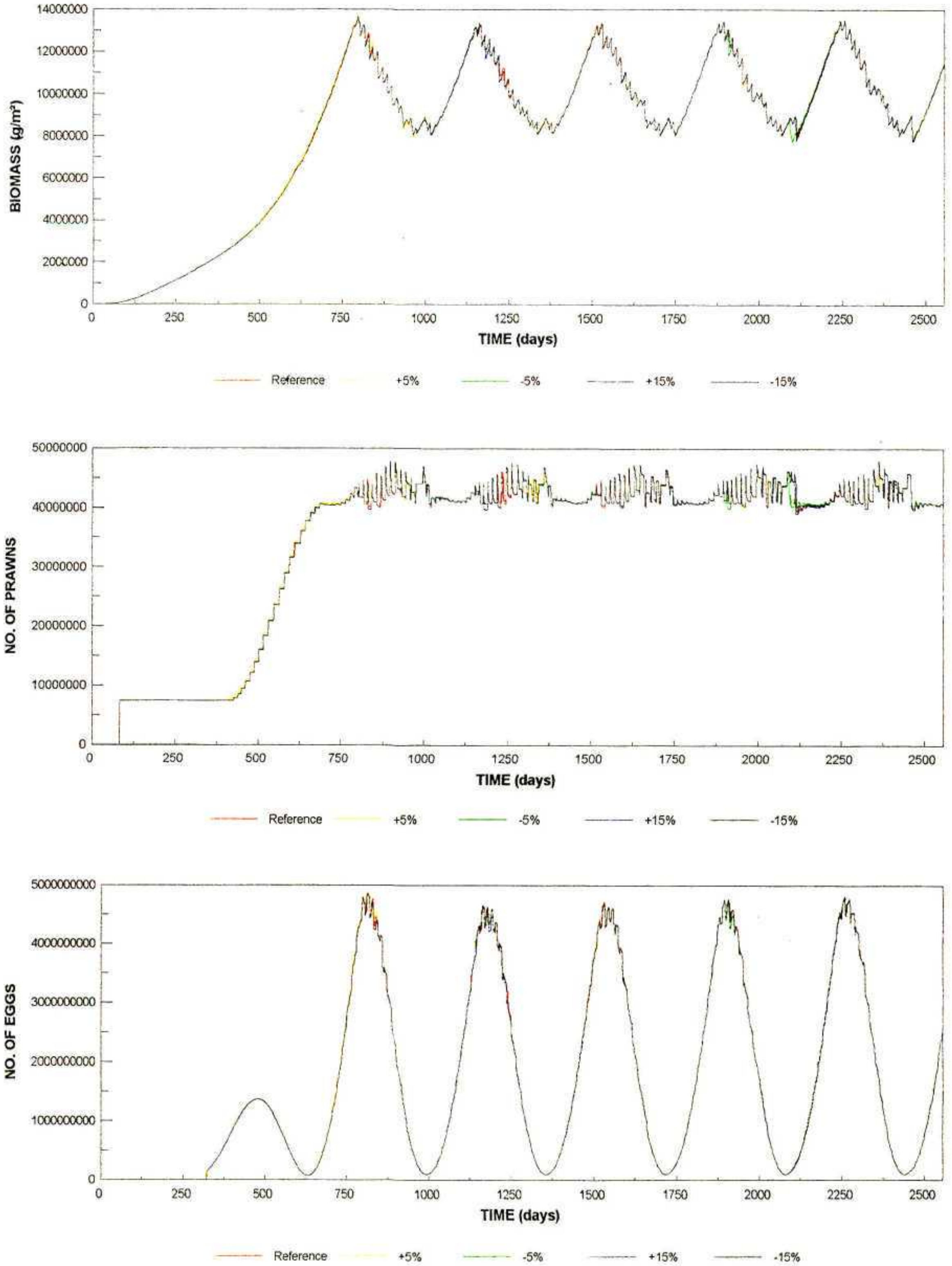


Figure 6.22 : Results of the sensitivity analysis showing the effect of increasing and decreasing the value of the relationship describing mortality due to the interaction of unfavourable temperature and salinity.

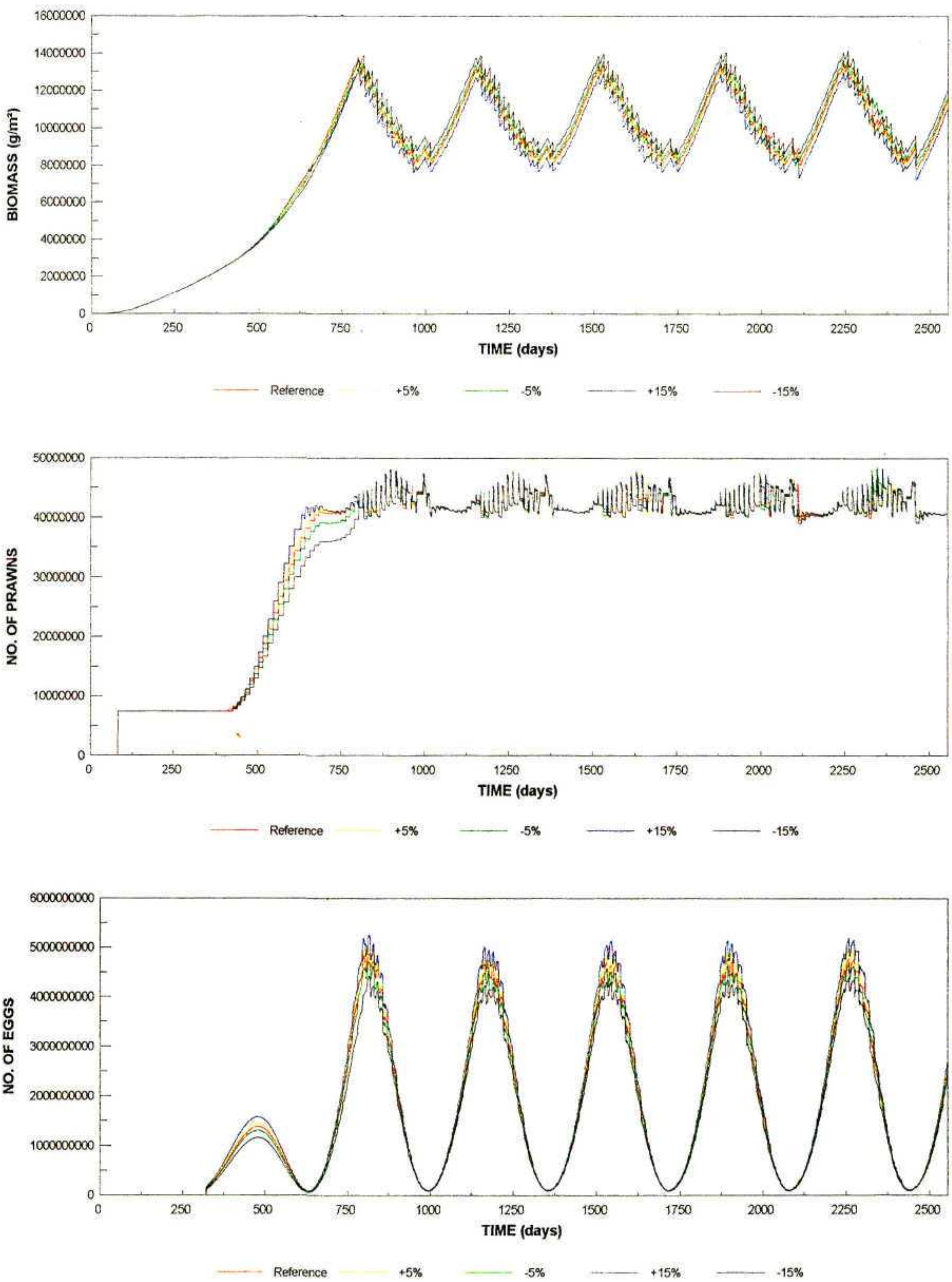


Figure 6.23 : Results of the sensitivity analysis showing the effect of increasing and decreasing the parameter describing fecundity for a given carapace size by a constant percentage over the period of simulation.

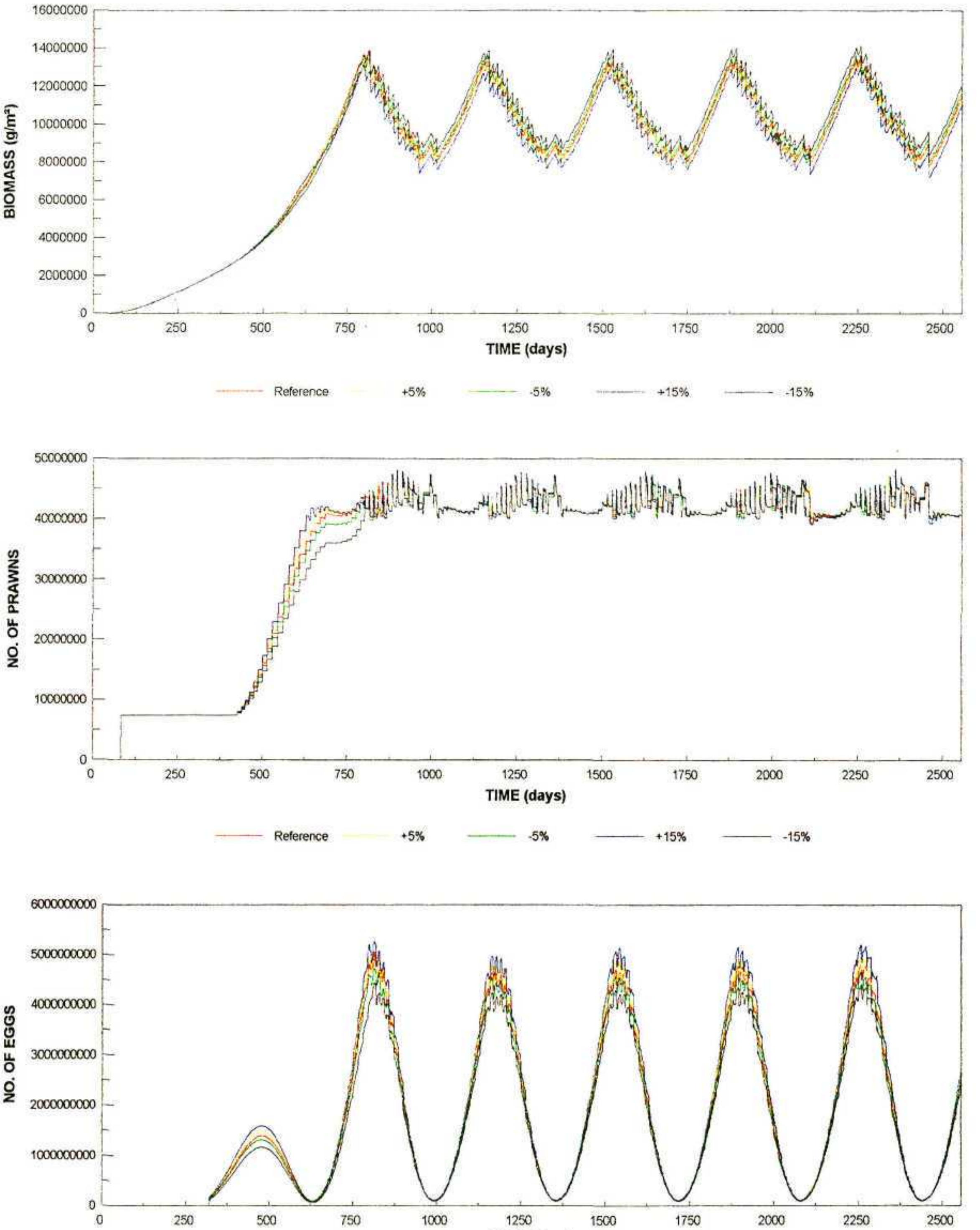


Figure 6.24 : Results of the sensitivity analysis showing the effect of increasing and decreasing the parameter describing the proportion of ovigerous females by a constant percentage over the period of simulation.

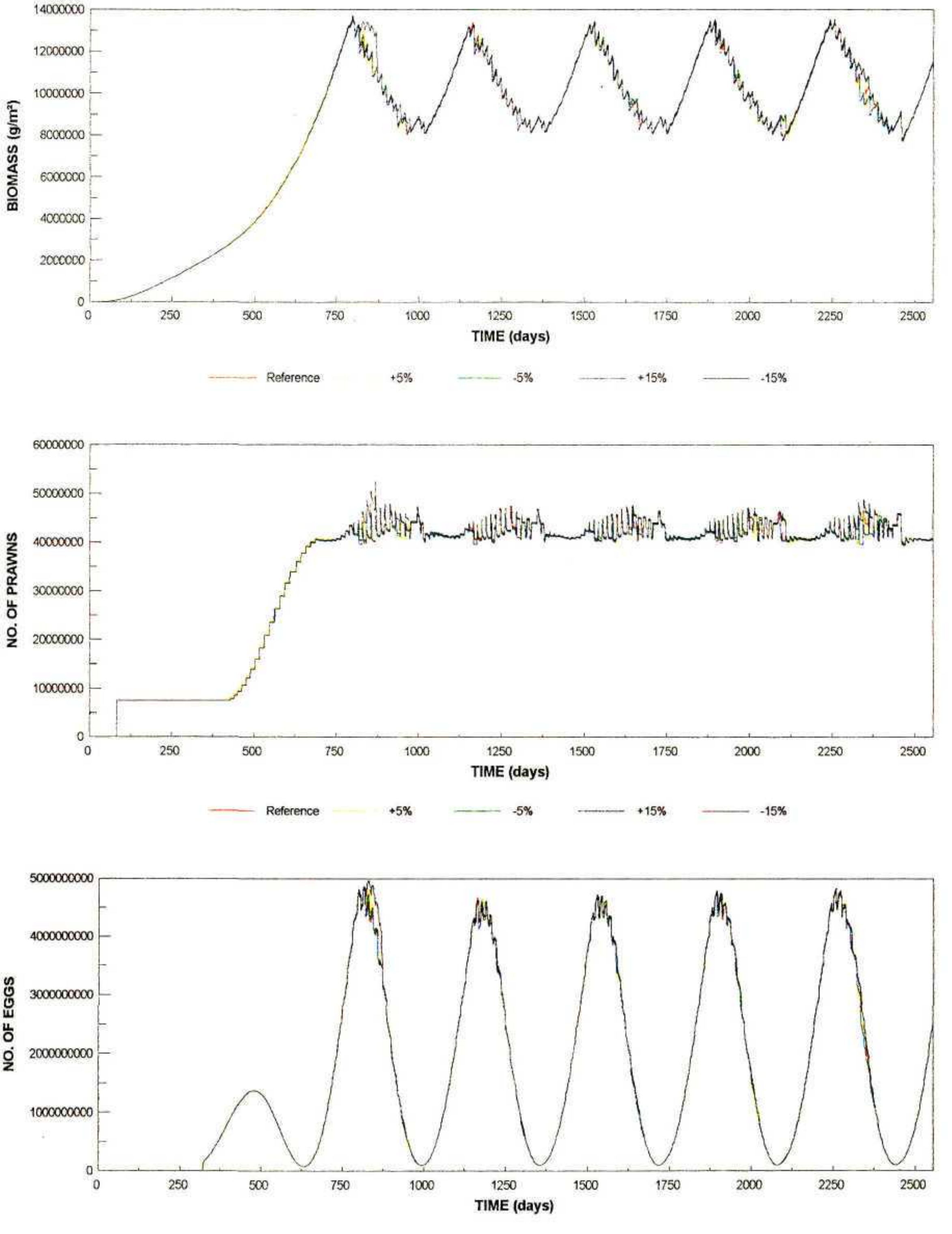
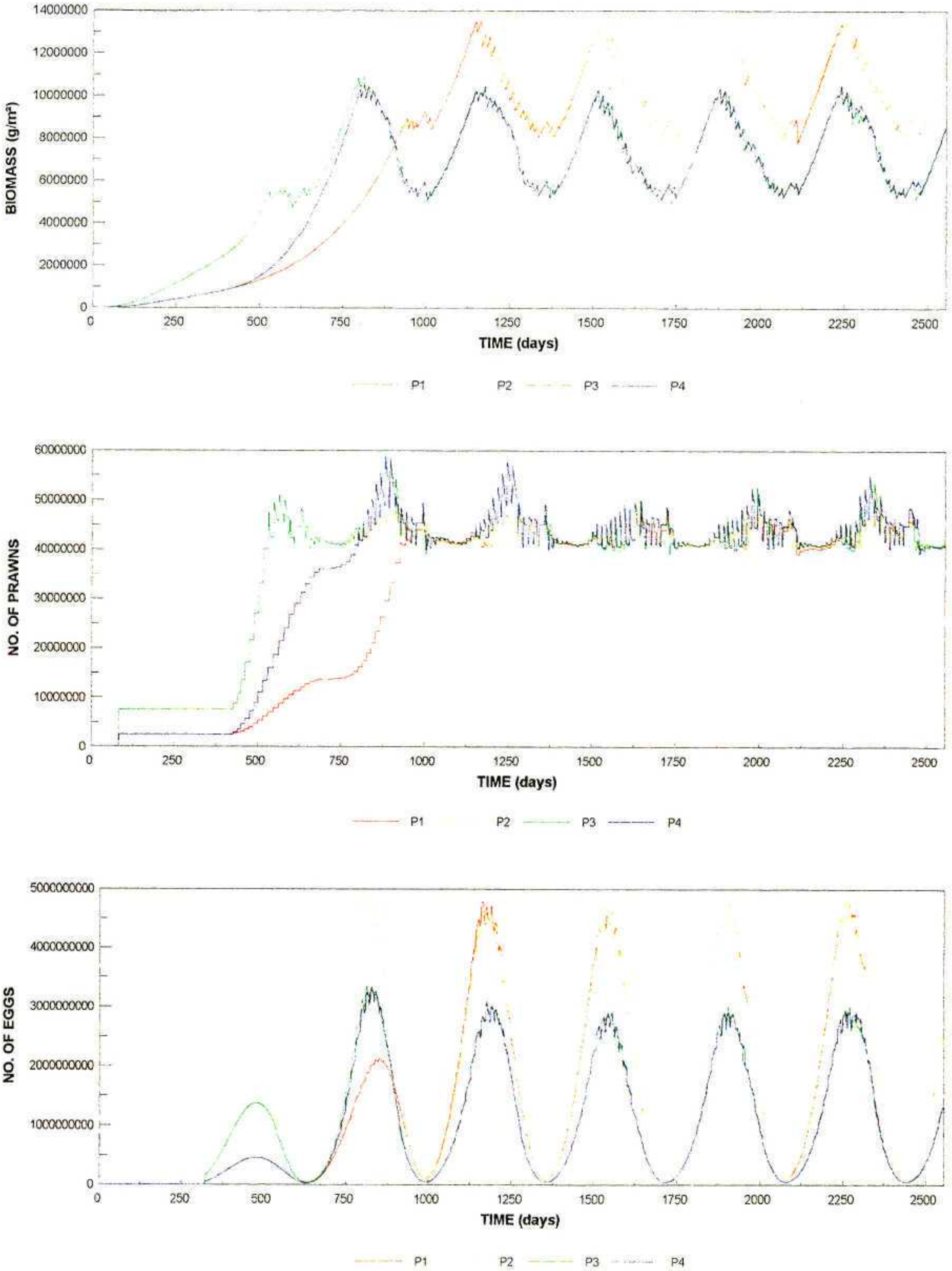


Figure 6.25 : Results of the sensitivity analysis showing the effect of increasing and decreasing the parameter describing density dependent mortality by a constant percentage over the period of simulation.



P1 : iPOP = 2 500 000 FMORT = 0.999

P2 : iPOP = 7 500 000 FMORT = 0.999

P3 : iPOP = 2 500 000 FMORT = 0.997

P4 : iPOP = 7 500 000 FMORT = 0.997

Figure 6.26 : Results of the sensitivity analysis showing the effect of selecting different initial populations (iPOP) and fractional mortalities of returning postlarvae (FMORT).

A comprehensive sensitivity analysis of the effects of errors of assumption or estimation of all parameters is beyond the scope of this thesis. The purpose of undertaking the sensitivity analysis in this chapter is to firstly demonstrate an approach which could be utilised as part of the process of using the model in an estuary management decision making context. The second purpose of this sensitivity analysis is to determine, whether in the case of the scenarios considered herein, inaccuracies of parameter determination would change the ranking of any of the scenarios.

Appendix B provides a summary of the average and maximum values obtained for each of the three principal indices, grouped according to the percentage change in the assumption or parameter under consideration. Table 6.4 below presents this information as a percentage change in the mean or maximum value obtained by each index for a given percentage increase or decrease in the parameter value.

Table 6.4 : Summary of sensitivity analysis expressed as percentage change in indices associated with a given change in parameter value

	+5%	-5%	+15%	-15%
MAXIMUM SIZE-AGE RELATIONSHIP				
BIOMASS	% Change	% Change	% Change	% Change
Mean	6.53	-6.07	22.72	-21.99
Maximum	8.32	-6.67	27.52	-21.46
TOTAL POPULATION				
Mean	1.70	-1.98	6.13	-9.44
Maximum	0.52	-0.59	17.43	-3.84
NUMBER OF EGGS				
Mean	13.25	-11.14	43.31	-36.35
Maximum	17.49	-12.17	50.43	-35.98
COMBINED SALINITY AND TEMPERATURE MULTIPLIERS				
BIOMASS				
Mean	2.34	-1.99	9.07	-8.61
Maximum	3.01	-2.01	11.04	-10.00
TOTAL POPULATION				
Mean	0.63	-0.99	3.39	-4.18
Maximum	-0.37	3.23	8.24	-1.45
NUMBER OF EGGS				
Mean	5.38	-3.65	17.13	-15.42
Maximum	6.55	-3.54	23.38	-18.40

	+5%	-5%	+15%	-15%
SALINITY AND TEMPERATURE MORTALITY MULTIPLIERS				
BIOMASS				
Mean	0.09	-0.02	-0.04	0.09
Maximum	0.16	-0.23	-0.11	-0.17
TOTAL POPULATION				
Mean	0.12	-0.01	-0.01	0.08
Maximum	0.41	0.12	0.01	0.11
NUMBER OF EGGS				
Mean	0.13	0.05	-0.04	0.10
Maximum	0.34	0.02	0.12	0.09
CARAPACE SIZE AND FEMALE FECUNDITY RELATIONSHIP				
BIOMASS				
Mean	-1.44	1.13	-4.25	4.29
Maximum	-1.71	-0.52	-4.23	2.74
TOTAL POPULATION				
Mean	0.34	-0.66	0.69	-1.63
Maximum	-0.41	-0.71	0.38	-1.00
NUMBER OF EGGS				
Mean	2.59	-3.22	7.53	-9.04
Maximum	-1.00	-6.53	4.09	-12.13
PROPORTION OF OVIGEROUS FEMALES				
BIOMASS				
Mean	-1.62	1.23	-4.28	4.18
Maximum	-1.43	-0.32	-3.99	3.04
TOTAL POPULATION				
Mean	0.09	-0.63	0.59	-1.78
Maximum	-0.12	1.16	0.62	-0.74
NUMBER OF EGGS				
Mean	2.50	-2.95	7.56	-9.02
Maximum	2.42	-3.35	7.71	-9.10
DENSITY DEPENDENT MORTALITY MULTIPLIER				
BIOMASS				
Mean	-0.25	0.37	-0.50	1.02
Maximum	0.10	0.16	-0.43	0.47
TOTAL POPULATION				
Mean	-0.22	0.37	-0.50	0.89
Maximum	-0.62	1.53	-0.77	9.59
NUMBER OF EGGS				
Mean	-0.22	0.39	-0.53	1.24
Maximum	-0.14	0.17	-0.77	2.32
INITIAL SETTINGS / CONSTANT				
Initial population (IPOP)		7 500 000	2 500 000	7 500 000
Fractional mortality (FMORT)		0.999	0.997	0.997
BIOMASS				
Mean		13.98	-15.14	-20.41
Maximum		0.93	-19.90	-22.47
TOTAL POPULATION				
Mean		14.62	22.63	14.03
Maximum		-0.05	22.50	22.46
NUMBER OF EGGS				
Mean		14.97	-28.01	-32.64
Maximum		0.86	-30.42	-30.80

As shown in Tables 6.4 and Appendix B, a 5% under- or overestimate of the maximum size for a given age relationship results in a change in biomass of the same order of magnitude. For higher levels of error (15%) the resultant effect on the biomass index is relatively higher. The effect of smaller error on the total population is minor, although in the case of higher error levels (15%) is more evident, resulting in either an increase of 6% or a decrease in population number of 9%. The effect on the number of eggs produced is considerable, increasing the mean number of eggs produced, if the maximum size for a given age relationship is overestimated by 15%, by more than 40%. This is due to the exponential relationship between the carapace size of a female prawn and the number of eggs such an individual can produce. Although this index appears to be sensitive to uncertainty associated with this relationship, it should be noted that the maximum size-age relationship is one of the stronger assumptions of the model (Table 6.3).

All indices appeared to be relatively less sensitive to errors of assumption associated with the temperature-growth and salinity-growth multipliers. This is also the case, and to a greater degree, with the salinity and temperature mortality functions. However this assessment may be misleading in that as the model was run using the natural runoff scenario, occurrences of deleterious salinities and temperatures is infrequent. As the occurrence of these conditions increases, it is likely that the indices will show greater sensitivity to errors in defining these relationships. The implication of this is that a sensitivity analysis should form part of all scenario-driven simulations used for estuary management.

Sensitivity analyses of the relationship between carapace size and female fecundity, as well as the proportion of the female population which is ovigerous at a given time, showed similar responses. In these cases a change of a certain percentage in the function results in a change in the indices of generally less than the value by which the assumed function was changed.

According to Hearne (1998, *pers. comm.*), for a small change in a parameter value the response in the performance indicator (e.g. biomass or total population) is linear (corresponding to the first order terms of a Taylor Series Expansion about the reference values). In this case the effect of a combination of parameter changes can be determined by simply summing the individual responses.

Figure 6.26 shows the influence of a choice of initial population and the fractional mortality of returning postlarvae on the performance of the indices. Higher fractional mortality results in a correspondingly lower biomass. The influence of the choice of initial population level only appears to influence the time the model takes to reach steady state.

The second purpose in undertaking the sensitivity analysis was to determine whether increasing or decreasing the value of a parameter would influence the ranking of alternative scenarios. This was done by selecting the maximum size-age relationship (as an example) and increasing and decreasing the value it returns in a simulation by 15%, as was done to produce Figure 6.20. However in the latter case only one scenario (Scenario 1) was used as input data, whereas in this case the model was run for all four of the principal scenarios. The results of this exercise are presented in Table 6.5 below.

Table 6.5 : Results of a sensitivity analysis whereby a constant error of + 15% and -15% was assumed in the value returned for the function describing the maximum size-age relationship, for all four principal scenarios and three principal indices.

	- 15 %	% CHANGE	REFERENCE	+ 15 %	% CHANGE
			BIOMASS		
Scenario 1	6 444 892	-22.9	8 247 292	10 134 759	22.9
Scenario 2	6 813 666	-22.4	8 778 973	10 814 996	23.2
Scenario 3A	8 038 082	-22.2	10 330 382	14 275 890	38.2
Scenario 3B	7 991 624	-21.8	10 217 946	12 860 376	25.9
			TOTAL POPULATION		
Scenario 1	31 061 497	-9.4	34 266 364	36 404 399	6.2
Scenario 2	31 040 931	-9.6	34 350 453	36 466 957	6.2
Scenario 3A	30 736 549	-7.4	33 186 330	39 338 666	18.5
Scenario 3B	28 980 466	-6.7	31 050 994	33 688 073	8.5
			NUMBER OF EGGS		
Scenario 1	1 030 308 216	-36.2	1 614 395 007	2 316 996 176	43.5
Scenario 2	1 123 712 207	-36.2	1 760 545 243	2 451 577 014	39.3
Scenario 3A	1 444 771 356	-34.8	2 217 330 926	3 371 721 054	52.1
Scenario 3B	1 517 147 274	-35.9	2 365 292 668	3 279 322 406	38.6

Table 6.6 below provides a ranking of each of the scenarios under consideration, for the three different assumptions. Maximisation of the index was assumed to be the objective function in ranking the scenarios.

Table 6.6 : Ranking of scenarios assuming an underestimation and overestimation of 15% in the relationship describing maximum size for a given age.

- 15 %	REFERENCE	+ 15 %
	BIOMASS	
Scenario 3A	Scenario 3A	Scenario 3A
Scenario 3B	Scenario 3B	Scenario 3B
Scenario 2	Scenario 2	Scenario 2
Scenario 1	Scenario 1	Scenario 1
	TOTAL POPULATION	
Scenario 1	Scenario 2	Scenario 3A
Scenario 2	Scenario 1	Scenario 2
Scenario 3A	Scenario 3A	Scenario 1
Scenario 3B	Scenario 3B	Scenario 3B
	NUMBER OF EGGS	
Scenario 3B	Scenario 3B	Scenario 3A
Scenario 3A	Scenario 3A	Scenario 3B
Scenario 2	Scenario 2	Scenario 2
Scenario 1	Scenario 1	Scenario 1

As is evident from Table 6.5, the sensitivity analysis shows that in the case of the total population and number of eggs indices, errors of assumption may affect the ranking of alternative scenarios. When considering the biomass index, a 15% overestimate or underestimate does not influence the relative ranking of the scenarios. Thus in the case of the total population index, while the last position in the ranking is consistently Scenario 3b, the first three positions vary, depending on the assumption of error. However it should be noted that differences between index scores are small, particularly in the case of Scenarios 1 and 2. A similar response is evident in the number of eggs index, except that for all three assumptions the last two places in the ranking are constant.

What has this sensitivity analysis shown concerning use of the *Upogebia africana* production model in the context of estuary management? Firstly it should be stressed that the model needs to be used as a supportive tool within the current decision making process. As discussed in Chapter 2, the most successful model for estuary management which has been tested in South Africa thus far, involves collective decision making by a panel of experts representing various disciplines within the general field of estuary science, in association with the affected community. Use of the model (and indeed the other models) in this context should occur on the basis that all specialists are conversant with all the assumptions of the model. The assumptions should be revisited at the beginning of each worksession to permit each scientist to gauge and estimate confidence in each of the assumptions for the particular estuary being considered. Confidence estimates generated in this manner could be used to define the parameters and the extent to which sensitivity analyses are undertaken within the decision making process.

Secondly the notion of sensitivity needs to be approached from a broad perspective. As suggested in the preceding discussion (Pg 151), the results of a sensitivity analysis are in some cases, dependent on the input scenarios used to derive the results. This once again suggests that a sensitivity analysis should form an integral part of running the model.

In summary, this Section set out to illustrate a methodology whereby the consequences of different assumptions regarding relationships within the model could be tested. In the case of the four principal scenarios, this exercise illustrated that errors of assumption could lead to changes in the ranking of certain scenarios, and therefore by inference the scenario which is chosen for management implementation. However it should be recognised that the primary input data for the model (inflow, salinity, temperature etc.) in the case of these scenarios was schematic, and in the case of temperature and salinity based on certain assumptions. Testing the model using accurate hydrodynamic data may provide a better indication of the performance of the model. Nevertheless, this emphasises the need for sensitivity analysis to form the core of any application of the model.

7 MODELLING RESULTS AND DISCUSSION

7.1 RESULTS

The results of applying each of the three modelling techniques in 12 scenarios for the Great Brak estuary are presented below. Assumptions and limitations of the approaches are discussed further in Section 7.2.

7.1.1 Results of the fish recruitment index

Results for each of the three sets of four scenarios are presented in Figures 7.1 to 7.3, and Figure 7.4 presents a summary of these results, in relation to the annual runoff for the scenario.

(i) Natural runoff scenario

Under the natural runoff scenario the mouth closes for approximately two weeks in June due to low inflow, thereby limiting recruitment. During the remainder of the simulation the mouth is open and the axial salinity difference is higher than 20 $g.kg^{-1}$, and as a consequence juvenile fish recruitment occurs at the potential maximum, resulting in a mean annual score of 206. Furthermore the mouth closure occurs at a time when only 4 of the possible 16 species are recruiting, and these species also recruit outside of the period of closure.

(ii) Pre-dam runoff scenario

A reduction in inflow due to attenuation by minor impoundments as well as abstraction results in a more frequent mouth closure, with the mouth closing in late December as well as in June. However as freshwater inflow is sufficient to maintain the axial salinity gradients, recruitment occurs at maximum potential, except for the periods of mouth closure. Although the mouth closure occurs at the peak of the recruiting period, this constitutes only 12% of the peak recruiting period, and thus the mean annual score remains high at 185.

(iii) Post-dam scenario (maximum expected flow)

Under this scenario the mouth is closed for 50% of the time, with reductions in the axial salinity gradient occurring in response to reduced inflow. Nevertheless the mouth is open for sufficient periods of time to permit recruitment for all 16 species to occur. The mean annual score is reduced to 94.

(iv) Post-dam scenario (average base flow)

Flow is sufficient to maintain an open mouth for a similar period of time, although axial salinity differences are severely reduced due to limited inflow. As a consequence, recruitment never reaches potential maximum and a very low annual score of 15 results.

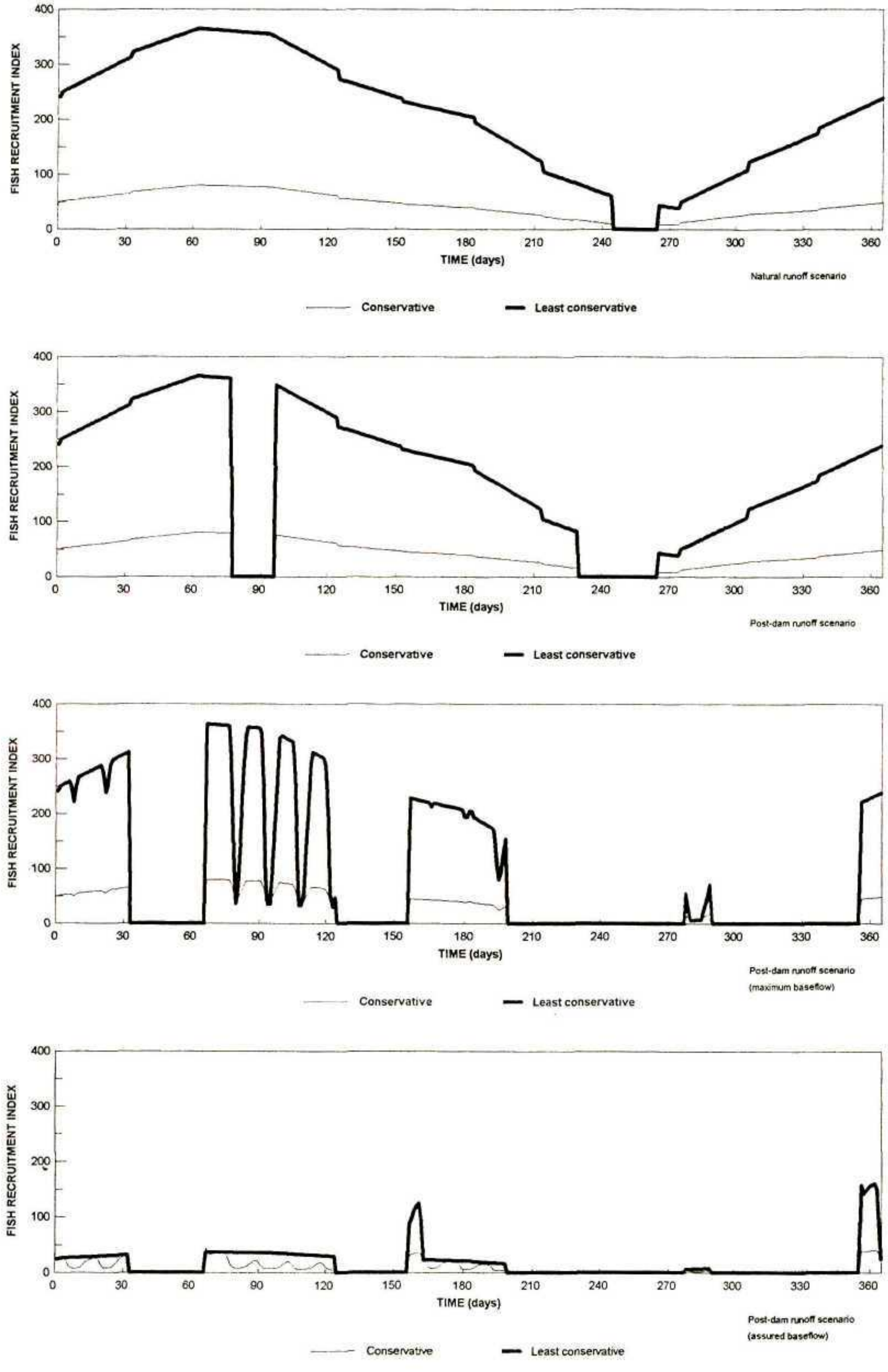


Figure 7.1 : Fish recruitment index for the natural runoff scenario, pre-dam runoff scenario, and the two post dam runoff scenarios under maximum and assured baseflow respectively. Day 1 is 1st October.

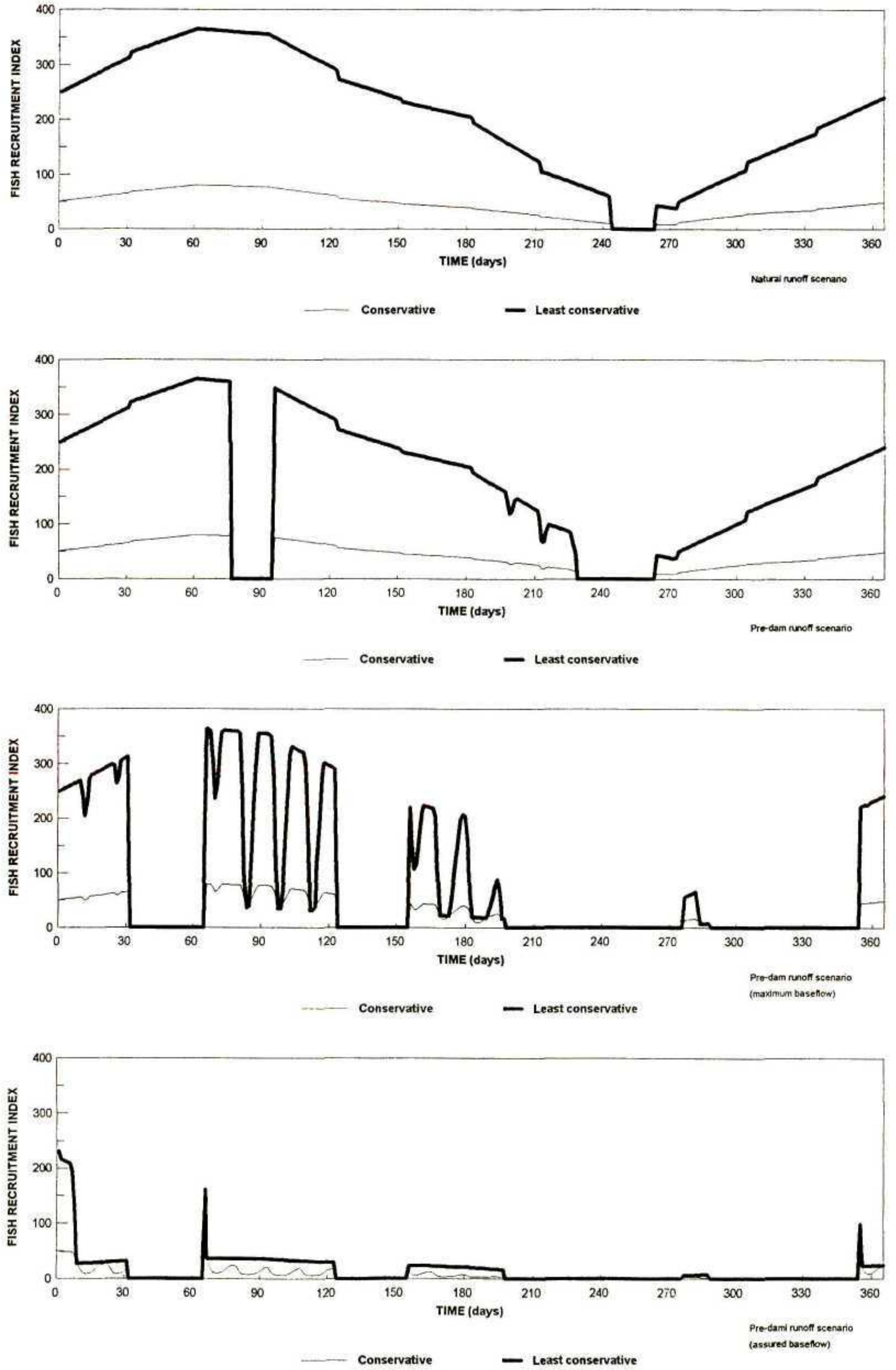


Figure 7.2 : Fish recruitment index for the natural runoff scenario, pre-dam runoff scenario, and the two post dam runoff scenarios under maximum and assured baseflow respectively, assuming a 1: 50 year flood in late summer. Day 1 is 1st October.

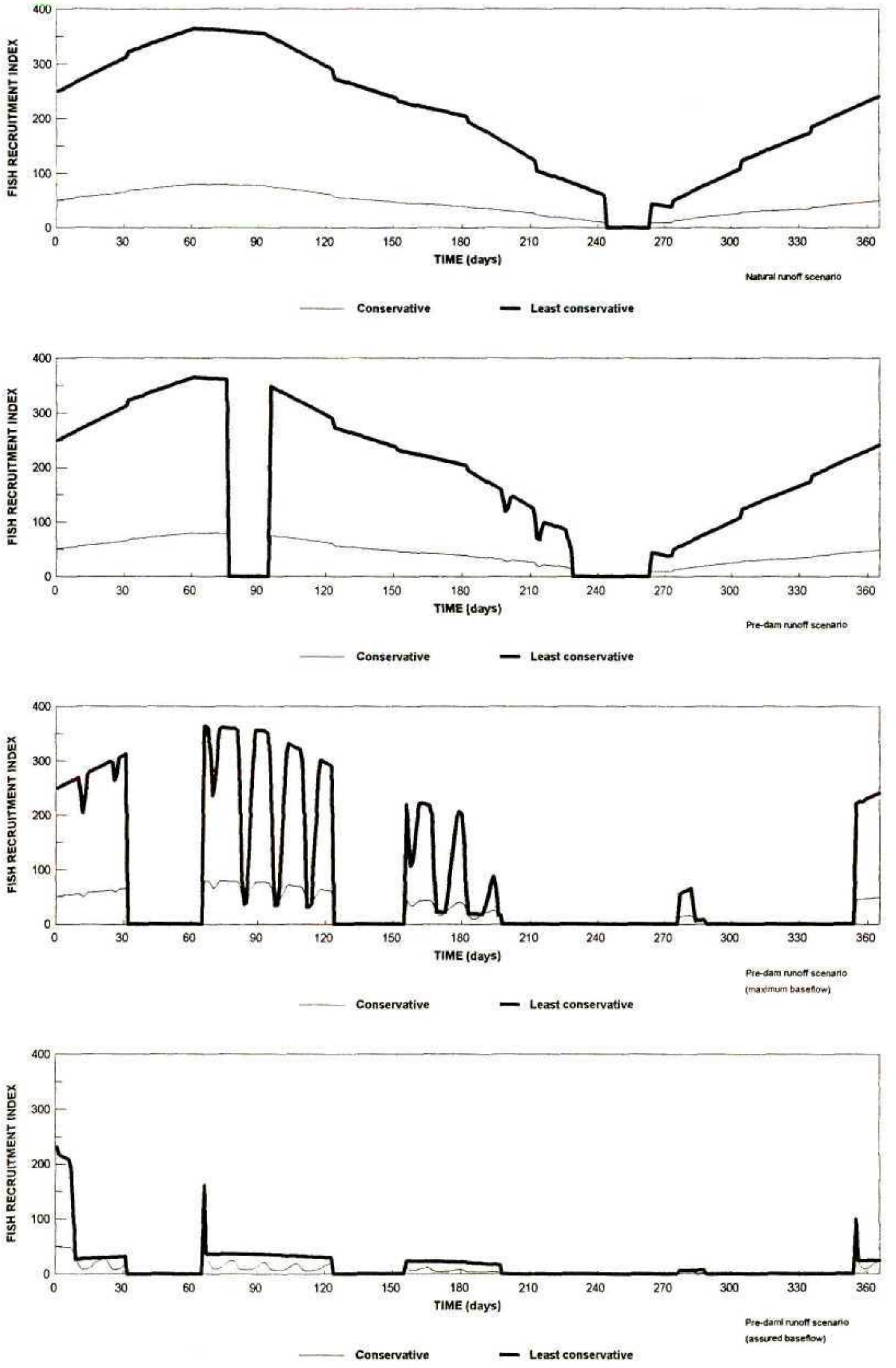


Figure 7.3 : Fish recruitment index for the natural runoff scenario, pre-dam runoff scenario, and the two post dam runoff scenarios under maximum and assured baseflow respectively, assuming a 50% reduction in baseflow during March to June.

(v) Disturbance scenarios

The two disturbance scenarios (1:50 year flood and 50% less baseflow during the period March to June) are characterised by slightly increased ($35.6 \times 10^6 m^3$) and sightly decreased ($30.0 \times 10^6 m^3$) mean annual runoff with respect to the natural average conditions ($34.7 \times 10^6 m^3$). However all three inflow scenarios permit the maintenance of an adequate axial salinity difference and thus while the mouth is open, recruitment is predicted to occur at full potential resulting in no differentiation between the scores for the three scenarios and a mean annual fish recruitment index of 206.

Under pre-dam conditions, the two disturbance scenarios have relatively greater effects, resulting in a slightly higher mean annual fish recruitment index for the flood scenario (188) in comparison with the average conditions (185) and the reduced baseflow conditions (184). Under post dam conditions the fish recruitment index reduces to 94 for the average and flood scenarios, and to 84 for the dry scenario with 50% less baseflow between March and June. Finally under conditions of very reduced freshwater inflow the flood scenario results in a mean annual fish recruitment index of 33, while the other two scenarios have a mean annual score of 15.

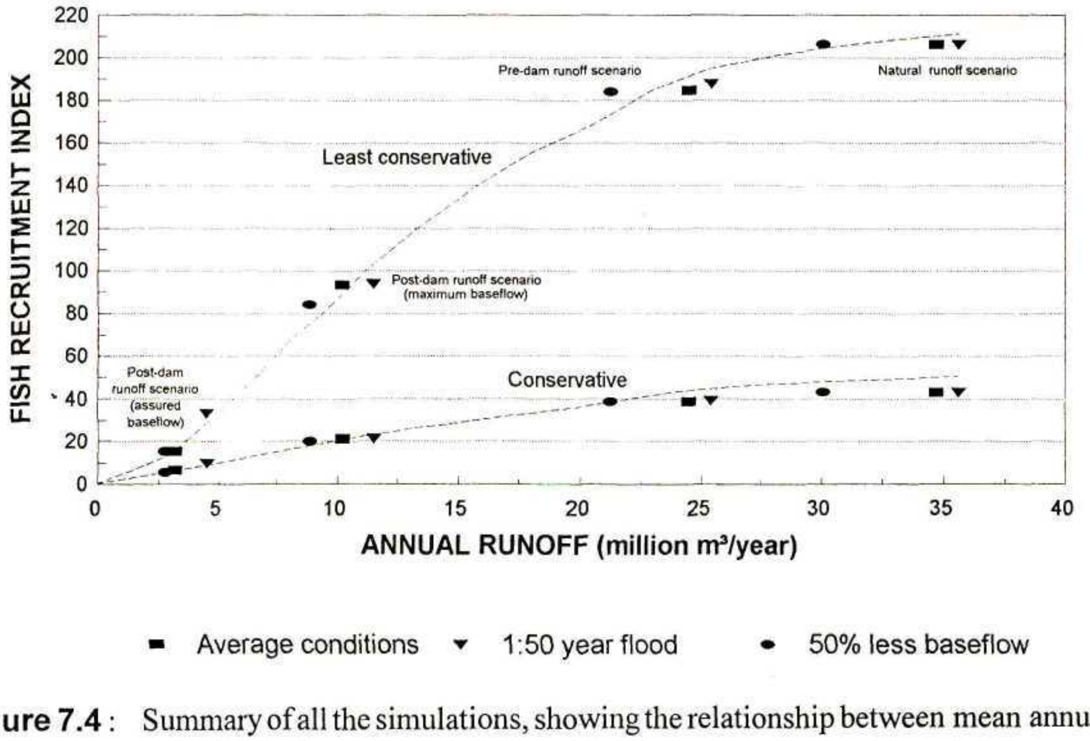


Figure 7.4 : Summary of all the simulations, showing the relationship between mean annual runoff and performance of the fish recruitment index.

7.1.2 Results of the Habitat Modelling

Results of the habitat modelling for the four scenarios under average conditions are shown in Figure 7.5. The figure summarises flux in water level as well as intertidal habitat and mean available habitat for *Upogebia africana*. As is evident from the four scenarios an increase in mouth closure results in higher water levels being maintained for longer periods of time and a cessation of tidal inundation. As a consequence, during these periods there is no intertidal habitat.

Figure 7.6 shows the decrease in mean annual intertidal habitat with reducing freshwater inflow for the four scenarios under average conditions. The change in total intertidal area as well as intertidal mudflats and intertidal areas characterised by fine sand is also shown. Thus the reduction in freshwater from natural conditions to the post dam scenario with assured baseflow results in a halving of the intertidal habitat.

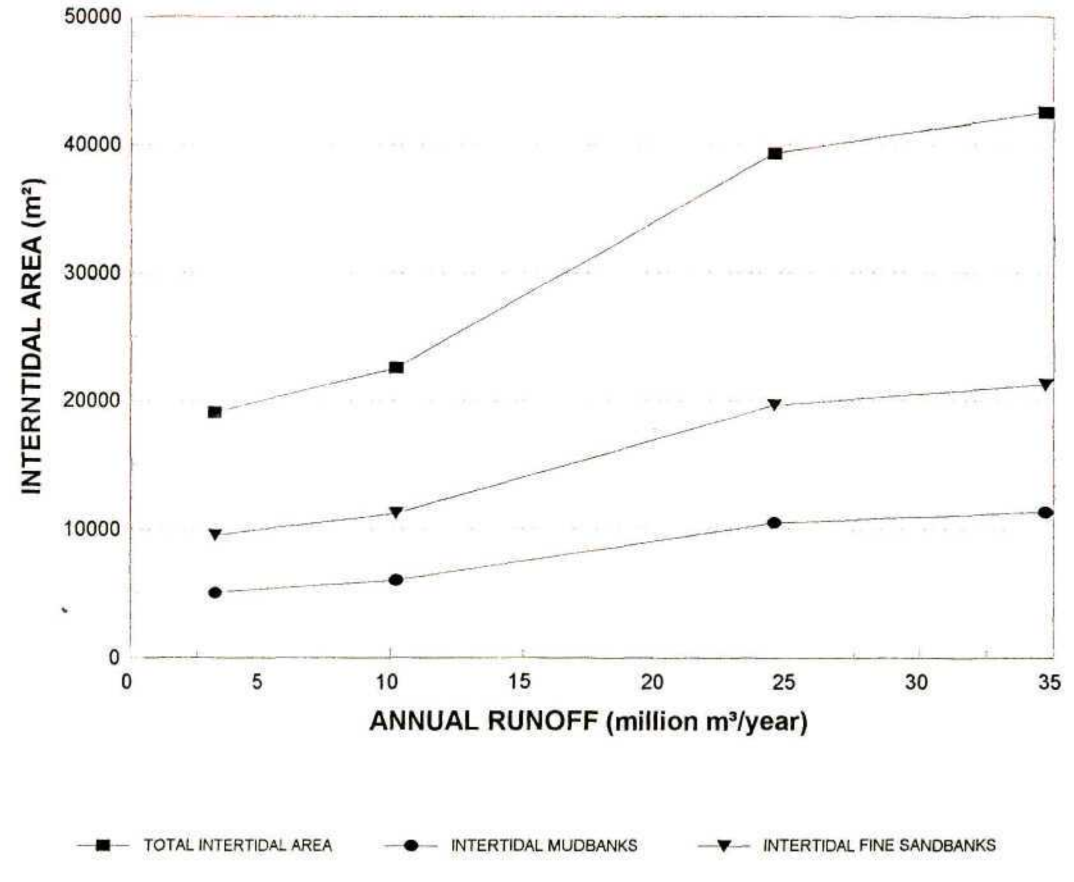


Figure 7.5 : Decrease in mean annual intertidal habitat, and intertidal habitats of different substrata, with reducing freshwater inflow.

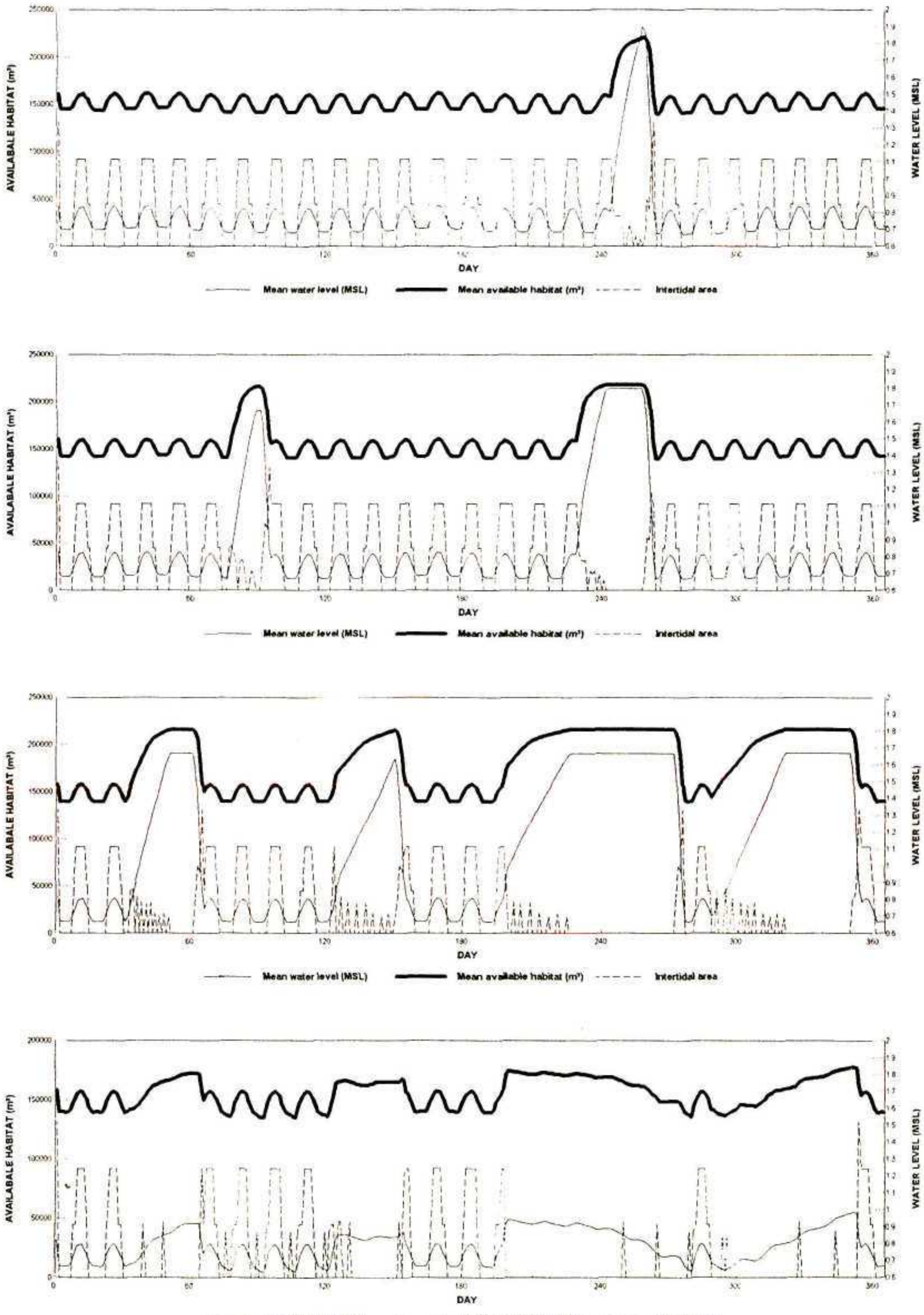


Figure 7.6 : Flux in water level, intertidal habitat and mean available *Upogebia africana* habitat for the four scenarios (natural runoff, pre-dam runoff, post dam runoff with maximum baseflow and post dam runoff with assured baseflow).

7.1.3 Results of the *Upogebia* Model

The *Upogebia* model provides several output variables, namely; total prawn population (number of individuals), the percentage of the population in the juvenile size classes (less than 7 mm carapace length), the number of postlarvae successfully recruiting back into the estuary, the population density (prawns.m^{-2}), the biomass of the population (g.m^{-2}), and the number of zoea exported to the marine environment. The model was run to steady state for each of the scenarios to permit the establishment of a population under the each of the different freshwater inflow regimes. Figures 7.7 to 7.10 show the results of the four scenarios under average conditions. The results of the two disturbance scenarios representing a 1 : 50 year flood and reduced baseflow conditions are given in Appendix B, Figures 1 to 8.

Under conditions of reduced inflow the population takes longer to establish and longer to reach steady state. Scenarios associated with higher inflow conditions show a greater proportion of the population consisting of juveniles. However, populations under low inflow conditions, although taking longer to establish, appear to persist with a higher biomass. As the total population numbers are relatively comparable for the scenarios, this obviously suggests that the prawns on average are larger, as is suggested by the increased proportion of adults in the reduced inflow scenarios. Consequently when the mouth does open relatively more zoea are exported and more postlarvae recruit back into the estuary, due to the fact that larger prawns have higher fecundity.

Chapter 5 referred to three management indices, namely; the total population number, the biomass and the number of eggs produced. With respect to the latter management index, eggs which hatch and zoea cannot be exported to the marine environment die within 24 hours, and consequently the number of exported zoea was considered a better management index than simply the number of eggs produced. The mean annual management index in respect of these three attributes are shown in Figures 7.11 to 7.13., in relation to the freshwater inflows which characterise each scenario. The performance of the population under natural conditions was selected as the reference simulation. In addition to these three management indices the other three outputs of the model (proportion of juveniles in the population, number of returning postlarvae and the population density) are also expressed as normalised indices in relation to freshwater inflow and are shown in Figures 7.14 to 7.16.

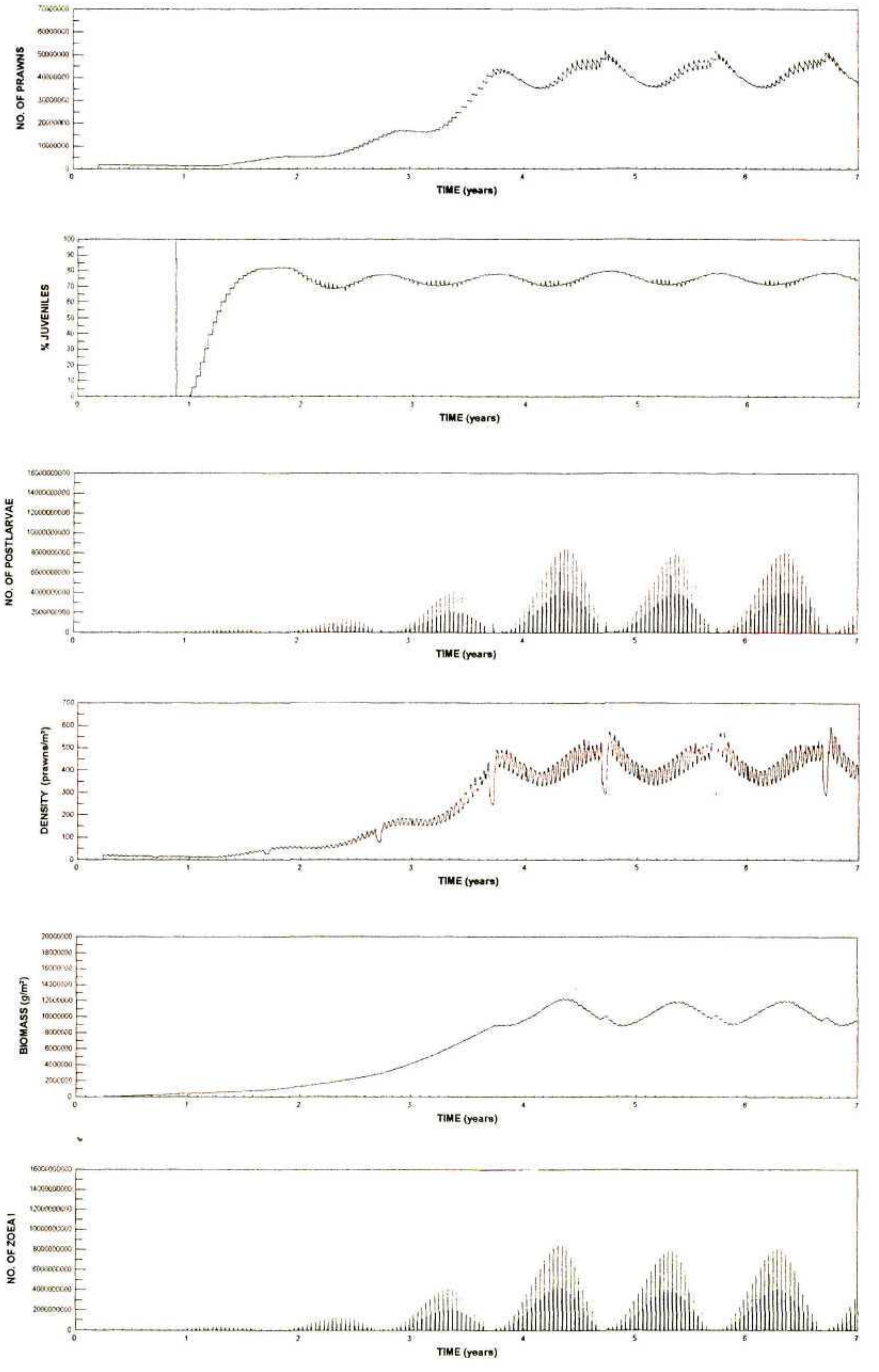


Figure 7.7 : Results of the *Upogebia africana* model run to steady state for natural runoff conditions.

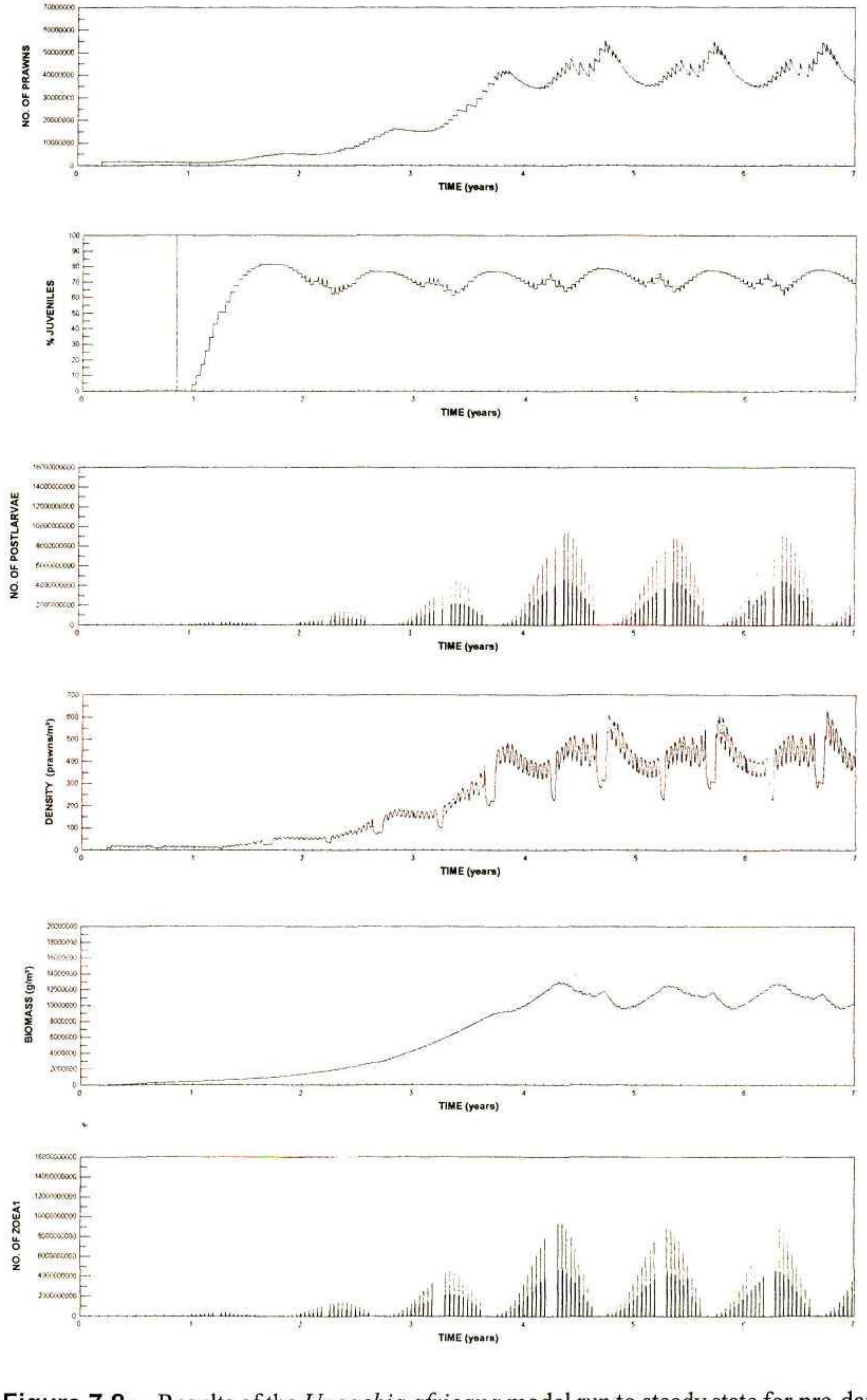


Figure 7.8 : Results of the *Upogebia africana* model run to steady state for pre-dam runoff conditions.

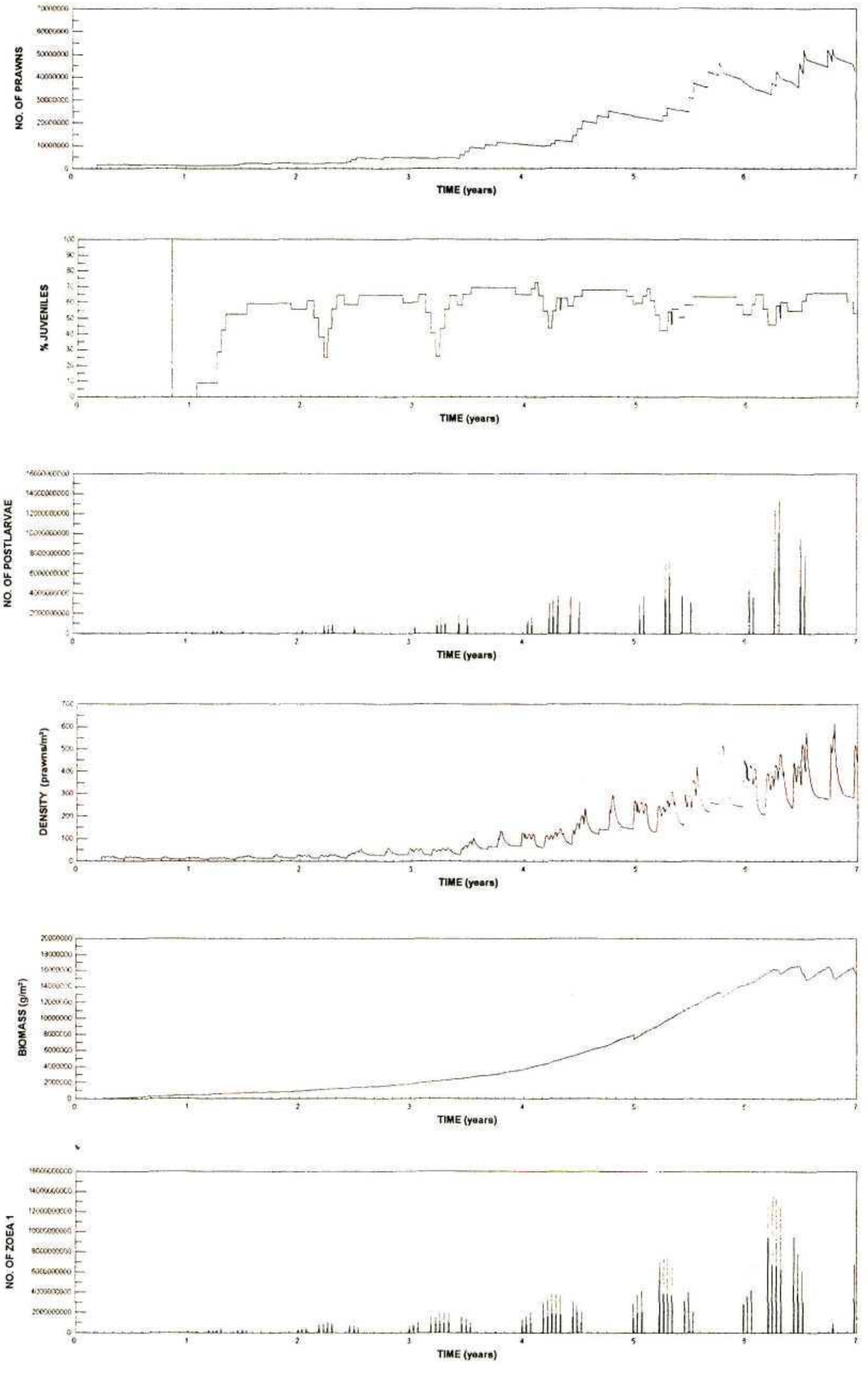


Figure 7.9: Results of the *Upogebia africana* model run to steady state for post dam runoff conditions, with maximum expected baseflow.

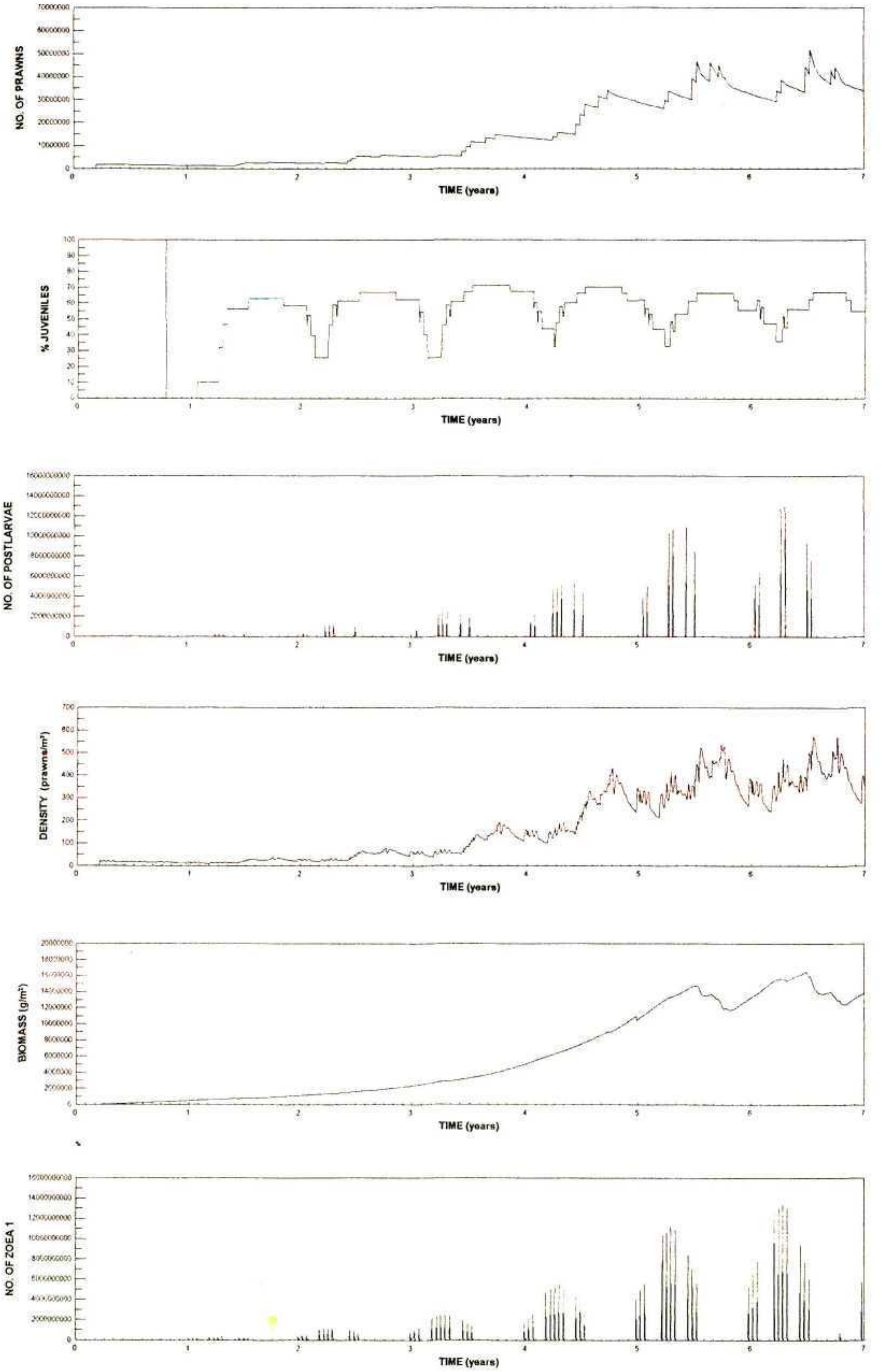


Figure 7.10 : Results of the *Upogebia africana* model run to steady state for post dam runoff conditions, with assured baseflow.

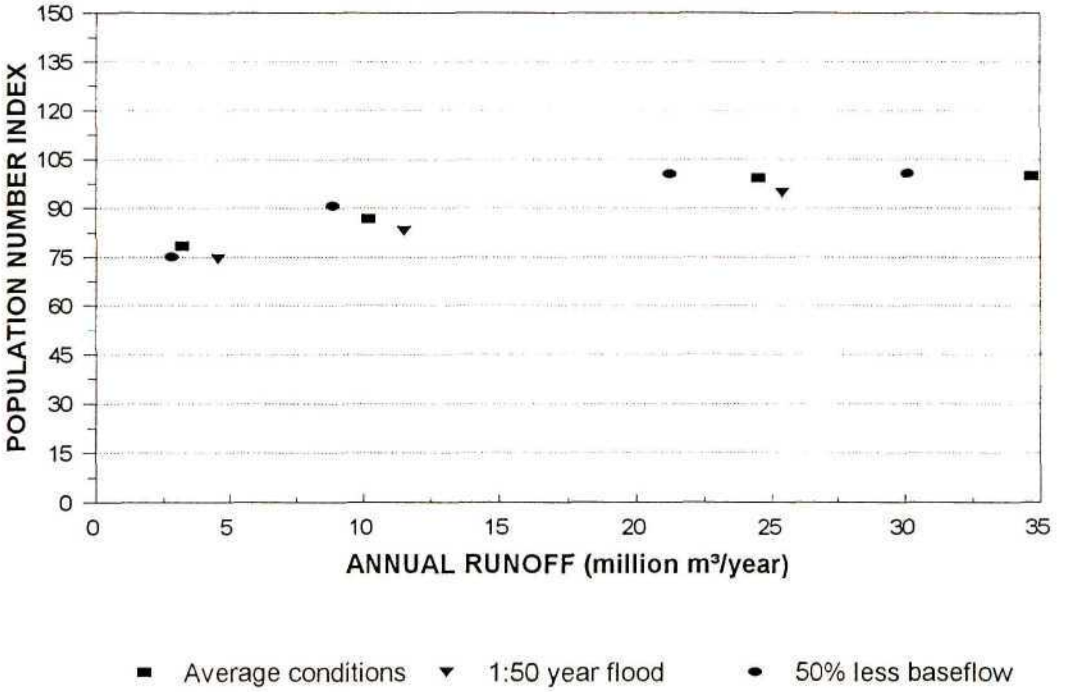


Figure 7.11 : Performance of the population number management index and associated freshwater inflow for all 12 scenarios.

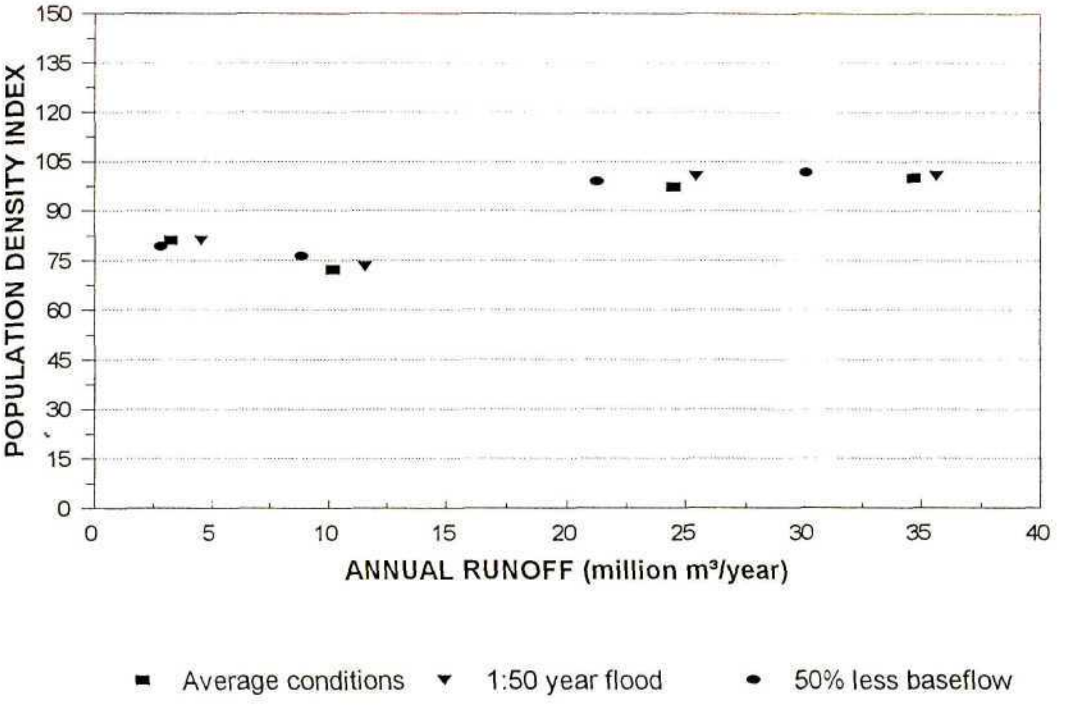


Figure 7.12: Normalised index of population density and associated freshwater inflow conditions for all 12 scenarios.

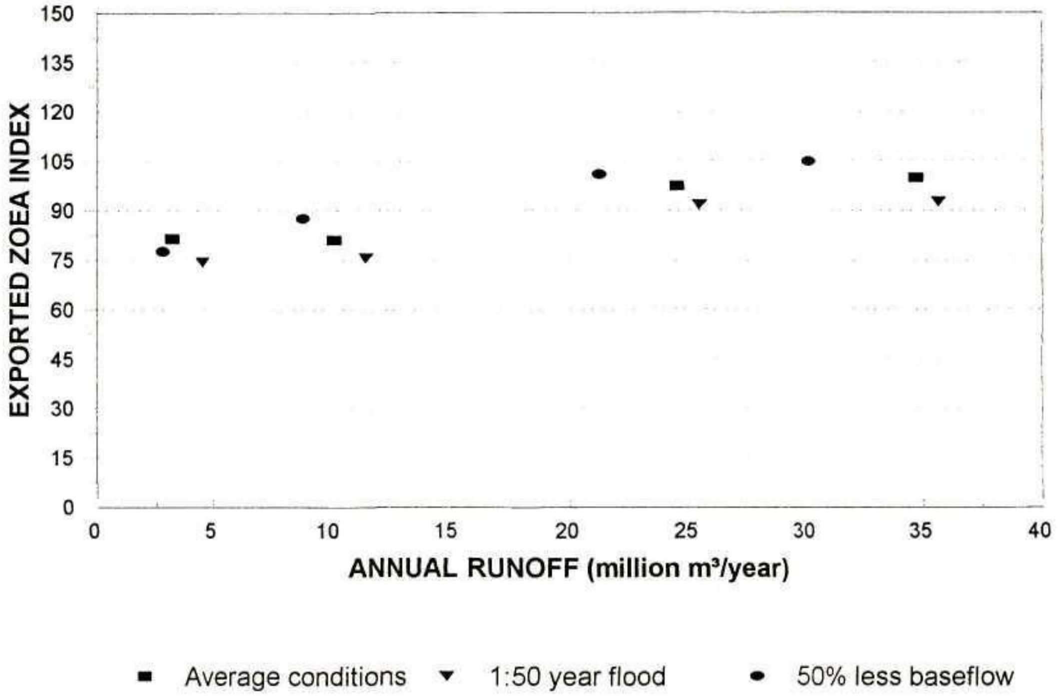


Figure 7.13 : Performance of the exported zoea management index and associated freshwater inflow for all 12 scenarios.

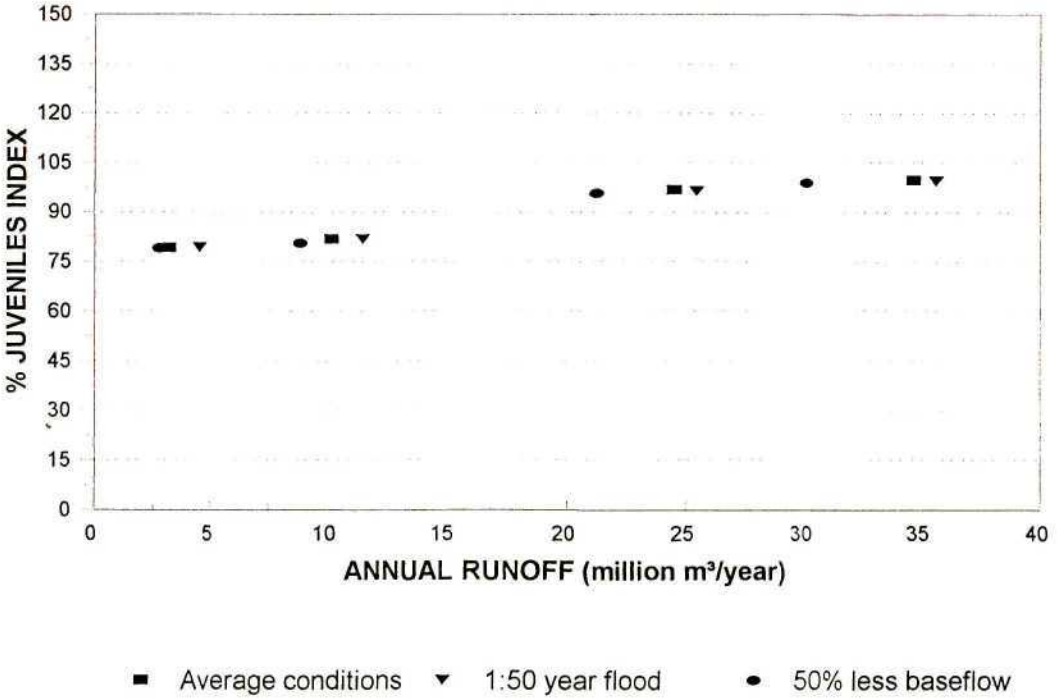


Figure 7.14: A normalised index showing proportion of the population which are juveniles for associated freshwater inflow conditions for all 12 scenarios.

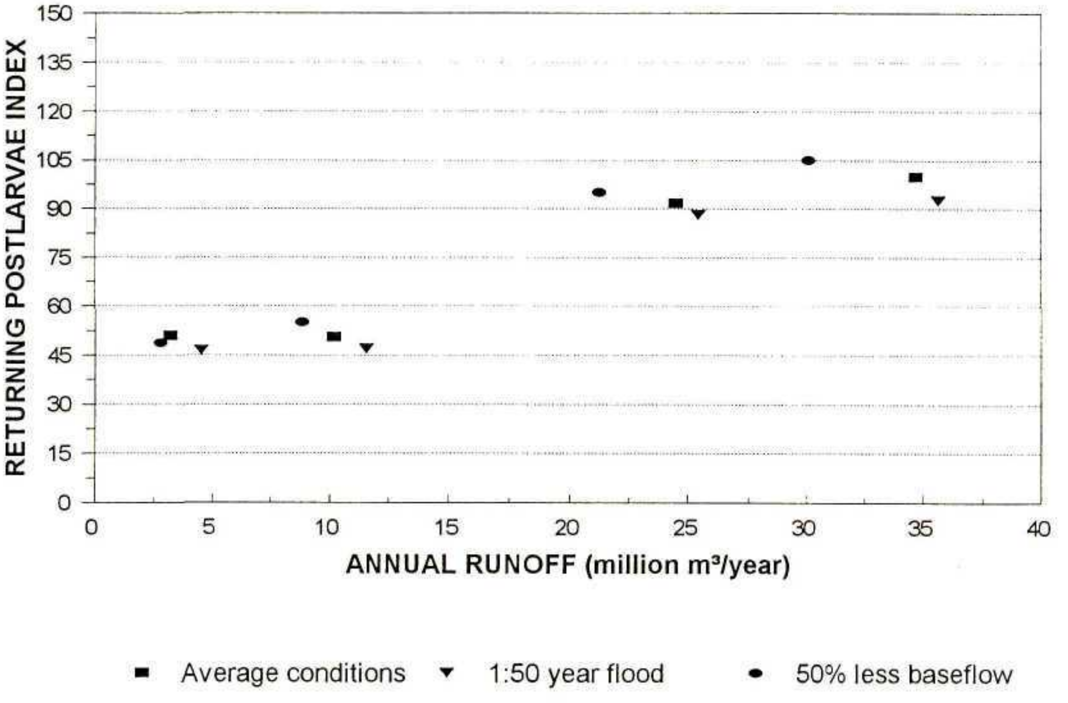


Figure 7.15 :Normalised index of the number of returning postlarvae and associated freshwater inflow conditions for all 12 scenarios.

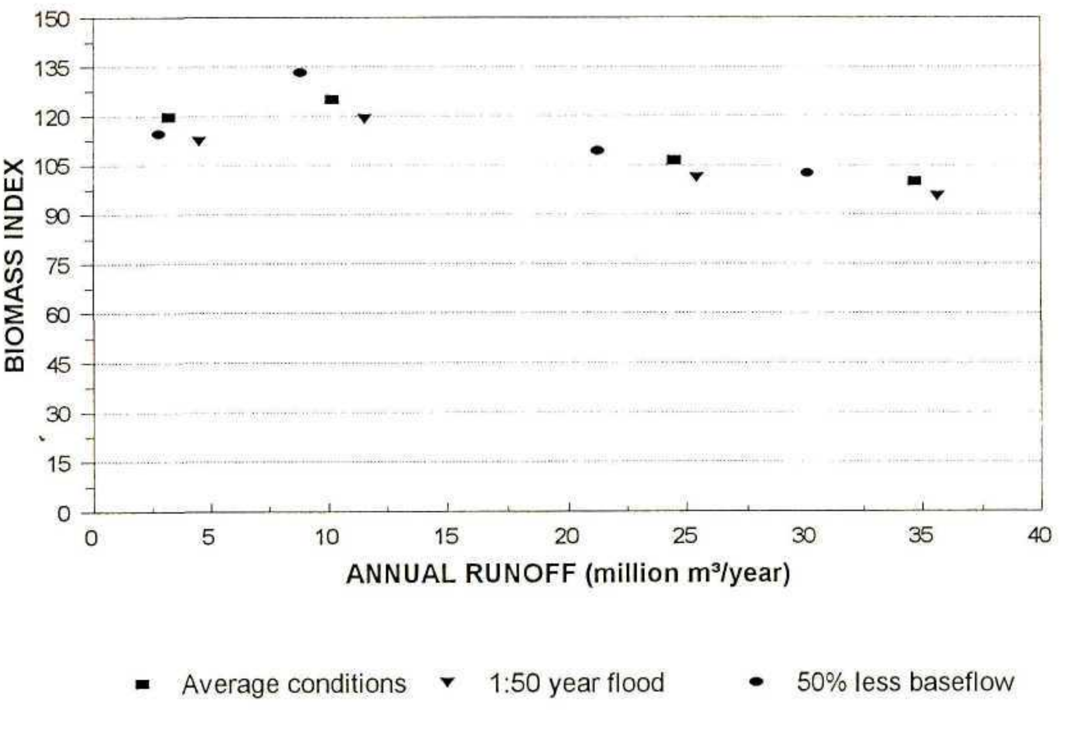


Figure 7.16: Performance of the biomass management index and associated freshwater inflow for all 12 scenarios.

7.2 DISCUSSION

7.2.1 Discussion of the fish recruitment index

A primary assumption of the Fish recruitment index is based in the observation that recruitment of juvenile marine fish is limited in the absence of strong axial salinity gradients. Understanding of recruitment processes of juvenile marine fish is currently very limited, and is hampered by the difficulties of larvae identification and sampling. While it has been suggested that juvenile fish are not responding to the axial salinity difference *per se* (Whitfield 1994b), it is not known whether all species respond to the flow conditions which give rise to axial salinity differences in the same way. Until evidence to the contrary is proposed, equal response of juvenile marine fish to prevailing axial salinity differences is assumed, and as this assumption is associated with the greatest degree of uncertainty, it may be the major weakness of the approach.

An additional source of error may be the assumptions regarding the recruitment preferences of the species included in the model. Although these data were compiled from a wider source of information and greater period of observation than the data pertaining to recruitment under various axial salinity gradients, it should be borne in mind that these data represent seasonal trends only. Recruitment of postlarvae is most probably more directly related to sea temperatures which trigger spawning and regulate the rate of egg and postlarvae development. This would suggest that the same species might recruit over a longer period in the warmer subtropical waters, in comparison with the cooler temperate waters.

The results of the fish recruitment index are also based on the assumption that the 26 species which are represented in the model adequately reflect the range of species utilising estuaries. However as further information regarding the recruitment preferences of other species becomes available, it can be easily added to the formulation. Similarly the weighting for each category of dependency or the weighting for endemism can be readily adjusted.

Although the fish recruitment index is based on several assumptions which still require additional verification, the formulation does combine the current understanding regarding recruitment processes with a value judgement based assessment of the conservation importance of each species considered in the model. If the assumptions of the model are found to be valid, then this methodology would provide a means to objectively compare different freshwater inflow scenarios on the basis of the consequences for the 'nursery value' of estuaries; one of the primary ecological functions attributed to these ecosystems.

Notwithstanding the considerable scope for testing and further development of the fish recruitment index, initial application in the Great Brak estuary suggested that annual runoff to the estuary could be halved without an appreciable decrease in the fish recruitment index, but with a potentially sharp decline thereafter. While the index represents recruitment opportunity rather than actual recruitment, the index does at least provide an indication of where ecological risk is likely to increase substantially.

7.2.2 Discussion of the Habitat Modelling

Chapter 4 presents a methodology for assessing change in various physical attributes of the estuarine environment. In particular, a method for assessing change in habitat which could potentially be used by a species is proposed. However these approaches are based on several key assumptions, the most significant of which relate to the manner in which data are obtained. As noted in Chapter 4, bathymetric characteristics, as well as sediment and vegetation distribution are seldom mapped at the same scale with the same degree of accuracy, and consequently it is usually difficult to integrate these datasets. In the case of the Great Brak estuary, for example, sediment distribution was mapped on the basis of schematic diagrams and narrative descriptions.

There are obvious advantages to presenting spatial data in the manner proposed. For example mapping these attributes at regular intervals permits rapid assessment of change, thus overlaying bathymetric maps compiled at different times will identify where deposition and erosion have occurred. Similarly two vegetation maps would show where die-back or colonisation is occurring, providing a useful monitoring tool. More importantly, this approach allows assessment of components in relation to other elements of environmental change. Variation in water level (exposure and innundation) and salinity are usually the primary factors influencing the distribution of vegetation in estuaries (Adams 1994), and often give rise to a distinct zonation (Day 1981d). As the tolerances of species to innundation are well known (Adams 1994), it is possible to predict the consequences of changing water levels for different vegetation communities. Indeed, if the approach described above provides a meaningful framework for the analysis of habitat potential in an estuary, there should be a strong correlation between the occurrence of vegetation and seasonal fluctuations in inundation and exposure, particularly given that water level appears to be a primary factor in determining habitat for vegetation (CSIR 1990). Testing of this hypothesis was not possible as the bathymetric and vegetation maps upon which the habitat analysis is based, were compiled from different sources, collected for different purposes and required considerable manipulation to be integrated into the same dataset. As efforts to produce a dynamic estuarine vegetation model are currently underway (Busse 1996, *pers. comm.*), this aspect was not pursued further.

A useful output of the approach which has been demonstrated however, is the facility to demonstrate the change in intertidal area, and species-specific habitat associated with each of the freshwater inflow scenarios. Intertidal areas are considered an important component of estuary habitat and are particularly important for particular groups of species such as waders. Estuarine mudbanks and sandbanks are also important for roosting birds (Cyrus 1996), if the prevailing water level becomes too high (>15 cm) these habitats are then not available to roosting birds. Although not demonstrated for the Great Brak estuary, precisely the same methodology can be used to evaluate flux in roosting habitat for birds as was used to determine change in intertidal area. Similarly the change in any defined set of environmental factors can be evaluated. For example if a biologically significant vegetation community is known to flourish under a particular inundation regime, the consequences of changing the freshwater inflow and therefore the innundation regime can be explored.

One key assumption of the above approach in its current form should however, be noted. In the application of the approach to the Great Brak estuary, a constant water surface elevation was assumed. In reality the water surface slopes seawards, and thus at the same inflow rate the water may be at 0.71 m (MSL) at point A in the estuary and 0.68 m (MSL) at point B, closer to the estuary mouth. The implication of this is that the analysis of habitat should be compartmentalised into areas of similar water surface elevation. As hydrodynamic models compute water levels for each of the contiguous network of cells, these data are usually readily available. The analysis of habitat in the estuary would then occur by hydrodynamic cell, and would be summed to provide an overall value for the estuary.

7.2.3 Discussion of the *Upogebia* Model

The *Upogebia africana* model represents a first attempt to link processes operating at a physiological scale to those operating at a population scale, in an approach to the management of estuaries in South Africa. This need has recently been recognised as an important component of ecosystem management, "*complete understanding or resolution of issues usually requires integration across several scales and levels of organisation. To determine mechanisms, we must often investigate processes operating at lower levels of organisation (say physiology or reproductive biology in the endangered species populations) as well as appreciate the context or higher levels of organisation within which the processes operate*" (Christensen *et al.* 1996).

This approach has consequently relied on several assumptions, the most significant of which probably concern the translation of a laboratory based understanding of the organisms response to change in temperature and salinity to that which may occur in a natural system. The assumed tolerances and preferences for salinity and temperature, the growth rates for a given physiological age, and indeed the very notion of a physiological age all require further investigation. As already noted elsewhere in the text, these tolerances are likely to be life-stage dependent, whereas the model formulation has assumed similar responses for all life stages. Important adaptive mechanisms such as acclimation have not been considered and neither has the likely notion that geographic distribution (and therefore seasonal temperature regime) is also likely to have a bearing on the response of a population to environmental change.

Assumptions regarding recruitment of postlarvae are perhaps the weakest of all. Virtually nothing is known about mortality of zoea in the marine environment and the conditions under which postlarvae return to the estuarine environment have only recently been determined (Wooldridge & Loubser 1996). The model has however provided evidence that continual recruitment into populations of *Upogebia africana* within an estuary is strongly dependent on the existing population. Should a population within an estuary be lost due to mouth closure or a discrete flooding or pollution event, re-establishment by recolonisation from an adjacent estuary may occur slowly.

The limits of population growth are also not well understood. Although maximum recorded densities approach 500 individuals per m^2, the dynamics of recruitment and the effects of predation are not known. For example it is not known whether populations are maintained at this upper level by predation or juveniles are not able to establish themselves if adult densities are too high. Despite these uncertainties, in formulation of the model a density dependent mortality is assumed. Thus increasing mortality is assumed to occur across all size classes as the population reaches its upper assumed limit. This assumption has yielded an interesting result in the behaviour of the model.

At low inflows the mouth is closed more frequently and consequently recruitment is limited. The population therefore takes a longer time to establish and is ultimately characterised by lower densities than the population established by a natural flow regime. In contrast, however, populations established under a reduced freshwater inflow regime appear to be characterised by a higher biomass, due to the fact that a smaller proportion of the population are juveniles. Because densities remain low during the establishment phase prawns are not subject to density dependent mortality and therefore relatively more grow to their maximum size, producing relatively more eggs and zoea. Thus in the low flow scenarios, while export of zoea occurs less frequently, more zoea are produced per export event.

Under higher inflow conditions the population reaches its maximum sooner bringing about density dependent mortality. However, as recruitment is not limited, replenishment of the population by postlarvae occurs more continuously resulting in a higher 'turnover' of prawns and fewer large adults. As a consequence biomass remains relatively lower, the population is characterised by a relatively higher proportion of juveniles, and the number of zoea and returning postlarvae are reduced with respect to the disturbed conditions.

An alternative assumption could be that as a population increases the chances of a cohort of postlarvae establishing successfully decrease. This could be as a result of them settling in less favourable regions of the estuary or directly by some form of undescribed competition. However if the initial assumption of density dependent mortality is not an artefact of assumption and is indeed correct, this may provide compelling evidence for the requirement of maintaining a measure of variability in estuarine systems. The American Society of Ecology has argued that variability is an essential requirement for healthy ecosystem functioning, "*sustainability does not imply maintenance of the status quo. Indeed change and evolution are inherent characteristics of ecosystems, and attempts to 'freeze' ecosystems in a particular state or configuration are generally futile in the short term and certainly doomed to failure in the long term*" (Christensen *et al.* 1996).

Despite the numerous assumptions and considerable degree of conjecture inherent in the current formulation of the model, preliminary testing did appear to be able to predict, at least in broad terms, patterns of response to the historical record of the mouth status. Indeed from a management perspective many of these assumptions are less relevant as a comparative assessment is usually required. In any event, the primary purpose of developing the model was to demonstrate an approach to modelling which would provide a basis for ecologically sustainable management, and while the approach may be limited by lack of understanding in the short term, it does provide a framework to test research hypotheses as well as a means of identifying research needs.

While this approach has been developed for *Upogebia africana* for the several reasons outlined in Chapter 5, it could equally be applied to other species with known life-stage requirements. Table 7.1 below provides a summary of other species which may be amenable as indicator species.

Table 7.1 : Additional species which may be amenable to the proposed modelling approach

KEY SPECIES		LIFE HISTORY
Rhabdosargus holubi	FISH	Marine resident, estuarine phase in life cycle
Ambassis natalensis	FISH	Estuary resident, marine spawning
Scylla serrata	CRAB	Estuary resident, marine spawning
Callianassa krausii	PRAWN	Typical estuary resident
Penaeus indicus	PRAWN	Marine resident, estuarine phase in life cycle
Sesarma catenata	CRAB	Estuary resident, marine spawning

8 CONCLUSIONS AND RECOMMENDATIONS

8.1 CONCLUSIONS

Resource management in South Africa is on the threshold of a new era; an era to be characterised by public participation, openness and transparency, and where the fundamental principle is the management of natural resources to achieve long term environmental sustainability. South Africa is not unique in this respect as is illustrated by the United States of America, where "*in recent years, sustainability has become an explicitly stated, even legislatively mandated, goal of management agencies charged with the stewardship of natural resources*" (Christensen *et al.* 1996).

However as suggested by Goodland (1995), while environmental sustainability is '*a rather clear concept*', there is considerable uncertainty regarding the details of its application. In part, this uncertainty stems from a lack of understanding of how to evaluate the criterion of environmental or ecological sustainability; particularly given the intergenerational requirement often associated with environmental sustainability. In other words, how do we know *now* that we are managing ecosystems which will still be healthy in the future. The concept gives rise to a cascading series of critical issues. For example what do we measure or monitor to determine the health or level of functioning of ecosystems? As we cannot possibly monitor all processes or organisms in all environments how do we choose which to measure and where? Furthermore what constitutes acceptable health and what becomes unsustainable?, particularly since what may be unsustainable for one organism may be ideal for others.

These complexities of ecosystem management have long been recognised, often resulting in utilitarian management approaches guided by the premise that it is possible to "*simplify the structure and composition of ecosystems to achieve efficient production of specific goods such as timber, fish or agricultural crops at no risk to sustainability*" (Christensen *et al.* 1996). As recognised by Goodland (1995), the historical management milieu has sought to establish the optimal, and usually the most profitable, harvest rate for natural products such as timber or fish. The focus of research has thus been to determine the 'Sustained Yield', on the assumption that this is equivalent to sustainable management. Both Goodland (1995) and the Ecological Society of America Committee on the Scientific Basis for Ecosystem Management (Christensen *et al.*

1996) support the view that finding an optimal solution for a single component of an ecosystem "*usually (possibly inevitably) results in declining utility or declining natural capital sometime in the future, and is therefore not sustainable*" (Goodland 1995). Indeed, current approaches to the management of freshwater inflow to estuaries along the Gulf of Mexico still appear to be based on the notion of sustained yield. Allocations of freshwater are determined on the basis of historical harvest data, albeit for several species. The objective of management is maximisation of fish, prawn and shellfish harvests, with the expectation that not only are these harvests sustainable, but also that the needs of all other components of the ecosystem will be met.

The review of current approaches to the determination of the freshwater requirements of estuaries indicates that little attention has been directed towards addressing this issue, despite the fact that it appears that diminishing freshwater supply to estuaries is a growing international concern. Methodologies which have evolved are somewhat parochial, tailored to the particular characteristics of the estuarine system, such as the San Francisco Delta-Bay complex, or rely on substantial historical, estuary-specific datasets, such as is the case for the Gulf estuaries. In addition to being inappropriate for South Africa and any other developing nations due to the requirement of substantial datasets, these approaches are unlikely to be able to demonstrate ecological sustainability and are therefore fundamentally flawed.

Recognising the need to address the issues of ecological sustainability more holistically, as well as the imperatives implicit in the pending legislation and policy in South Africa, this study set out to develop a resource management approach for estuaries which contributes to demonstrating ecological sustainability. A management approach and set of predictive techniques have been developed, applied in a case study and critically reviewed. The approach is designed to minimise the uncertainty of managing for ecological sustainability by providing the means to demonstrate the ecological consequences of management actions. It is argued that not only is this approach consistent with the principles of adaptive management, but considerably enhances their applicability through emphasising good modelling practice with directly linked and compatible monitoring procedures. In so doing, the risks of adaptive management are foreshortened, if not by the predictive capability of models, at least by the insight generated through developing models and synthesising understanding.

While for the most part the approaches suggested in this study have not been tested, it should be recognised that they are entirely based on current understanding of estuarine ecosystems in South Africa. In other words the information upon which they are based *is* currently used by scientists to make decisions about estuary management, although in a considerably less integrated manner. Thus if nothing else, these models represent a structural framework for assessing the strengths and weaknesses of current wisdom. In this respect, the recommendations below in Section 8.2 identify the most urgent requirements for extending current understanding. Despite the fact that these models have had disparate and generally poor information sources for their formulation, their application in the Great Brak estuary yielded results which were not only consistent with the recorded consequences of reductions in freshwater discharge, but in the case of the *Upogebia africana* model, also provided new insights regarding the population biology of the species.

The subsequent sensitivity analysis illustrated the need for models, particularly the *Upogebia africana* model, to be subject to sensitivity analysis when utilised in an estuary management context. In doing so, it is imperative that the scientists using the model in a workshop setting are aware of the assumptions of the model and understand their implications. Not only does this provide a basis for designing the sensitivity analysis for the exercise, but on every occasion for use of the model, will provide the opportunity for scientists to improve assumptions or contribute new data or understanding into the model formulation.

Management of estuaries in South Africa is underdeveloped and usually occurs on an *ad hoc* basis. Furthermore, ecological objectives are seldom explicitly incorporated as management objectives, if indeed these exist at all. The Great Brak estuary was selected as a case study as this is one of the few estuaries in South Africa where management objectives have been identified and a management committee is in place to attend to the management of the estuary. It is also one of the few estuaries where a comprehensive, long-term monitoring strategy has been initiated. However, as is evident from the discussion in Section 2.2 and Figure 2.7 (page 44), ecological criteria are not explicitly stated in the estuary management approach, and decisions are taken primarily on the basis of either the dam level or the water level in the estuary. While ecological principles have guided the formulation of the flow diagram, the approach relies entirely on the results of the monitoring programme to evaluate the acceptability of past decisions. Furthermore, as no ecological goals have been explicitly stated management will tend towards avoidance of 'ecological disaster' situations rather than actually managing for sustainability.

The advantage of the methodologies introduced in this study is that the consequences of certain management actions can be more readily quantified. If this occurs within the context of the suggested management framework (Figure 2.8, pg. 52) progress in meeting specified ecological goals can be determined, and confirmed by means of a direct monitoring programme. In so doing, the defensibility of management actions will be increased.

Because these approaches have been developed for application in the context of the data poor situation in South Africa, it is anticipated that they will have general applicability in regions which are characterised by lack of information. Moreover, it is argued that these techniques would be directly applicable to the estuaries of 'developed' countries such as the United States of America, where species specific information is likely to be greater. Considerable progress has already been made in these countries regarding the use of optimisation methods to maximise indices of biological functioning and minimise flow releases, particularly for Texas estuaries. These techniques could be equally applied to the management indices proposed in this study.

Finally, although this study has presented several approaches to quantifying the impacts of freshwater flow reduction in ecological terms, it has not addressed the issue of goal setting. Further attention needs to be directed toward identifying the acceptability of change, thereby defining minimum standards for ecological performance, including the notions of risk and variability.

8.2 RECOMMENDATIONS

8.2.1 Recommendations emerging from the fish recruitment index

Application in data poor regions

(a) Assuming that the longitudinal salinity difference is a valid indicator of the extent of juvenile fish recruitment along other coasts, the index could be applied for these regions with limited data acquisition, through following the steps identified below :

- Undertake a classification of the estuarine fish fauna occurring in the region, with particular reference to those species which are dependent on estuaries, and those which are rare, endangered or endemic

- The preferred recruitment periods for key species should be identified. This may be a representative subset of species if sufficient information is not known for all the species identified above

- The model as formulated herein could then be applied and tested in a case study

Application in data rich regions

(b) Having undertaken the steps identified in (a) above the index could be validated through comparing the results of the index applied for certain species with historical observed catch data of that species, such as is available for many estuaries and several species in the Gulf of Mexico estuaries. Extension of the approach to include species other than those valued as commercial harvests would provide for a broader and more representative indication of the ecological sustainability of freshwater management policies in the region.

Further development of the approach

(c) Further investigation regarding the primary factors influencing the recruitment of juvenile marine fish into estuaries should be undertaken. This research should seek to establish causal mechanisms and should be oriented toward developing a predictive understanding of the processes of recruitment. In particular the suggestion that actual volume of freshwater entering the coastal environment should be further investigated. These findings should then be incorporated into the fish recruitment index.

(d) The recruitment preferences of additional species should be incorporated into the fish recruitment index, particularly those which are dependent on estuaries (Category II) and those which may be rare, endangered or endemic. This will ensure that the needs of a greater range of species are included in the index, and will contribute to building confidence in the approach. In addition, consideration needs to be given to whether geographic location should also be considered in determining the timing of recruitment into estuaries.

(e) The performance of the fish recruitment index needs to be tested against actual recruitment in relation to particular hydrodynamic conditions, including flow rates and flux in longitudinal salinity gradient.

8.2.2 Recommendations emerging from the Habitat Modelling

(a) Primary estuary datasets such as bathymetry, sediment and vegetation distribution need to be obtained by means of a standard protocol and at commensurate standards.

(b) Consideration should be given to linking the dynamic vegetation model currently under development (Busse 1996, *pers. comm.*) with the spatial capabilities of GIS.

(c) The concepts of habitat potential and habitat suitability as defined in this document need to be considered further. In particular habitat suitability weightings for *Upogebia africana*, and other important species should be determined.

8.2.3 Recommendations emerging from the *Upogebia* Model

Application in data poor regions

(a) If sufficient data are not available to implement the approach for a key invertebrate species such as *Upogebia africana*, a similar approach to that of the fish recruitment index could be adopted for invertebrates. The fish recruitment index approach could be extended to also include a measure of preference for prevailing salinities or temperatures.

Application in data rich regions

(b) As is the case for the fish recruitment index, the approach of the *Upogebia africana* model could be implemented for species where sufficient ecophysiological data are available. Similarly predictions could be evaluated against the wealth of historical data available for the commercially important species.

(c) Populations of *Upogebia africana* in the Great Brak estuary should be monitored at an increased level of frequency to determine whether the predicted seasonal decrease in density occurs or not. Temperature and salinity within the estuary should be monitored at fortnightly to monthly intervals to provide input to the model. This will permit more rigorous testing of the validity of the approach.

(d) Further investigation should be undertaken to determine levels of predation in established populations of *Upogebia africana* as well as to determine the processes of population regulation and the factors impacting upon successful recruitment. The results of such an investigation could provide a basis for the inclusion of more realistic assumptions regarding density dependent mortality.

(e) Consideration should be given to the inclusion of differential growth rates within cohorts in order to predict resultant size class frequency distributions more realistically.

(f) The temperature and salinity tolerances and preferences of all phases of the life cycle of *Upogebia africana* should be determined. In particular the phenomenon of acclimation needs to be properly addressed.

8.2.4 General recommendations

The above recommendations provide direction for further development of the approaches and suggestions for application of these approaches in both data rich and data poor contexts. Although the results of such investigation and testing may considerably improve the reliability and assuredness of the approach it should be stressed that by far the most important recommendation of this study is the implementation of these techniques in the context of the management approach proposed for estuaries. In particular :

(a) Attention needs to be directed at the development of protocols to assist in the formulation of management goals and objectives for estuaries. These goals should be determined through a full public participation process and should ultimately be expressed in terms which are outputs of the modelling approaches and can be readily monitored.

This study has contributed to the identification of potential indices which could be considered. For example, population number, biomass or percentage of the population which is juvenile in the case of the *Upogebia africana* model. However, as illustrated in the case study scenarios which had less freshwater inflow, sometimes these scenarios showed greater performance in an indicator such as biomass. Thus while one indicator may show improvement another (such as the actual size class structure of the population) may reflect an uneven distribution and therefore poorer measure of performance. The implication of this is that a range of indicators needs to be identified, and considerable attention needs to be directed towards identifying acceptable goals for ecosystem components.

(b) Finally, the management approach described herein should be implemented as a case study as part of an effort to identify a basis for a national management standard, subsequent to detailed evaluation and public comment.

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APPENDIX A

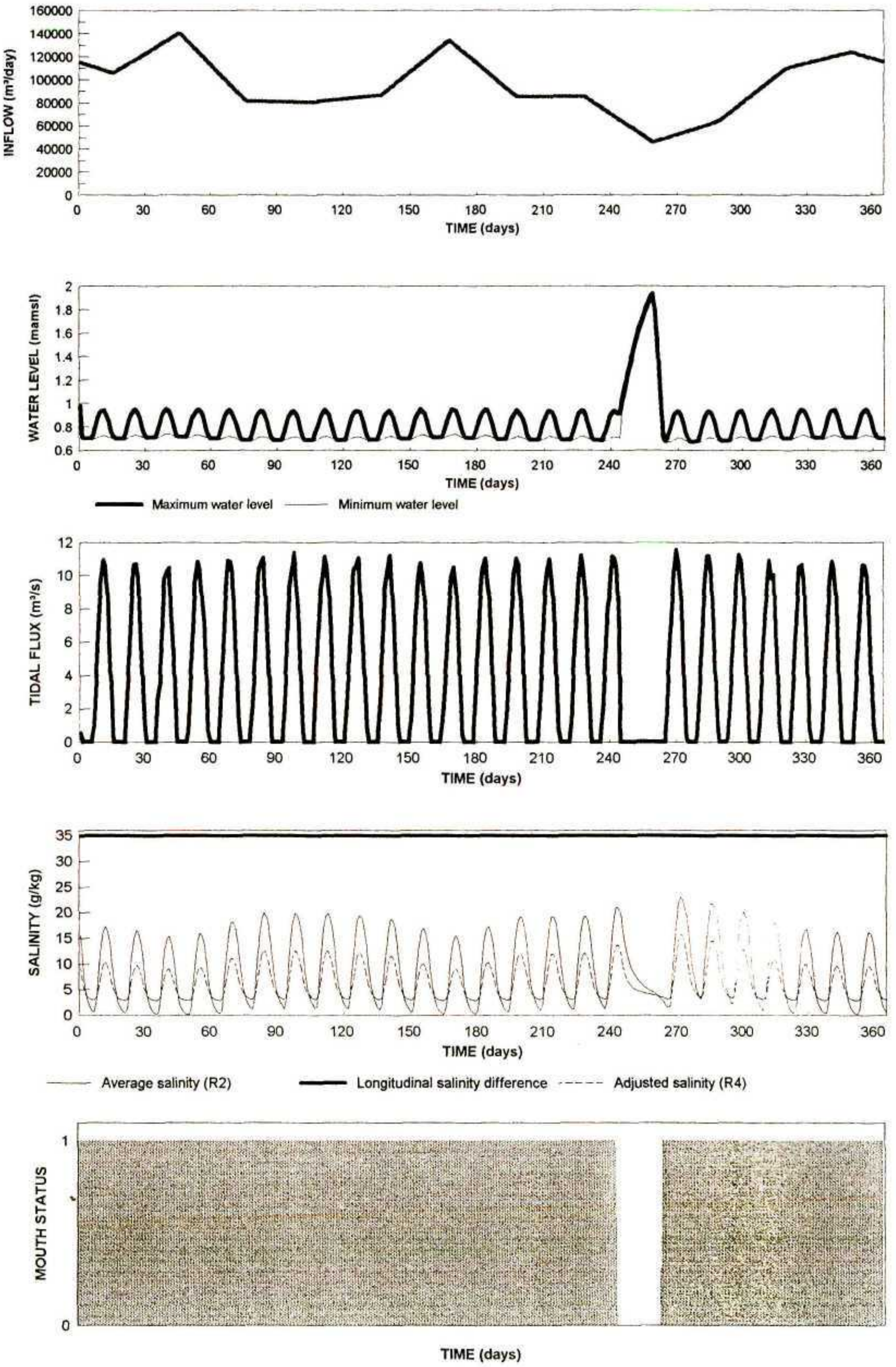

Figure 1 : Flux in key characteristics of the estuary environment in relation to freshwater inflow anticipated under natural conditions, with a 1:50 year flood in late summer (Slinger 1995)

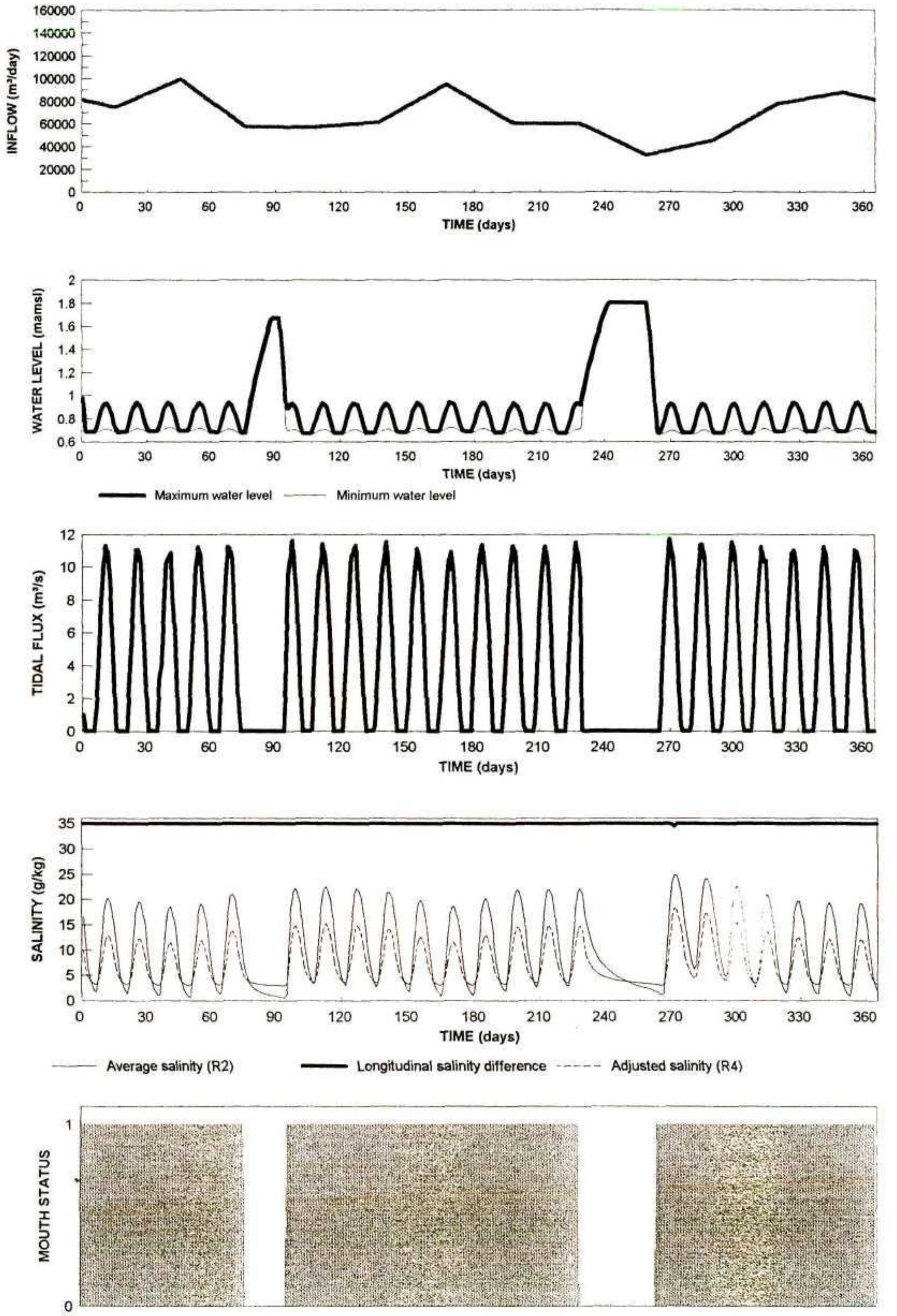

Figure 2 : Flux in key characteristics of the estuary environment in relation to freshwater inflow anticipated under pre-dam conditions, with a 1:50 year flood in late summer (Slinger 1995)

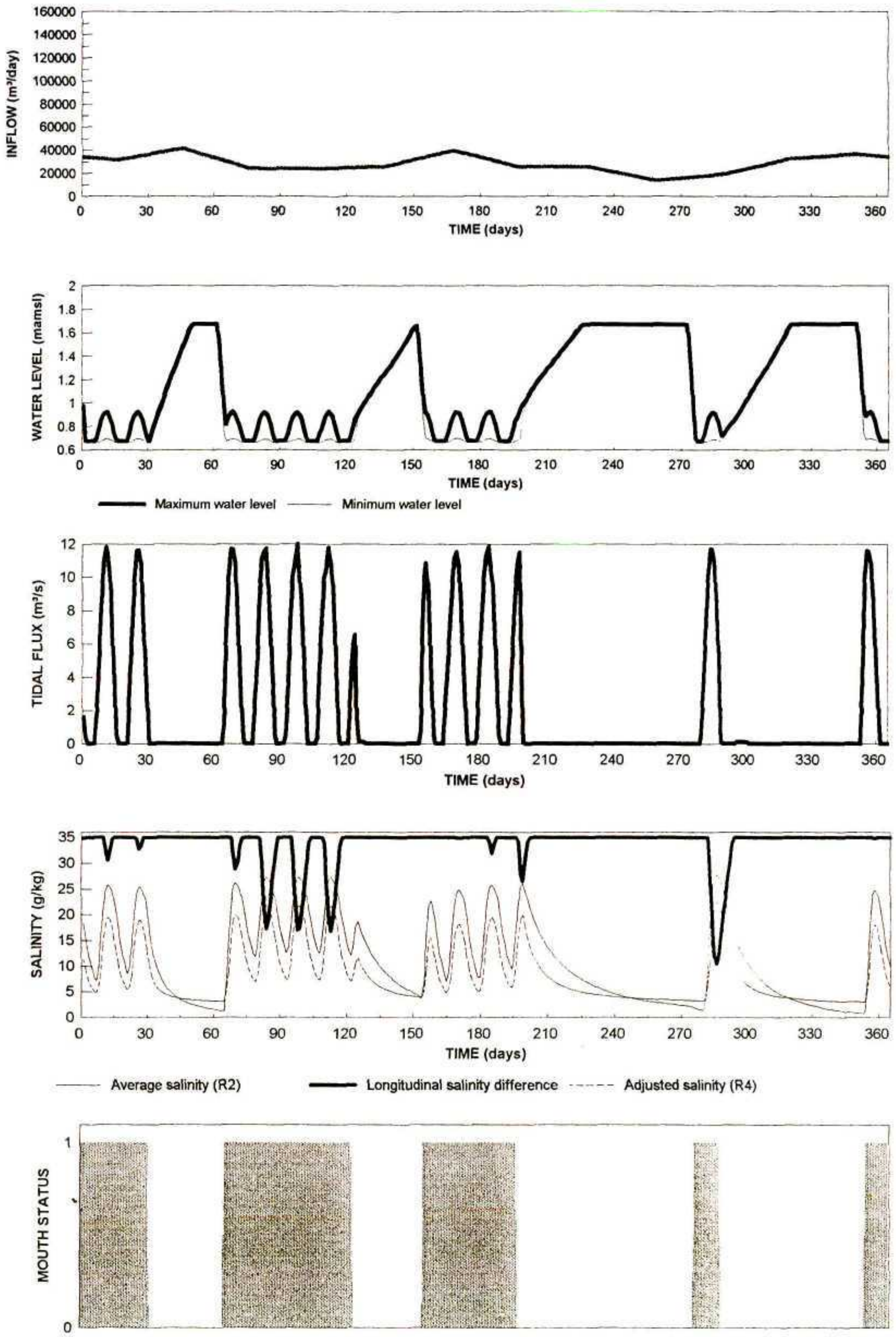

Figure 3 : Flux in key characteristics of the estuary environment in relation to freshwater inflow anticipated under post-dam, maximum baseflow conditions, with a 1:50 year flood in late summer (Slinger 1995)

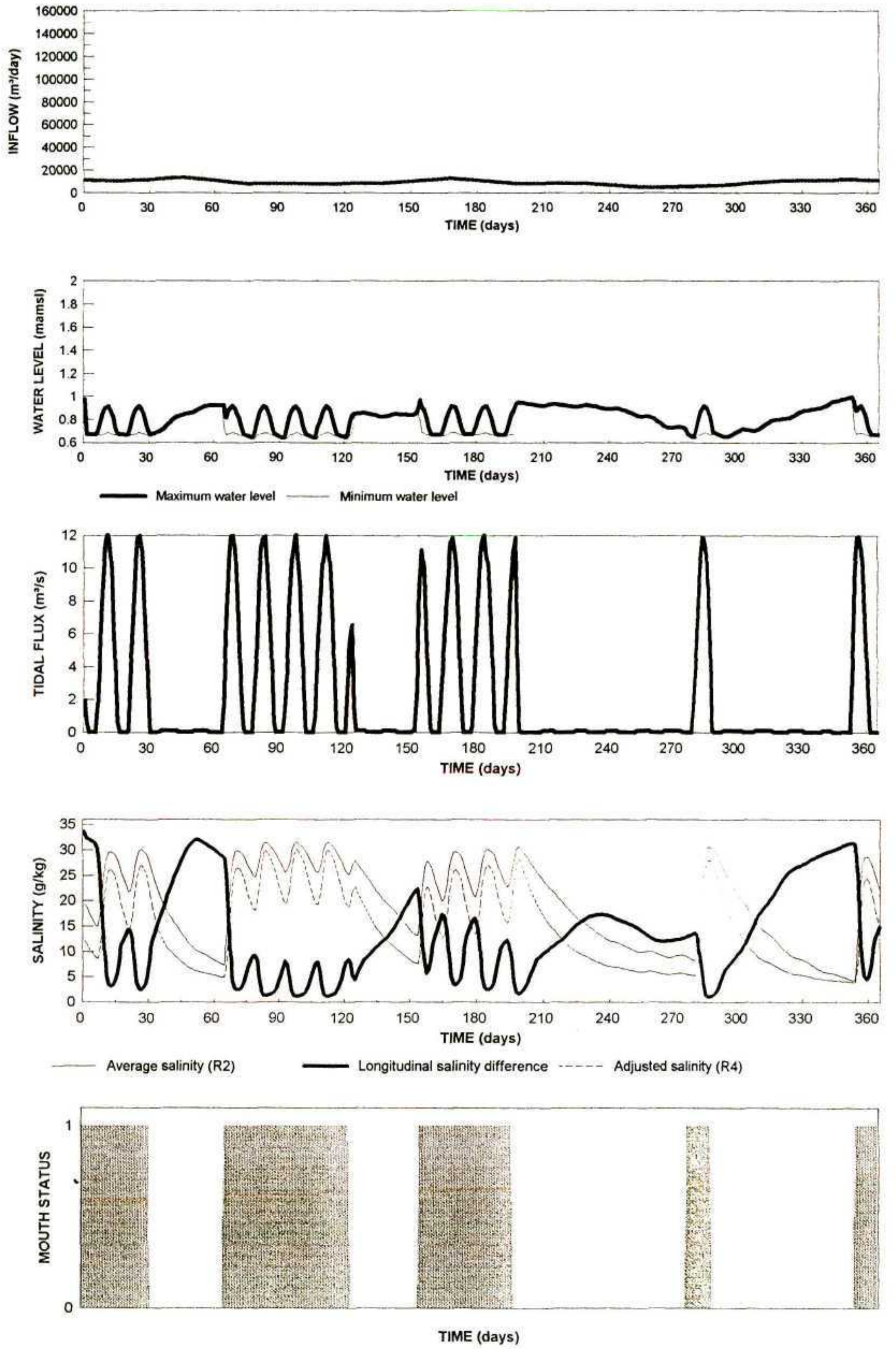

Figure 4 : Flux in key characteristics of the estuary environment in relation to freshwater inflow anticipated under post-dam, minimum baseflow conditions, with a 1:50 year flood in late summer (Slinger 1995)

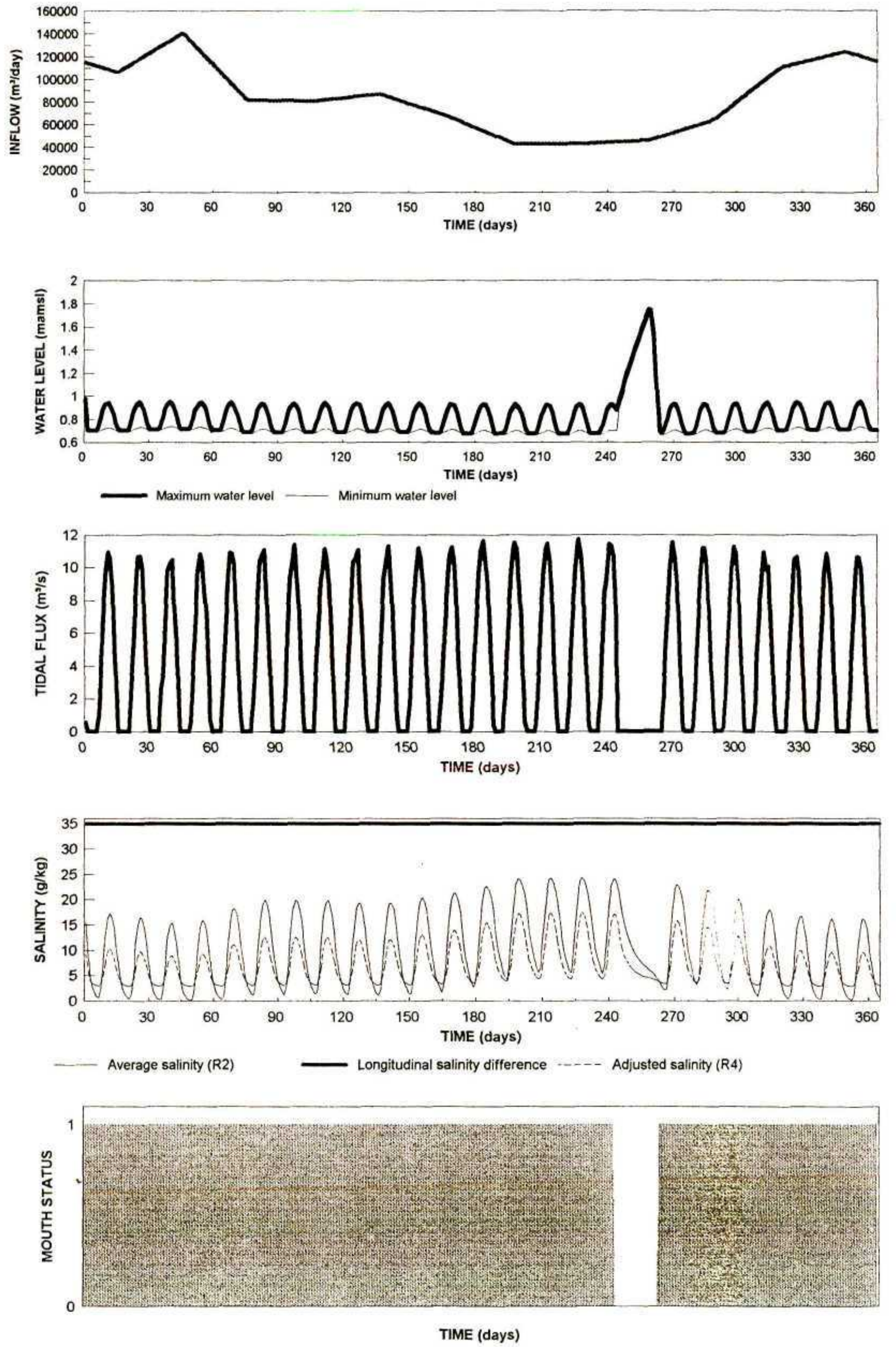

Figure 5 : Flux in key characteristics of the estuary environment in relation to freshwater inflow anticipated under natural conditions, with 50% less baseflow between March and June (Slinger 1995)

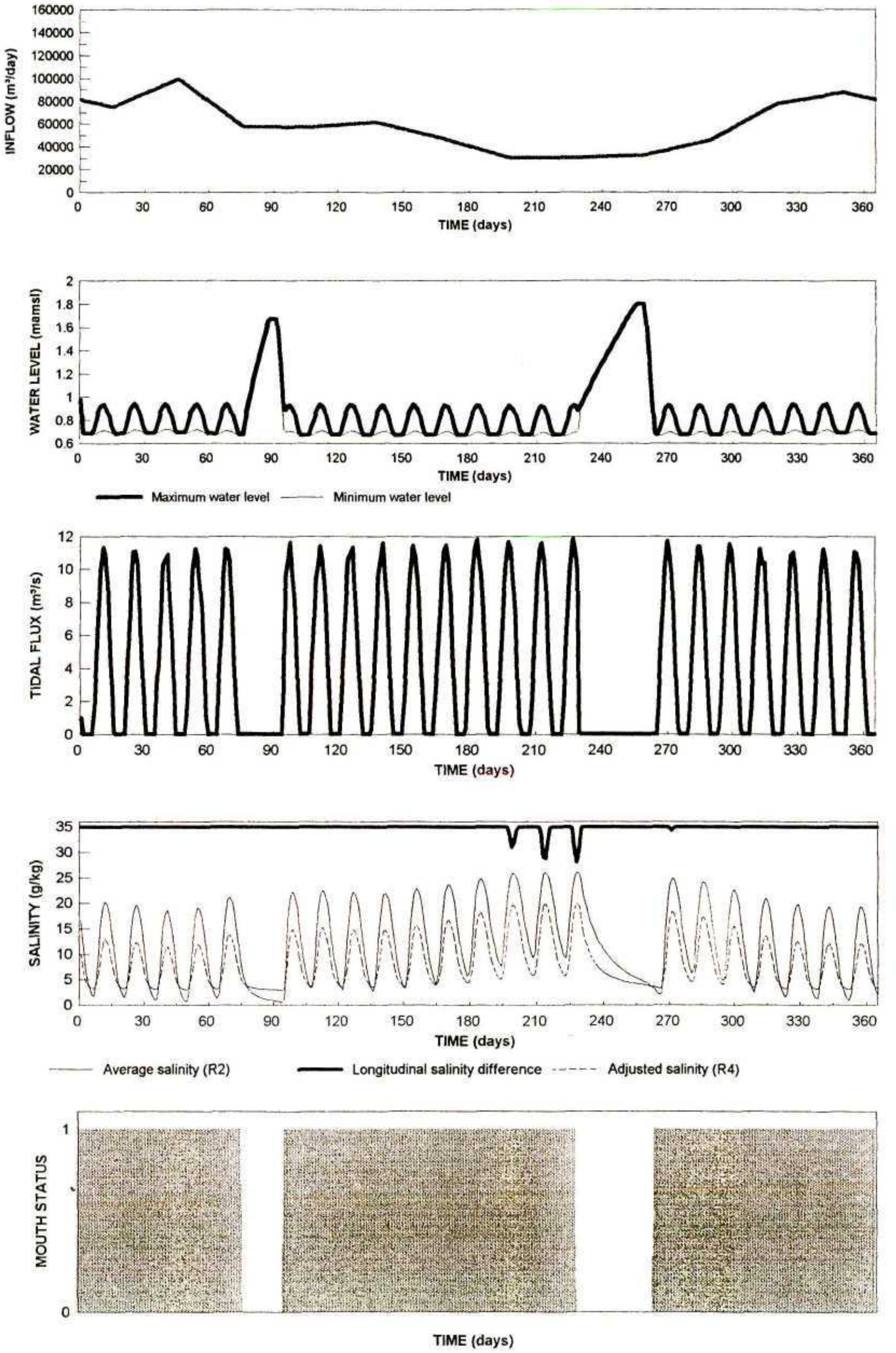

Figure 6 : Flux in key characteristics of the estuary environment in relation to freshwater inflow anticipated under pre-dam conditions, with 50% less baseflow between March and June (Slinger 1995)

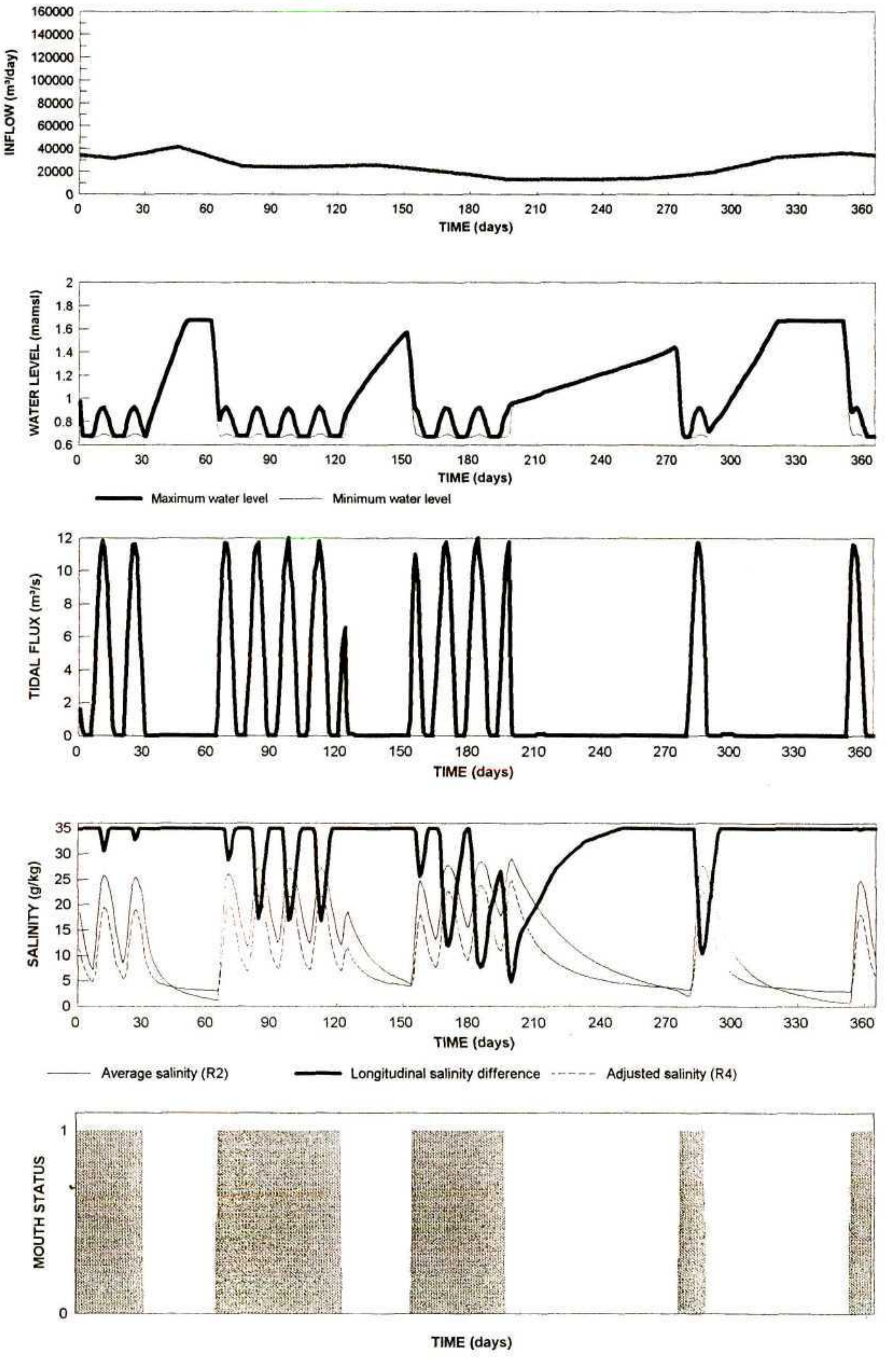

Figure 7 : Flux in key characteristics of the estuary environment in relation to freshwater inflow anticipated under post-dam, maximum baseflow conditions, with 50% less baseflow between March and June (Slinger 1995)

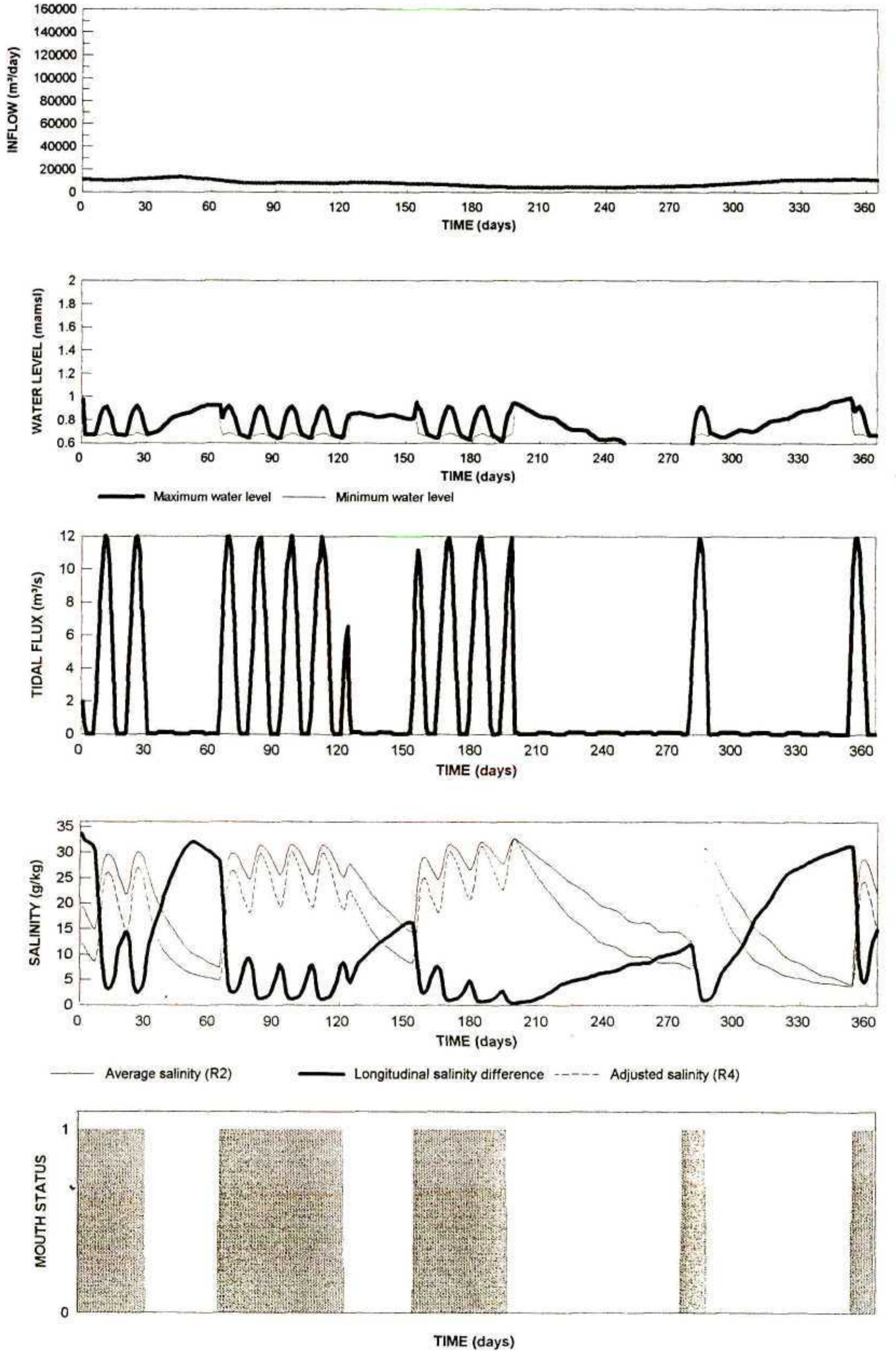

Figure 8 : Flux in key characteristics of the estuary environment in relation to freshwater inflow anticipated under post-dam, minimum baseflow conditions, with 50% less baseflow between March and June (Slinger 1995)

APPENDIX B

Summary of averages and maxima determined in the sensitivity analysis of key assumptions/parameters in the *Upogebia africana* production model.

	Reference	+5%	-5%	+15%	-15%
MAXIMUM SIZE-AGE RELATIONSHIP					
BIOMASS					
Mean	8,255,267.30	8,794,733.74	7,754,012.05	10,131,266.43	6,440,284.91
Maximum	13,691,766.31	14,830,396.30	12,778,991.47	17,459,985.25	10,753,781.55
TOTAL POPULATION					
Mean	34,279,916.32	34,861,162.12	33,601,758.69	36,381,419.20	31,042,870.05
Maximum	47,963,466.05	48,212,467.06	47,681,131.73	56,323,619.05	46,120,184.62
NUMBER OF EGGS					
Mean	1,617,410,799.67	1,831,652,388.58	1,437,219,097.43	2,317,841,087.34	1,029,426,073.55
Maximum	4,863,796,411.31	5,714,666,981.25	4,271,808,863.60	7,316,707,063.28	3,113,652,866.15
SALINITY AND TEMPERATURE MULTIPLIER					
BIOMASS					
Mean	8,248,111.55	8,441,406.99	8,083,621.19	8,995,985.86	7,537,926.35
Maximum	13,673,294.50	14,085,001.36	13,399,063.13	15,183,200.49	12,305,501.12
TOTAL POPULATION					
Mean	34,263,817.36	34,481,376.76	33,923,401.12	35,424,997.91	32,833,302.05
Maximum	48,179,614.31	48,001,640.65	49,737,164.48	52,147,220.05	47,480,750.91
NUMBER OF EGGS					
Mean	1,615,127,056.72	1,701,968,723.52	1,556,237,839.70	1,891,870,017.19	1,366,146,378.38
Maximum	4,857,525,692.74	5,175,583,748.22	4,685,483,513.25	5,993,108,675.36	3,963,568,883.90
SALINITY AND TEMPERATURE MORTALITY					
BIOMASS					
Mean	8,245,608.52	8,253,033.33	8,244,006.40	8,242,162.15	8,253,066.67
Maximum	13,688,309.14	13,710,770.80	13,657,061.65	13,673,132.03	13,665,443.66
TOTAL POPULATION					
Mean	34,262,855.82	34,303,060.38	34,259,677.80	34,258,914.83	34,290,526.74
Maximum	47,912,524.26	48,107,888.44	47,970,788.45	47,915,054.34	47,967,528.70
NUMBER OF EGGS					
Mean	1,613,231,768.27	1,615,339,839.70	1,614,027,310.32	1,612,543,343.58	1,614,905,085.21
Maximum	4,850,321,843.25	4,866,835,619.85	4,851,235,080.04	4,855,950,900.18	4,854,726,440.55
FEMALE CARAPACE SIZE AND FECUNDITY					
BIOMASS					
Mean	8,257,685.61	8,138,630.55	8,350,671.14	7,907,022.69	8,612,144.16
Maximum	13,768,017.60	13,532,963.50	13,696,106.10	13,185,522.78	14,145,662.24
TOTAL POPULATION					
Mean	34,305,370.56	34,423,291.32	34,078,013.26	34,540,514.95	33,744,583.22
Maximum	48,083,174.12	47,887,452.67	47,741,411.77	48,268,253.09	47,602,073.01
NUMBER OF EGGS					
Mean	1,617,355,703.68	1,659,284,044.31	1,565,266,647.81	1,739,076,929.73	1,471,199,170.42
Maximum	5,041,446,134.84	4,991,218,362.19	4,712,024,683.24	5,247,542,753.55	4,429,746,714.60

	Reference	+5%	-5%	+15%	-15%
PROPORTION OF FEMALES WHICH ARE OVIGEROUS					
BIOMASS					
Mean	8,267,098.93	8,133,085.39	8,368,902.56	7,912,881.72	8,612,299.55
Maximum	13,742,923.99	13,546,719.29	13,699,187.03	13,194,649.61	14,160,184.95
TOTAL POPULATION					
Mean	34,346,088.22	34,377,985.95	34,130,243.95	34,547,411.31	33,735,635.40
Maximum	47,951,219.79	47,891,562.03	48,509,464.33	48,247,607.85	47,595,469.20
NUMBER OF EGGS					
Mean	1,617,790,114.69	1,658,199,713.26	1,570,130,127.65	1,740,041,215.81	1,471,841,544.07
Maximum	4,875,418,773.21	4,993,539,133.99	4,712,227,361.34	5,251,247,221.29	4,431,723,849.72
DENSITY DEPENDENT MORTALITY					
BIOMASS					
Mean	8,250,420.31	8,229,630.77	8,281,150.36	8,208,814.32	8,334,452.37
Maximum	13,677,908.14	13,690,991.32	13,699,702.86	13,619,682.67	13,741,823.57
TOTAL POPULATION					
Mean	34,266,914.32	34,191,024.89	34,394,482.92	34,095,333.94	34,572,461.22
Maximum	47,954,866.79	47,655,454.81	48,687,858.74	47,585,294.84	52,553,284.74
NUMBER OF EGGS					
Mean	1,615,290,062.42	1,611,664,507.55	1,621,612,597.27	1,606,797,544.50	1,635,341,638.59
Maximum	4,856,523,382.79	4,849,499,980.27	4,864,590,605.23	4,819,322,631.07	4,968,978,557.91

SELECTION OF INITIAL VALUES AND CONSTANTS				
Initial population (IPOP)	2,500,000.00	7,500,000.00	2,500,000.00	7,500,000.00
Fractional mortality (FMORT)	0.999	0.999	0.997	0.997
BIOMASS				
Mean	7,227,631.10	8,237,934.39	6,133,537.46	5,752,389.99
Maximum	13,534,917.36	13,661,296.01	10,841,137.38	10,493,921.41
TOTAL POPULATION				
Mean	29,877,080.11	34,245,677.47	36,637,792.46	34,068,166.50
Maximum	47,985,379.76	47,962,181.68	58,784,464.43	58,760,569.91
NUMBER OF EGGS				
Mean	1,401,875,537.92	1,611,689,566.61	1,009,206,607.67	944,332,503.79
Maximum	4,812,528,181.60	4,853,685,595.87	3,348,765,233.38	3,330,083,394.99

APPENDIX C

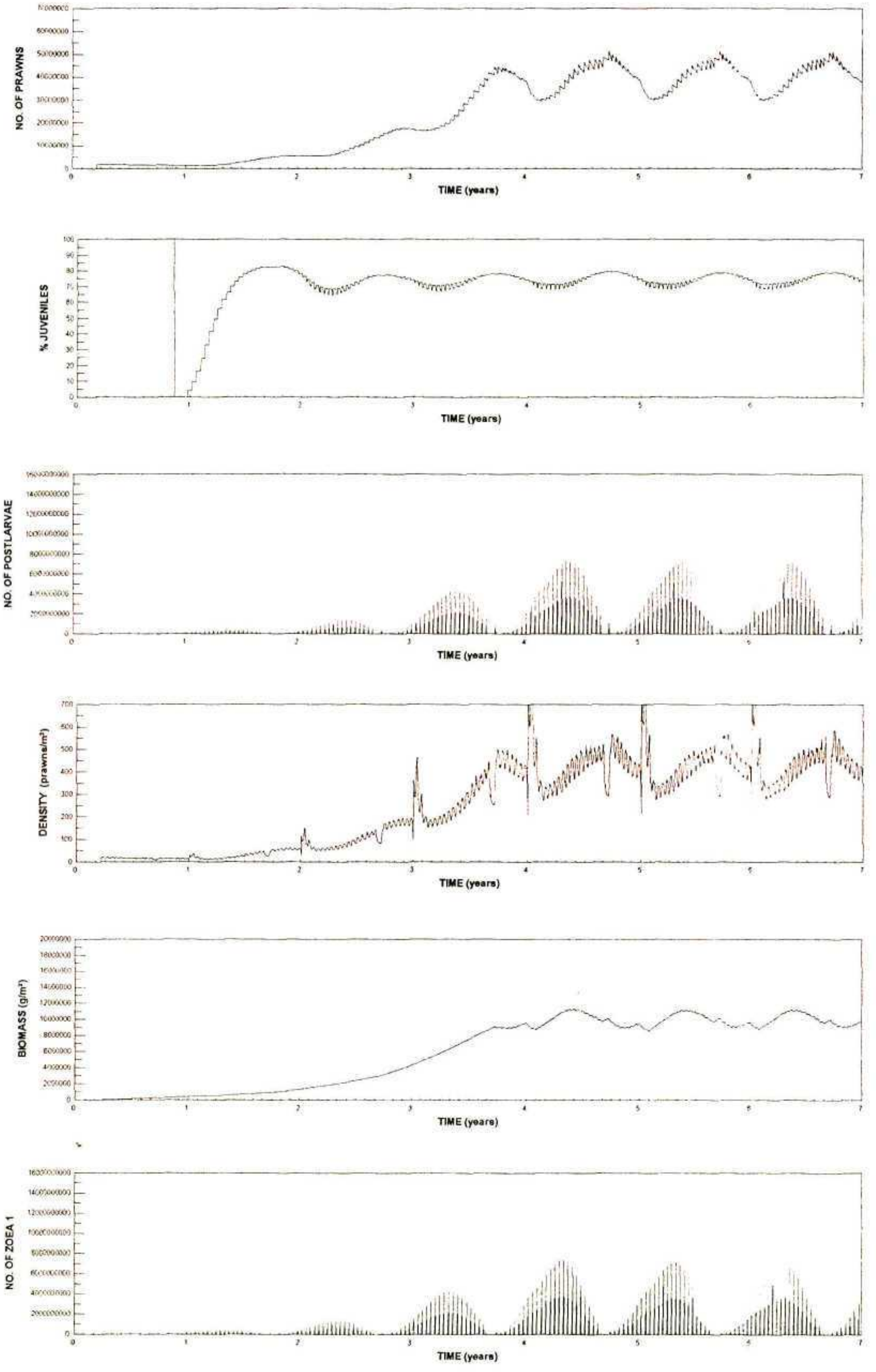

Figure 1 : Results of the *Upogebia* model run to steady state for natural conditions, with a 1:50 year flood in late summer

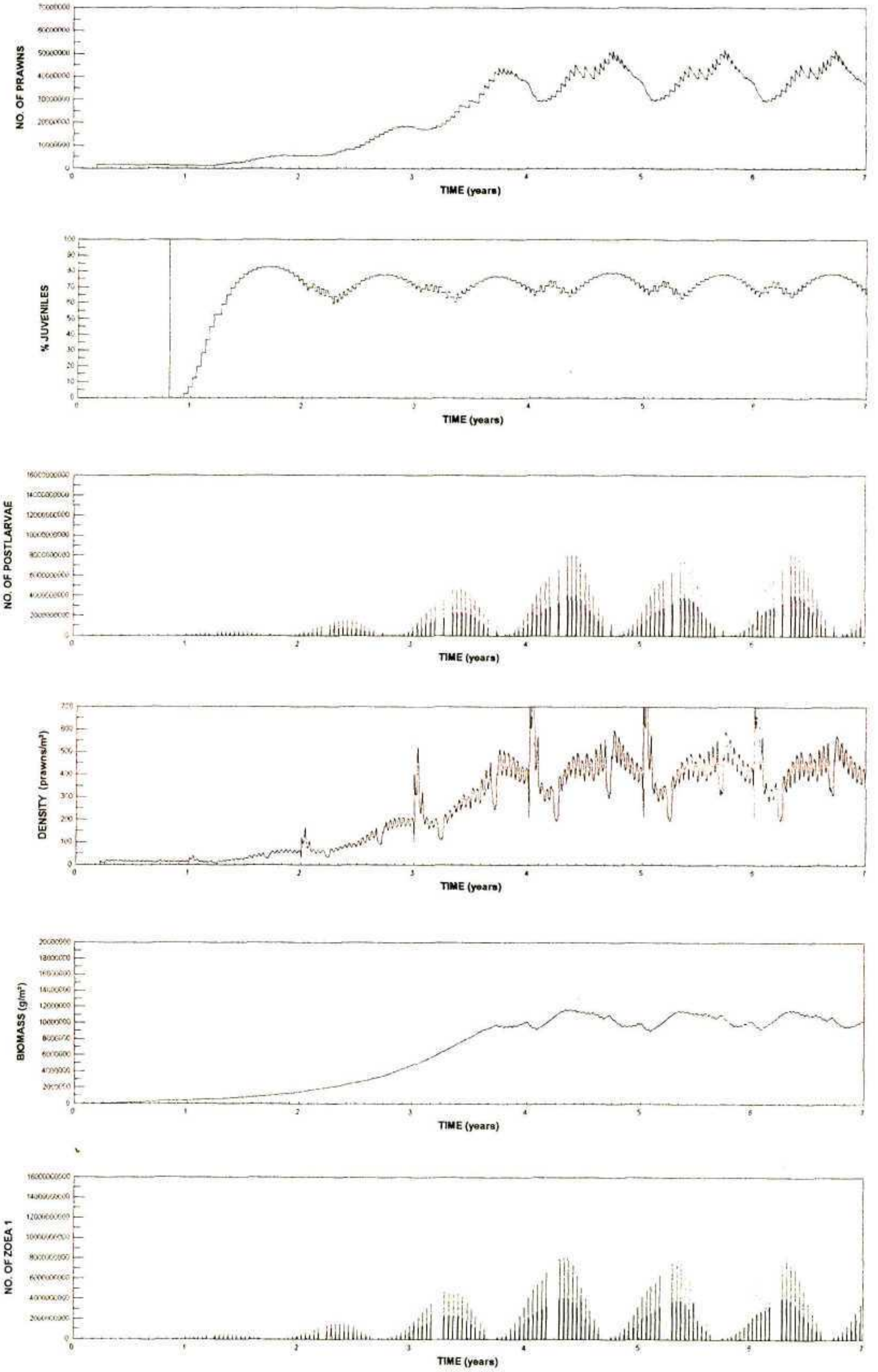

Figure 2 : Results of the *Upogebia* model run to steady state for under pre-dam conditions, with a 1:50 year flood in late summer

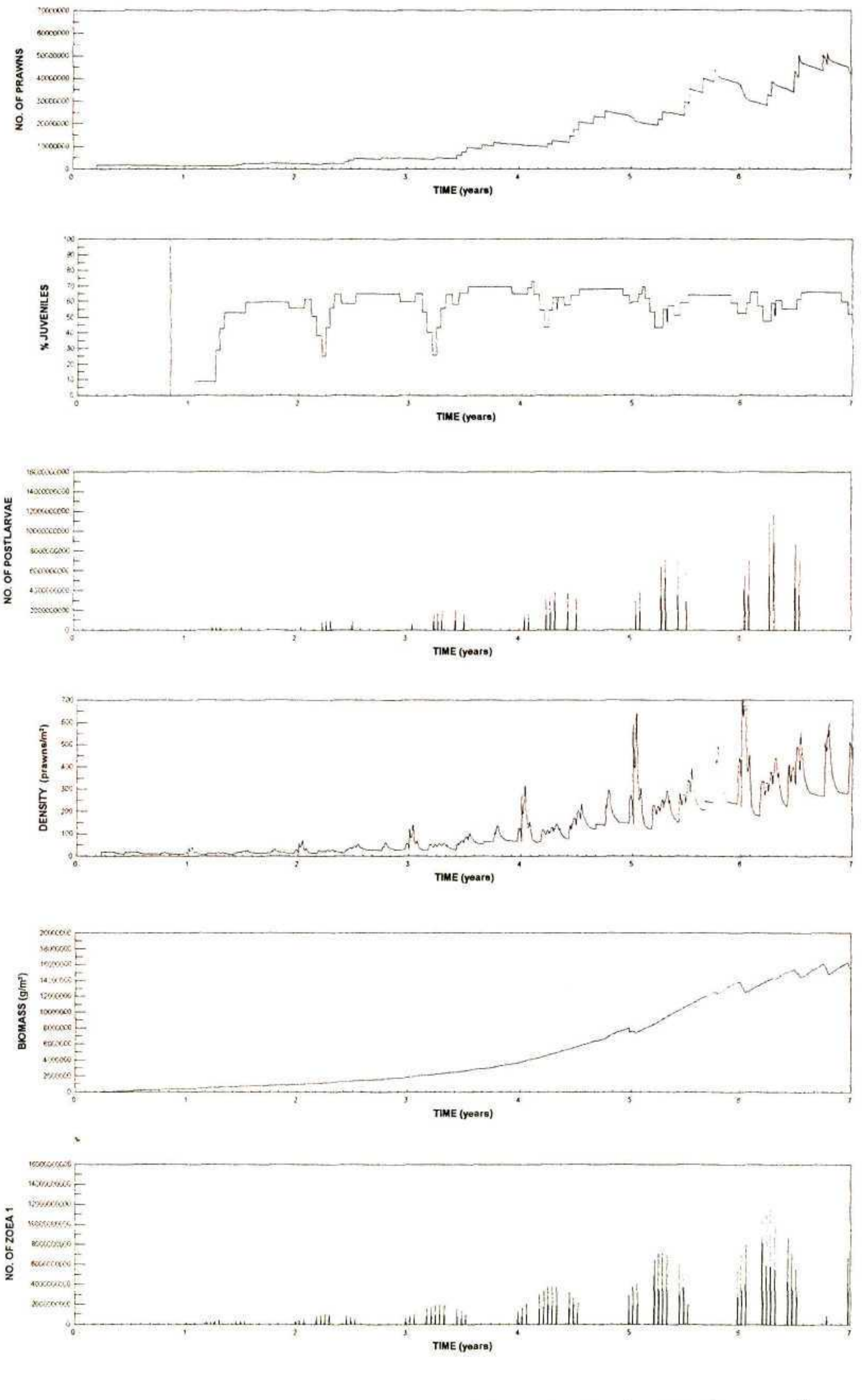

Figure 3 : Results of the *Upogebia* model run to steady state for post-dam, maximum baseflow conditions, with a 1:50 year flood in late summer

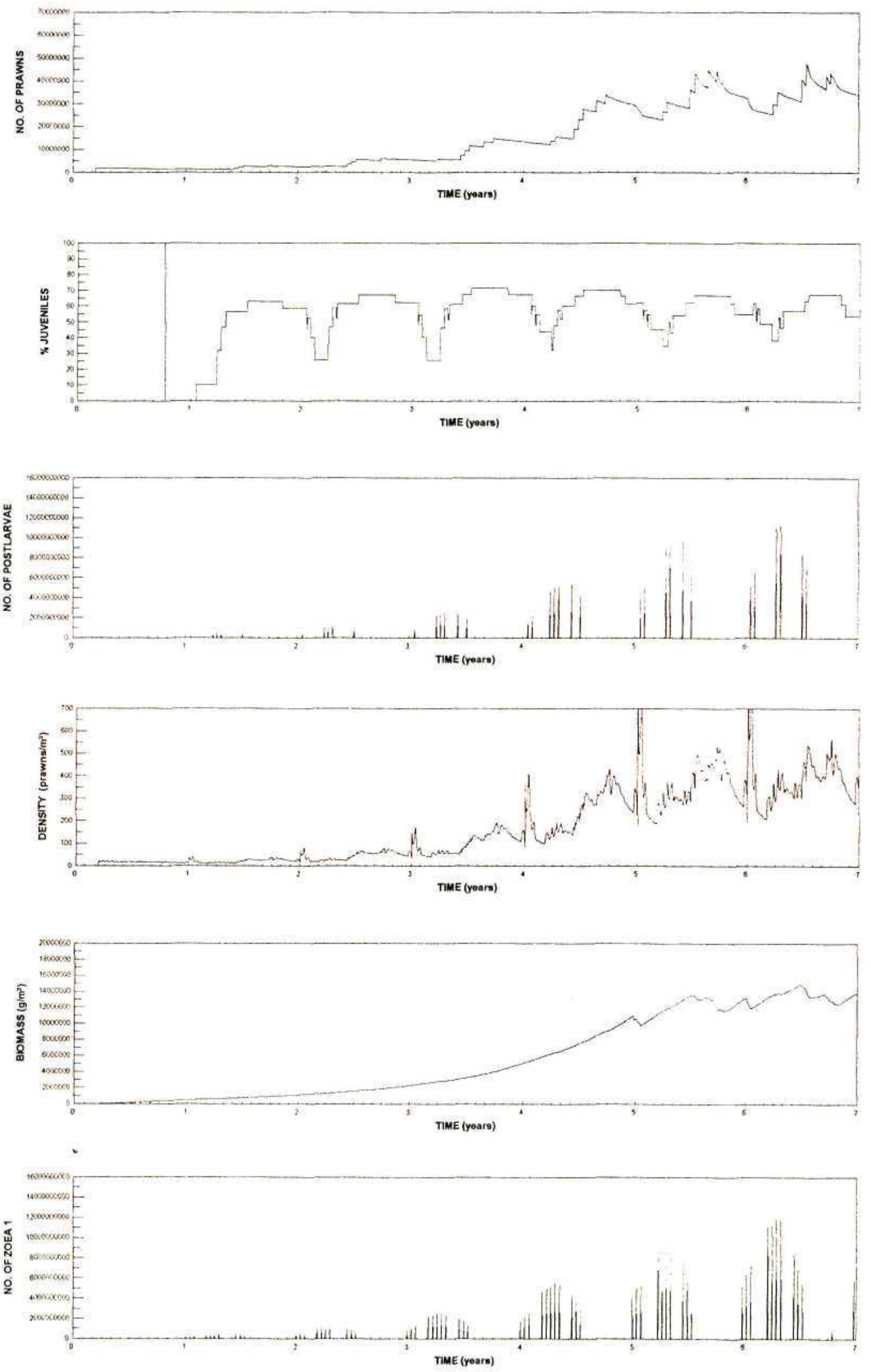

Figure 4 : Results of the *Upogebia* model run to steady state for post-dam, minimum baseflow conditions, with a 1:50 year flood in late summer

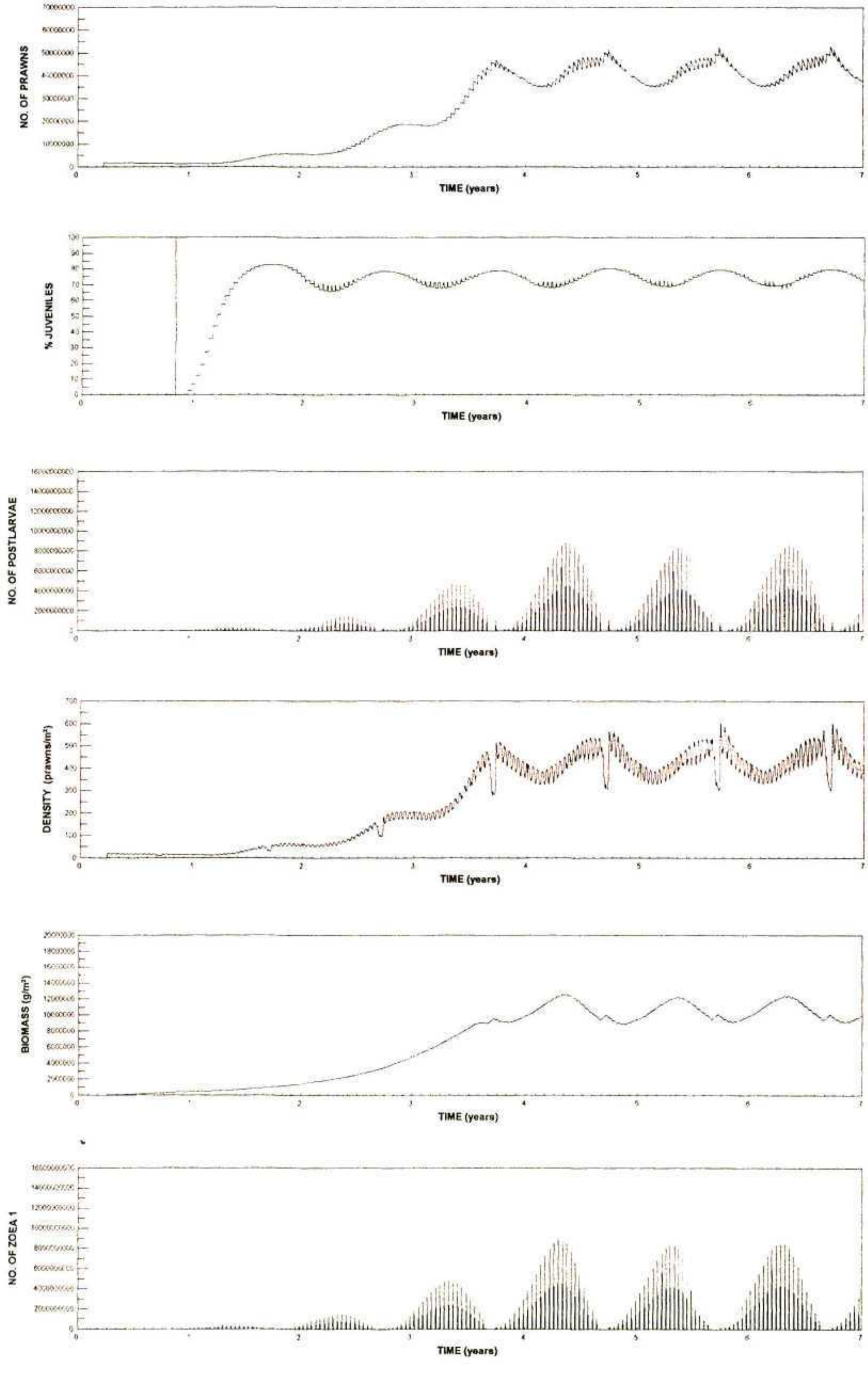

Figure 5 : Results of the *Upogebia* model run to steady state for natural conditions, with 50% less baseflow between March and June

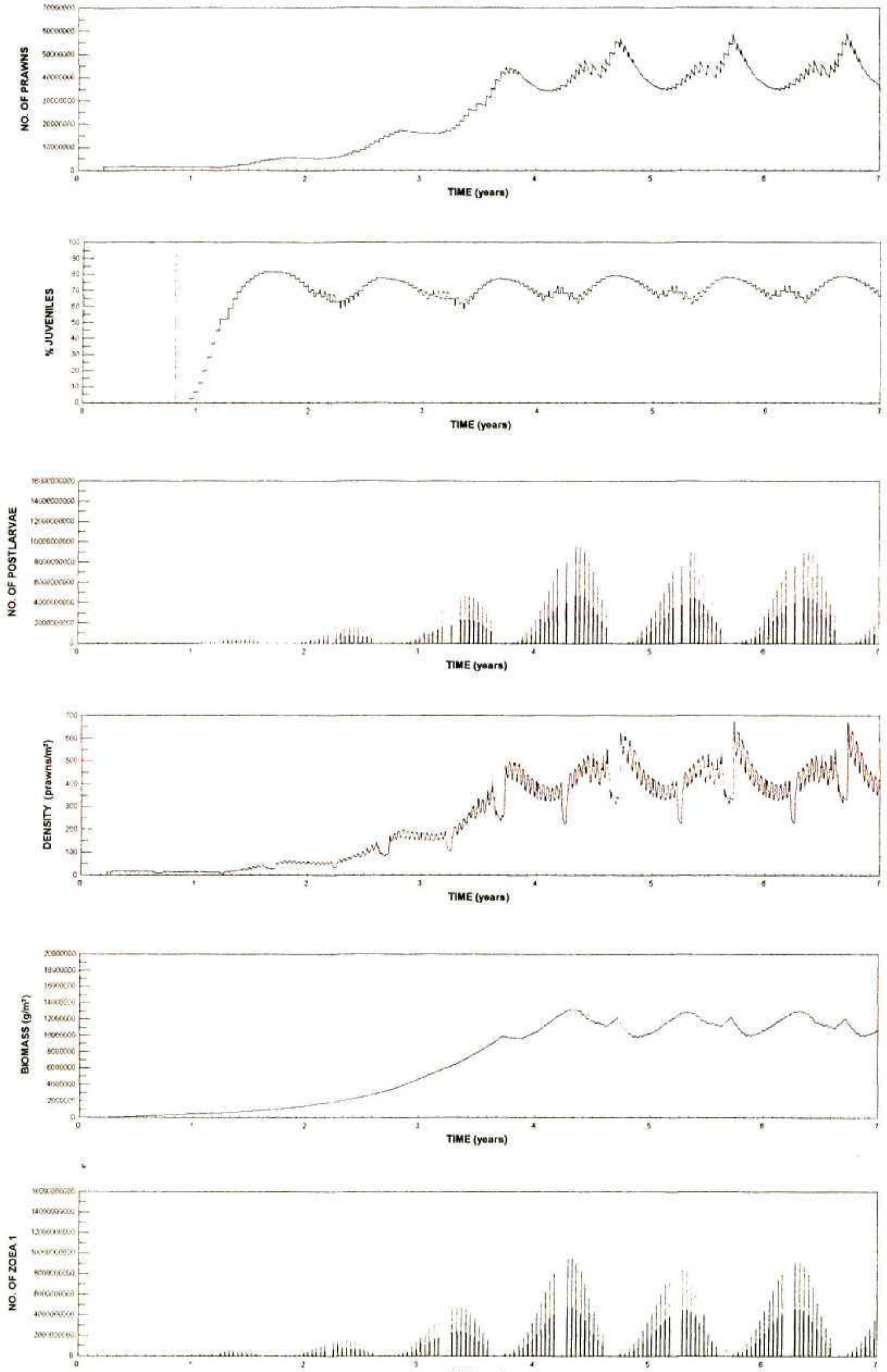

Figure 6 : Results of the *Upogebia* model run to steady state for pre-dam conditions, with 50% less baseflow between March and June

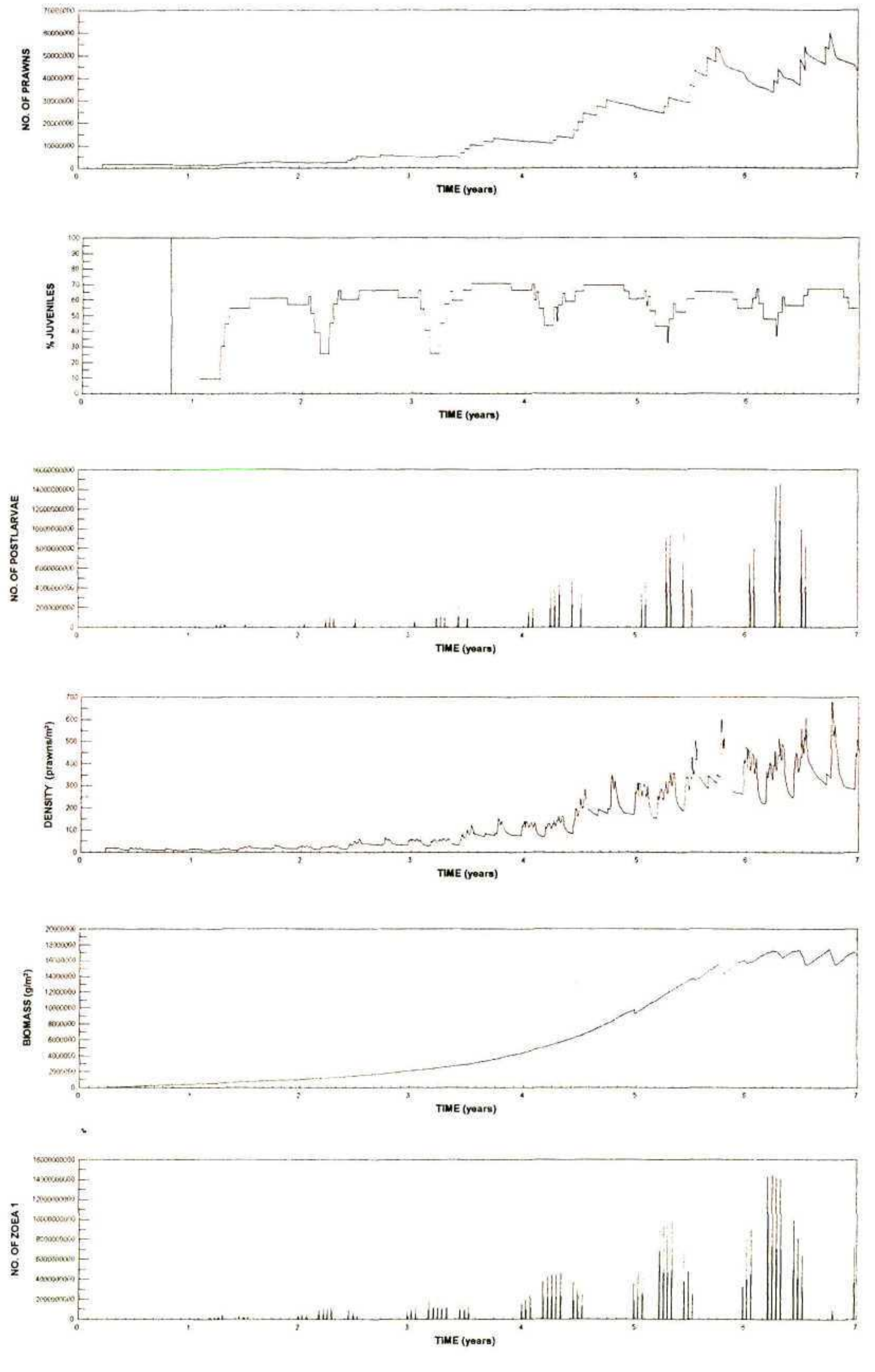

Figure 7 : Results of the *Upogebia* model run to steady state for post-dam, maximum baseflow conditions, with 50% less baseflow between March and June

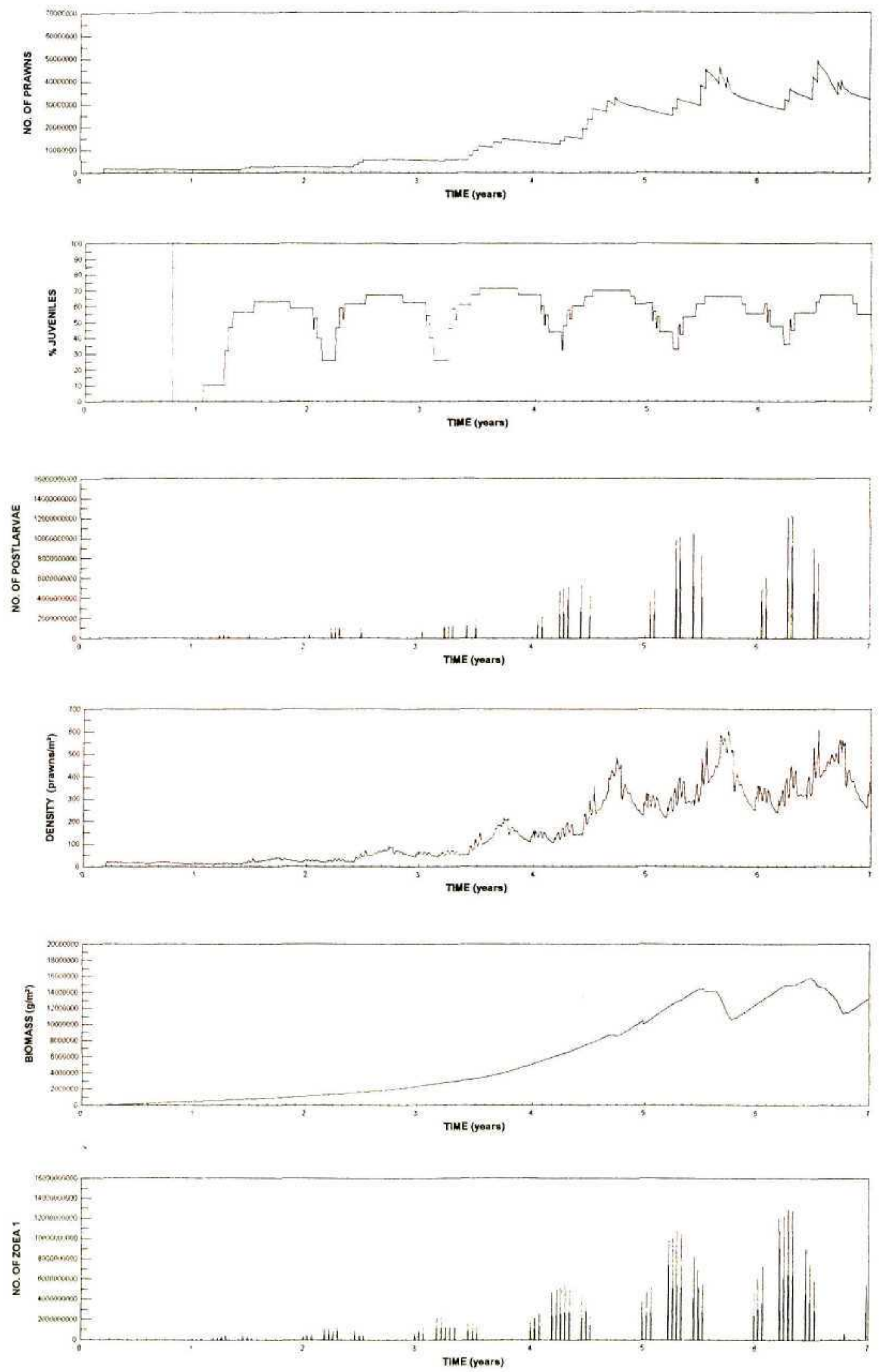

Figure 8 : Results of the *Upogebia* model run to steady state for post-dam, minimum baseflow conditions, with 50% less baseflow between March and June