# Measuring the contribution of ecological composition and functional services of estuaries ecosystems to the dynamics of Kwazulu-Natal coast fisheries

by

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# Declaration

I, Jacobus Gert Crafford, declare that this thesis, which I hereby submit for the degree PhD Economics at the University of Pretoria, is my own work and has not previously been submitted by me for a degree at this or any other tertiary institution.

Parts of the thesis have been submitted for publication in Journals.

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## Abstract

The Millennium Ecosystem Assessment (MEA) defines ecosystem services (ES) to be the direct and the indirect contributions of ecosystems to human well-being and emphasized that regulating ES are amongst the least understood but potentially most valuable services offered by ecosystems. On the one hand, this lack of understanding of regulating ES has been a major reason for the overexploitation and degradation of ecosystems but at the same time causing unnecessarily risk-averse environmental policy leading to delays in economic development interventions.

The value of regulating ES is best determined through an economic-ecological production function approach, which derives the value of regulating services as intermediate inputs into the production of final economic goods and services. This study applies the ES framework of the MEA and uses the ecosystems' concepts of composition, structure, and function to formulate and estimate economic-ecological production functions. Regulating services analysed here included various habitat services, salinity regulation and nutrient cycling. The objective of the research was to find empirical evidence of the significant effects of these services on fish species diversity and fish biomass or stock.

The research is based on a case study, and uses existing ecological and economic knowledge and data sourced from existing scientific databases and studies, to develop and demonstrate empirical production functions that measure relationships between ecological infrastructure and the economy in the KwaZulu-Natal (KZN) fisheries along the east coast of South Africa.

The study was implemented in two phases. The first phase used cross section data and a static model to measure relationships between fish production and compositional and functional elements of both estuarine and marine ecosystems and allowed for estimation of accounting prices of the KZN estuarine and marine ecosystems' attributes. A system of ecological production functions was estimated to measure the effects of estuarine ecosystem compositional and structural elements on fish production using the SURE regression analysis method with highly significant statistical performance and estimated parameter effects consistent with scientific knowledge. The results provided compelling evidence of the importance of estuarine composition and structure on fish species diversity and fish biomass production.

The second phase extended a bio-economic fishery model to establish an explicit link between coastal ecosystems' ecological composition (biodiversity) and functional (nutrient supply) attributes and the dynamics and productivity of KZN coastal fisheries. Results confirmed the importance and strong contribution of the tested ecological attributes. In-sample simulation indicates that current fishing efforts and harvest rates are sustainable, but are sensitive to changes in nutrient influx and rainfall. This confirms the need to modify conventional fisheries models to include environmental variables as additional predictors of fish stocks in addition to historical catch records and catch effort for short-term management and control of fishing efforts and permits. This study provided strong empirical evidence for the linkage between nutrient levels and productivity of coastal fisheries and enabled investigating runoff and rainfall related climate change effects on the KZN fisheries.

The research results provided strong empirical evidence that ecosystems play a significant role in economic production. In the case study, the regulating services relating to species diversity, extent and type of habitat, salinity and nutrient cycling all displayed significant effects on fish production, particularly for their impacts on

commercial line-fishing and recreational angling along the KZN coast. Various coastal developmental and global change hazards may put the functioning of these ecosystems at risk and this research demonstrated how these risks may be evaluated and prudently managed.

**Keywords**: biodiversity, coastal fisheries, ecosystem services, estuaries, regulating services valuation, production functions, resource economics

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# Abbreviations

| | |
|---|---|
| 2SLS | Two-Stage Least Squares |
| AC | Agulhas Current |
| amsl | above mean sea level |
| AR | Annual Runoff |
| B&H | Begg and Harrison |
| CBA | Cost-Benefit Analysis |
| CE | Coastal Ecosystem |
| CEBC | Centre for Evidence-based Ecology |
| CV | Coefficient of Variation |
| CVM | Contingent Valuation Method |
| DAFF | Department of Agriculture, Forestry and Fishing |
| DWS | Department of Water and Sanitation |
| EPA | Environmental Protection Agency (of the USA) |
| ES | Ecosystem Service |
| GDP | Gross Domestic Product |
| GIS | Geographic Information System |
| Ha | Hectares |
| HVM | Hedonic Valuation Method |
| KZN | KwaZulu-Natal |
| m | Meter |
| MEA | Millennium Ecosystem Assessment |

| | |
|---|---|
| mS/cm | Millisiemens / centimetre |
| MSY | Maximum Sustainable Yield |
| Mt | Mega tonne |
| OLS | Ordinary Least Squares |
| SANBI | South African National Biodiversity Institute |
| SURE | Seemingly Unrelated Regression Equations |
| TEEB | The Economics of Ecosystems and Biodiversity |
| TEV | Total Economic Value |
| USA | United States of America |

## Schedule of Symbols

| Symbol | Definition | Data Source* |
|---|---|---|
| $H_t$ | Fish harvest, measured in tons | DAFF |
| $E_i$ | Economic effort (inputs) of a commercial fishery measured as number of fishing crew hours per period | DAFF |
| W | Unit cost of effort, measured in Rand / ton | n/a |
| $p_t$ | Price of fish, measured in Rand / ton | n/a |
| C* | Average cost of fishing measured in Rands / harvest | n/a |
| K | Carrying capacity | n/a |
| $X_t$ | Stock of fish in period t, measured in tons | n/a |
| Q | A 'catchability' coefficient | n/a |
| R | Intrinsic growth rate of the fish stock | n/a |
| $C_t$ | Catch per unit effort in period t, measured in tons/unit effort | DAFF |
| A | Estuaries ES attributes effect on carrying capacity (K) | n/a |
| $S_i$ | Ecosystem attributes measuring various biodiversity attributes, further defined by the symbols preceded by "S-" | n/a |
| $S\text{-}N_t$ | An indicator of nutrient load in the fish production system | Harrison |
| $S\text{-}AR_t$ | Denotes mean annual runoff, measured in million cubic meters per year. It is an indicator of nutrient input. | DWS |
| $S\text{-}TYPE_t$ | An index that refers to the classification of estuary i as defined and measured by the Whitfield physical classification of estuaries (Table 3) | Harrison |
| $S\text{-}HAB_t$ | A vector of estuarine habitat types | Harrison |
| $S\text{-}SAL_t$ | A measure of salinity and is an indicator of the ratio of seawater to fresh water in estuary *I*, measured by conductivity in milli-Siemens per centimetre (mS/cm) | Harrison |
| $S\text{-}SPECIES_t$ | An indicator of species abundance or diversity in the estuarine system, measured as the number of fish species counted per estuary during trawl samples | Harrison |
| $S\text{-}BIOMF_i$ | Measures the total weight of fish biomass caught in grams per trawl sample in estuary *i* | Harrison |
| $S\text{-}SHRLN_i$ | An indicator of ecosystem structure and habitat, measured by the length of estuary *i* shoreline in meters. | Begg |
| $S\text{-}NUTRI_i$ | An index of the nutrient capacity of the estuarine system calculated by dividing the catchment area (in $km^2$) by the volume of the estuary water body (in $m^3$) | Begg |
| $S\text{-}TEC_t$ | An indicator of the density or biomass of fish eggs as sampled by Dr Allan Connell; measured by the total number of fish eggs sampled | Connell |
| $S\text{-}SEC_t$ | An indicator of species abundance or diversity of fish eggs as sampled by Dr Allan Connell; measured by the number fish egg species counted | Connell |

*Refer to section 3.7

# 1. Chapter 1: Introduction

## 1.1 Motivation and problem statement

The fact that most environmental valuation studies had focussed primarily on the direct use values of the environment, and placed comparatively little effort into understanding the indirect linkages between ecological functioning, ecosystem services and the production and consumption of economic goods and services has been cited by many authors to be a major weakness (MEA 2005, Perrings, 2006, Barbier *et. al.* 2009, TEEB 2010, Simonit and Perrings 2011). Such indirect linkages between the environment and the economic system are commonly referred to as regulating ecosystem services. The Millennium Ecosystem Assessment (MEA) identified regulating services as amongst the least understood but potentially most valuable services offered by ecosystems (MEA, 2005). These services enable ecosystems to continue to produce other direct benefits to humans. Such services may also be interpreted as providing 'insurance' value as it allows the system to continue to function over a range of conditions (e.g. stresses or shocks, often of anthropogenic origin) (Loreau *et.al.* 2002, Simonit and Perrings 2011).

The lack of understanding of regulating services is considered the main reason for overexploitation and degradation of ecosystems assets and the ecosystem services they provide to humans (MEA 2005, 2007, TEEB 2010, WAVES 2013). Dasgupta (2008) attributed this exclusion to a historical propensity by mainstream development economists to judge ecosystems as luxury goods which would improve as wealth improves; and to the low visibility of continuous ecosystem degradation. This neglect has not only directly contributed to overexploitation and degradation of ecosystems, especially in developing countries, but has, arguably, also led to often extreme precautionary and risk-averse environmental policy in some countries (Sunstein 2003, Pindyck 2006).

In order to address these problems, the MEA (2005) and subsequently The Economics of Ecosystems and Biodiversity – TEEB (2010) have introduced a new way of thinking about the value of indirect services of ecosystems (regulating and supporting) underlying the direct benefits derived by human society from ecosystems. The MEA and subsequently TEEB define ecosystem services as the direct and indirect contributions of ecosystems to human well-being distinguishing between four types of ecosystem services: provisioning, cultural, regulating and supporting services. The regulating services are the services that ecosystems provide by acting as regulators, and are closely associated with the supporting services, which underpin almost all other services through its function of providing living spaces for humans, plants and animals. Regulating services thus play an indirect role in the economy, and mitigate environmental risks.

There is consensus among the professional community of resource economists that the production function approach is best suited as a valuation method for intermediate ecosystem services. Such approach employs what is known as ecological-economic production functions to apply knowledge of ecosystem functioning and processes to derive the marginal value of supporting and regulating ecosystem services as intermediate inputs into goods and services that are produced and consumed by economic agents (Loreau *et.al.* 2002; MEA 2005; Perrings 2006; Simonit, Perrings 2011).

Ecological-economic production functions require understanding of the attributes of ecosystems, including the concepts of ecological infrastructure and biodiversity. The South African National Biodiversity Institute (SANBI, 2014) describes ecological infrastructure as a network of natural assets "*that conserve ecosystem values and functions and provide associated benefits to society*". A useful characterization of biodiversity is provided by Noss (1990) who describes biodiversity as the composition, structure, and function of an ecosystem: "…c*omposition has to do with the identity and variety of elements in a collection, and includes species lists and measures of species diversity and genetic diversity. Structure is the physical organization or pattern of a system, from habitat complexity as measured within communities to the pattern of patches and other elements at a landscape scale. Function involves ecological and evolutionary processes, including gene flow, disturbances, and nutrient cycling.*" Thus the attributes of ecosystems can be measured through a range of indicators that describe their components, their structure and their functional processes.

This study applies the above ecosystem services framework and uses the concepts composition, structure, and function of an ecosystem to formulate and estimate production functions that link ecological infrastructure and biodiversity to economic activity. This requires a case study of a suitably bounded system, where empirical data may be collected through some means. The practical challenge to doing this lies in the application of econometric analysis to appropriate time series or cross section data for specification and estimation of parameters of ecological-economic production functions characterizing the system under study. This problem has multiple dimensions. Firstly, ecological science specifies and generates information on the compositional, structural, and functional attributes of an ecosystem, and their interrelationships. Secondly, appropriate economic data compatible with available ecological measures of compositional, structural, and functional attributes are not always available.

This research attempts to approach the above challenging complexities by using a case study where both marine and estuarine ecosystems are assessed, and where fish production is used as the dependent variable measuring the economic output of the system, while the independent explanatory variables combine various indicators of composition, structure, and function of an ecosystem with conventional economic input parameters.

Thus, using existing scientific knowledge and data sourced from available databases and studies, this research develops and empirical estimates ecological production functions to measure such relationships in the Kwazulu-Natal (KZN) fisheries along the east coast of South Africa as a case study. This is demonstrated for marine and estuarine ecosystems separately. The study accordingly contributes to the needed effort to develop and apply improved methodologies to support the valuation of regulating and in some cases, supporting ecosystem services.

### 1.2 Objectives of the study

The central challenge addressed in this thesis is to explicitly and empirically link ecological and economic systems, and thus demonstrates and measures the role, importance and management requirements of regulating ecosystem services. This is a challenge because it requires integration of data that measures ecological attributes and data that measures economic attributes into a single (marginal) analysis framework.

The main objective of this research is therefore to apply an ecosystem services framework for empirical specification and measurement of the complex linkages between ecological infrastructure and biodiversity and the economy. This is to be achieved through specification of an ecological-economic production function to enable valuation of the marginal contributions of various attributes of the marine and estuarine ecosystems to productivity of the fisheries of KZN coast in SA.

The specific objectives of the study are to:

- Extend bio-economic fishery models to develop and apply an ecological-economic production function framework to establish an explicit link between coastal ecosystems' services and productivity of coastal (estuarine and marine) fisheries
- Estimate empirically parameters of the developed ecological-economic production functions linking compositional, functional and structural attributes of coastal ecosystems to fishery production
- Provide information on the marginal value and importance of regulating coastal and marine ecosystem services for improved marine and coastal policy and management

## 1.3 Hypothesis

The main hypothesis of this work is that ecosystems play a fundamental role in economic production. The concepts of biodiversity, regulating services and ecological infrastructure provide us with insights into how to analyse ecosystems in terms of its composition, structure and functionality. Chapter Sections 4.3 and 5.3 develop detailed systems' hypotheses given the modelled system and data available for specification of its parameters.

## 1.4 Approach and methods

This research extends bio-economic models to develop ecological-economic production functions measuring the relationship between regulating ecosystem services and economic output. Central to all production function methodology analyses is adequate measurement of the attributes of the ecological and economic systems under study and application of the appropriate econometric estimation procedures. Ecological attributes to be specified include biodiversity-specific measures to analyse the relationship between habitat extent, habitat quality, ecological processes and biodiversity. The production functions' response and determining variables were specified through inputs provided by local ecological experts. Empirical evidence on the relevant response and determining variables of the specified production functions were gathered using best practices of evidence-based ecology.

The study is implemented in two phases. In the first phase, cross section data and a static model were applied to measure the relationship between fish production from recreational angling and the compositional and functional elements of both estuarine and marine ecosystems. In the second phase, time-series data were fitted to an adapted dynamic bio-economic fishery model to measure the marginal contribution of regulating and supporting services of the coastal ecosystems of KZN to the dynamics and productivity of its coastal fisheries.

## 1.5 Structure of the thesis

The next chapter presents survey of relevant literature which traces the history of environmental and resource economics valuation and the development of the ecosystems services concept. It focuses in particular on the problem of regulating (and supporting) services valuation. It concludes, using literature as evidence, that a production function valuation approach is most suitable for valuing regulating (and supporting) ecosystem services. Chapter 3 describes the case study area chosen to be the estuarine and marine ecosystems of the KZN coastline of South Africa and presents the methodology followed to develop the ecological-economic-production functions. Chapters 4 and 5 present, respectively, models, data used and empirical specification and analyses results and implications of static and dynamic production functions developed and estimated using cross section and time series data sources. Chapter 6 concludes and distils implications of the study for policy and management of coastal and marine ecosystems and future research.

# 2. Chapter 2: Literature Review

## 2.1 Introduction

The purpose of this chapter is to provide an overview of the development of resource economic valuation techniques in the context of the concept of ecosystem services and critically assess knowledge gaps in valuing regulating services.

There is a growing literature that attempts to internalise ecosystems' attributes into development planning and policy analysis, through estimating and integrating shadow prices of ecosystem services and the trade-offs involved between elements in bundles of services impacted by development decisions. In spite of these efforts, the value of ecosystems' assets and their services remain largely missing, and even when quantified, underestimated (Adamowicz 2004).

This is attributed firstly to our deficient understanding of the nature of the complex dynamics of the interactions between ecosystems' functioning and human well-being (Perrings 2006). Gaps in current scientific knowledge of the interdependence between the coupled socio-ecological systems translate into misinformed decision making and adoption of wrong policies and actions that fundamentally result in unsustainable use of these natural assets and weak willingness to conserve them. Several factors that characterize the complex dynamics of socio-ecological interdependence are the cause of this.

Among the main reasons is the fact that human society recognizes only the value of a subset of services that are directly used as final products for consumption, production or recreation. On the other hand, the role and value of other more fundamental services that are not directly used as final products, but form crucial supporting and regulating roles underlying the ecosystem functionality, are not recognized nor well understood.

In spite of their crucial role as the basis of all other provisions of nature, the literature on valuing such intermediate services is sparse, leaving an important gap in our knowledge of sustainable management of ecosystems for human well-being. Perrings (2006) described several typical weaknesses in environmental economic study approaches. One of most significant weaknesses was that most environmental valuation studies had focussed primarily on the direct use values of the environment, and put comparatively little effort into understanding the indirect linkages between ecological functioning, ecosystem services and the production and consumption of other economic goods and services.

There is thus a dearth of literature on the valuation of these sets of services (Barbier *et. al.* 2009), especially in dealing with the uncertainty over the supply of ecosystem services (Pindyck 2006).

In response to these weaknesses, the MEA (2005 and 2007) introduced a radical new approach (or framework) to the analysis of the interface of the ecology and the economy. So radical is this approach that it has been described by Perrings (2006) as equal in significance (within the field of environmental economics) to the Hartwick rule for reinvestment of Hotelling rents.

### 2.2 Ecosystem services defined

Central to this approach is the definition of the concept of ecosystem services. The MEA and TEEB define ecosystem services as the direct and indirect contributions of ecosystems to human well-being. They distinguish between four types of ecosystem services: provisioning, cultural, regulating and supporting services.

Provisioning services describe the material or energy outputs from ecosystems. Cultural services include the non-material benefits people obtain from contact with ecosystems. Regulating services are the services that ecosystems provide by acting as regulators. They control and normalise ecosystem functioning and thus insures the benefits supplied by ecosystems (MEA 2005; Barbier *et. al.* 2009, Barbier *et. al.* 2011). Regulating services play an indirect role in the economy, and mitigate environmental risk. Supporting services underpin almost all other services through its function of providing living spaces for humans, plants and animals. Examples of these services, relevant to aquatic ecosystems, may include:

- Climate regulation. Ecosystems influence climate both locally and globally. For example, at a local scale, changes in land cover can affect both temperature and precipitation. Wetland or estuarine ecosystems can act as carbon sinks. At the global scale, ecosystems play an important role in climate by either sequestering or emitting green-house gases.

- Water regulation. The timing and magnitude of runoff, flooding, and aquifer recharge regulate water provisioning in a system.
- Erosion control. Vegetative cover plays an important role in soil retention and the prevention of landslides.
- Water purification and waste treatment. Ecosystems can be a source of impurities in fresh water but also can help to filter out and decompose organic wastes introduced into inland waters and coastal and marine ecosystems.
- Regulation of human diseases. Changes in ecosystems can directly change the abundance of human pathogens, such as cholera, and can alter the abundance of disease vectors, such as mosquitoes.
- Biological control. Ecosystem changes affect the prevalence of pests and diseases.
- Pollination. Ecosystem changes affect the distribution, abundance, and effectiveness of pollinators.
- Storm protection. The presence of coastal ecosystems such as mangroves and coral reefs can dramatically reduce the damage caused by hurricanes or large waves.

Supporting services are those that are necessary for the production of all other ecosystem services. They differ from provisioning, regulating, and cultural services in that their impacts on people are either indirect or occur over a very long time, whereas changes in the other categories have relatively direct and short-term impacts on people. Some examples of supporting services are primary production, production of atmospheric oxygen, soil formation and retention, nutrient cycling, water cycling, and provisioning of habitat.

The key benefit of the MEA/TEEB framework is that it allows investigators to systematically unpack a development problem into its ecosystem attributes and the ecosystem services dependent on them.

The MEA did not attempt a comprehensive and systematic 'total' valuation of ecosystem services, because the judgment was that the theory, methods and data sources were insufficiently developed to support a credible effort of that sort (Kinzig *et. al.* 2007). Subsequent to the publication of the MEA, Perrings (2006) identified a number of challenges to the field of environmental economics in the post MEA era. Those challenges relate to the practical implementation of the MEA approach and framework, and includes that economists need to (after Perrings 2006; Kinzig *et. al.* 2007):

1. "*Understand the consequences of ecological change induced by current economic activity*
2. *Understand the distribution of possible outcomes of alternative activities and, where feasible, the probabilities attached to those outcomes*
3. *Develop appropriate mitigating or adaptive policies*."

Another important aspect of the MEA framework of ecosystem services is that it implicitly links biodiversity to the economy and to human well-being. The term ecosystem has been defined as a natural unit consisting of all plants, animals and micro-organisms (biotic components) in an area functioning together with all of the non-living physical (abiotic) factors of the environment (Christopherson 1996). Biodiversity is the living component of the ecosystem. Accordingly, the ecosystem is interpreted in the MEA to represent a portfolio of abiotic and biotic assets that produce a specific set of ecosystem services, which are of benefit to human well-being. Biodiversity is described by Noss (1990) more than simply the number of genes, species, ecosystems, or any other group of things in a defined area. Noss (1990) rather favors a characterization of biodiversity that identifies the major components at several levels of organization which includes composition, structure, and function.

The valuation of biodiversity-based services thus requires more than the valuation of the diversity itself, as for instance in the case of ornithological or botanical tourism, or bio-prospecting. In most cases the value of biodiversity is indirect, or embedded in the provisioning or cultural services that are ultimately consumed (Kinzig *et. al.* 2007).

Thus, if we are to understand and enhance the resilience of such coupled systems we need robust models of the linkages between biodiversity and ecosystem services (Loreau *et. al.* 2002; Naeem and Wright 2003; Reich *et. al.* 2004; Hiddink *et. al.* 2008), and between biodiversity change and human well-being (Kinzig *et. al.* 2007).

It is appropriate at this stage to consider the evolution of environmental valuation techniques.

## 2.3 Environmental valuation techniques: a brief history and summary of future challenges

The evolution of environmental goods and services valuation techniques is characterised by a historically increasing demand for precise quantification of the values of these goods and services, and driven by various incidences of environmental disasters (Brown 2000).

One widely used approach employed by economists for decision-making support is the cost-benefit analysis (CBA). This approach was formalized in the United States of America with the purpose of justifying public expenditures on alternative investment options competing public funds such as water, roads, and other public utilities' networks construction projects. In the 1930s and 1940s, questions arose about social accounting and how to treat intangible cost and benefit factors such as saving lives, and dealing with recreation. CBA was formalized in 1958 with the publication of the "Green Book," a document intended to provide federal agencies in the USA with a consistent conceptual framework for doing benefit-cost analysis. The Green Book advised on the treatment of National Income benefits and costs; how to measure them conceptually; how future prices should be treated; the importance of using a discount rate; the proper period of analysis; risk; taxes; and cost allocation procedures for multiple purpose projects. The Green Book was likely grounded in the "new welfare economics" thinking of the 1950s based on the works of Bergson (1938), Hicks (1940), Kaldor (1939) and others.

The Green Book (1958) recognized three types of benefits *market goods, non-market goods and intangibles.* It contained rigorous treatment of key economic concepts and an emphasis on developing measurable indices for valuing goods and services. Consumer and producer surplus as measures of value, and not price, began to be emphasized (drawing on Dupuit, 1844; and Hotelling, 1931). Moreover non-market goods and intangibles were not measured, but rather treated qualitatively as no proven valuation methods for these existed.

In 1947 the US National Parks Service sent a letter to ten expert economists asking how to estimate the value of one or more national parks. Only Harold Hotelling responded, with the concept termed the travel cost method (TCM) (Brown, 2000). Subsequently, by 1962, recreation had become a primary purpose of USA federal water projects because its value could be measured. The TCM relied indirectly on market values using observed behaviour, such as number of trips and their associated costs.

Ridker (1967) pioneered the hedonic valuation method (HVM) to estimate the marginal value of air quality in residential areas by regressing the value of residential properties on their characteristics. The estimated coefficient for each characteristic was considered to measure its marginal value. Thus, by 1970 HVM and TCM were the two indirect methods through which resource economists used prices generated by individual behaviour in the market to indirectly estimate the value of "non-market" goods such as air quality, scenic views and recreation sites.

These methods however, still proved inadequate to address a broader range of environmental goods and services leading to the development of the contingent valuation method (CVM). The CVM was intended to value public goods for which no observed activity could yield an economic value, either direct or indirect. Krutilla (1967) had formulated the rationale for these non-use values in making allocation decisions regarding natural resources. In addition, the work of Ciriacy-Wantrup (1947) had provided a precursor for a number of studies of academic nature to follow. However, for many years thereafter, the CVM method was not regarded as a legitimate valuation technique, based on the so-called Ohio case (McManus, 1994; Brown 2000). This changed with the 1989 Exxon-Valdez oil spill, which occurred off the coast of Alaska damaging natural resources for which the trustees were the State of Alaska and the Federal Government. The level of injury to natural resources appeared to be very substantial. Exxon hired world famous economists to discredit non-market valuation techniques, arguing that they had no professional legitimacy. On the other hand, the lead federal agency (The National Oceanic and Atmospheric Administration - NOAA) appointed a panel comprised of two Nobel prize winners (K. Arrow and R. Solow), Paul Portney (Head of Resources for the Future), and three others to evaluate the legitimacy of non-use valuation methods. The said panel recommended a thorough set of procedures to follow if there was going to be litigation (Arrow *et. al.* 1993).

Environmental and resource economists have since then applied the resulting two broad approaches (stated and revealed preferences methods) to value ecosystem assets and their services. In the stated preference method, economists ask people to place a value on ecological resources. In the observed behaviour (revealed preferences) method, economists study the actual choices of people to infer the value people place on ecological resources (Freeman 2003, Pearce *et. al.* 2005, Dasgupta 2010). Both of these approaches however, have limitations.

One of the limitations is that most valuation studies have focussed on a single dimension of a problem only, and has thus failed to adequately address the complexity inherent in ecosystem processes and functionality (Perrings 2006). For many years, economists circumvented this problem by using the total economic value (TEV) framework (Pearce 2005). Although this framework provided an adequate categorisation of both market and non-market ecosystem services, it failed to address ecosystem complexity. The MEA's concept of ecosystems services (MEA 2005, MEA 2007) significantly improved on the TEV framework through explicitly defining the linkages between ecosystems and human well-being.

Thus, through the MEA, and later the TEEB (2010), the approach to value of ecosystems has evolved to the understanding that the value of ecological assets is the discounted value of the ecosystem services which it produces. And ultimately, the shadow price of ecosystem assets could ideally be estimated through the human capital asset defined as "health" (Dasgupta 2010) or human well-being (MEA 2007), and specifically through valuing the marginal increase in life expectancy. This can be done either through estimating the social cost of a marginal increase in life expectancy, or through estimating the value of the increase in life expectancy.

However, neither method has been particularly successful, for two reasons. First, the value of a statistical life is controversial when making cross-country comparisons, and secondly, both methods require perfect economies, whilst the economies of most countries are highly imperfect (Dasgupta 2010). At a project level, empirical data in support of such methods are also inadequate.

Moreover, not only are ecological assets' shadow prices functions of the stock of all assets, but in addition, they are functions of the degree to which various assets are substitutable for one another, for all time periods current and future (Dasgupta 2010).

As reported in a seminal work by Perrings (2006), a large number of studies have employed the revealed and stated preference approaches to quantify the value of changes in ecosystem services to human societies in terrestrial (Daily *et. al.* 1997, Daily 1997), marine (Duarte 2000, Remoundou *et.al.* 2009, Sumaila *et.al.* 2000, Sumaila 2002, Pendleton 1995) and agro-ecosystems (Björklund *et. al.* 1999). Others made attempts to estimate the value of ecosystem services for the whole world, or large parts of it (Costanza *et. al.* 1997; Bolund and Huhammar 1999; Norberg 1999; Limburg and Folke 1999; Woodward and Wui 2001). Concerns over the reliability of such estimates however have been raised due to various weaknesses both in the valuation framework used and valuation techniques applied (MEA 2007, Dasgupta 2010).

Another limitation of several existing valuation methodologies has been that the people interviewed (stated preference method) or behaviour observed/recorded (revealed preference method) are often not aware of environmental risks (Perrings, 2006; Dasgupta, 2010). People often have little conception of the role of ecosystem assets in the generation of ecosystem services, or of the link between those services and the production of commodities. This is because so many ecosystem services are intermediate inputs into the production of final goods and services (Perrings, 2006). This holds true not only for the supporting and regulating services, but also for some provisioning and cultural services, such as fresh water, genetic resources or aesthetic services.

Understanding ecosystem responses to biodiversity and other ecological production factor change requires new theoretical and experimental work linking for instance nutrient cycling, biodiversity and ecosystem functioning in complex systems at different scales using spatially explicit models (Loreau et. al., 2001).

Perrings (2006) described and analysed these weaknesses based on meta-analysis which found that most studies had focussed on a single dimension only, and ignored the multiple environmental goods and services effects of a shock introduced to a local system. Furthermore, most environmental economic studies had focussed primarily on the direct use values of the environment, and put comparatively little effort into understanding the indirect linkages between ecological functioning, ecosystem services and the production and consumption of marketed goods and services. Ecosystems and the goods and services they provide are, for the largest part, intermediate inputs into goods and services that are produced or consumed by economic agents. As with other intermediate inputs, their value derives from the value of those goods and services (Perrings 2006). This was aggravated by a dependence on stated preference studies (such as CVM) of sample populations who had insufficient knowledge of the role of ecosystem stocks in the generation of environmental goods and services, or of the link between those goods and services and the production of commodities (Perrings 2006).

Another set of concerns related to the way in which valuation studies addressed the problems of risk and uncertainty (Perrings 2006). Since the value of ecosystem stocks is the discounted stream of net benefits they provide, it is sensitive to uncertainty about the environmental and market conditions under which they will be exploited. Ecosystem cost and benefit relationships tend to be highly nonlinear and therefore, damage (or consequence) might be barely noticeable for low levels of hazard (e.g. pollution) but then becomes severe or even catastrophic once some (unknown) threshold is reached. Furthermore, the precise shapes of the relationships are often unknown, which is of particular importance if there is a threshold at which the effect of a hazard becomes extremely severe Pindyck (2006).

Most valuation studies simply sidestepped the problem, whilst others addressed it indirectly through the discount rate. Where uncertainty about future consequences of the use of the environment includes the likelihood of severe and irreversible consequences, this was considered unsatisfactory, and, as social-ecological systems are complex, coupled and adaptive, the capacity to predict the future consequences of current actions are limited at best (Perrings 2006). Closely associated with this was the problem that little effort also went into understanding the value of the role of the environment in either mitigating or exacerbating risks (Perrings 2006). A risk to an asset is defined as the likelihood that a particular hazard may affect (or have consequence or change the state of) an asset (Schütz *et. al.* 2006). The analysis of risk thus requires us to assess not only the mean effect of a change to an ecosystem (the first moment of the distribution of possible outcomes), but also the probability of an outcome (i.e. risk effects) that depends on higher moments of the distribution (e.g. dispersion and spread) (Mäler 2005).

The production function approach is considered to be best suited as a valuation method for intermediate ecosystem services because it is able to address many of these valuation weaknesses (Mäler 1991, Barbier 2000, Barbier 2003, Perrings 2006, Kinzig *et.al.* 2007, Barbier *et.al.* 2009). The next section expands on the concepts and methods of production function analysis.

## 2.4 The production function valuation approach

Production function approach studies quantify values for ecosystem services that contribute at least part of the shadow value of those resources. They apply knowledge of ecosystem functioning and processes to derive the value of supporting and regulating ecosystem services (Mäler 1991, Perrings 2006, Kinzig *et. al.* 2007, Barbier 2000, Barbier 2003, Barbier *et. al.* 2009). They do this through deriving the value of ecosystems and the services they provide as intermediate inputs into goods and services that are produced or consumed by economic agents.

The theoretical foundations for the production function approach were pioneered by Mäler (1991). A number of studies employed the production function model to specify and estimate parameters of the linkage between ecosystem attributes and production of final goods and services. Barbier (2000, 2003, 2007) demonstrated how to value mangrove habitat area as an input into a fisheries production model, both for static and dynamic cases. While this work succeeded in integrating habitat as a component of an ecosystem into a fisheries production model, it did not integrate variables that explain habitat quality.

Few production function studies investigated the value of regulating services. Barbier and co-workers (2009) proposed a damage function approach to deal with the “natural hazard regulation” function as defined in the MEA, but concludes that much work is required to advance knowledge in this field.

Building on Barbier’s early work, Rodwell and co-workers (2003) inserted a habitat quality function into a dynamic fisheries production model for Kenyan off-shore fisheries. Rodwell’s work is significant because it demonstrated the application of a system of production functions, as opposed to a single function. This system of functions consisted of a catch model, a production model, a recruit transfer model and a habitat quality model. However, Rodwell and co-workers failed to find empirical data to confirm the exact relationship between habitat quality and fish biomass or natural mortality rates, but rather applied simulations to implement the system of production functions.

Chopra and Adhikari (2004) investigated the link between ecological parameters and tourism through studying a wetland in the Keoladeo National Park in Northern India. They used empirical data captured between 1984 and 1988 to model a series of production functions that related various attributes of ecological parameters to tourist visitation numbers. The resultant production function analyses showed that tourist visitation elasticity is high with respect to an ecological health indicator, whereas conflictingly, results of a conventional TCM study showed that tourists have inelastic demand for ecosystem services. The authors concluded that the conflicting study outputs resulted from economic valuation typically focussing on short run use values, whereas conservation biologists were more concerned with the underlying long-run conservation value of the system being studied. This is possibly a spurious conclusion, as the TCM was inadequately specified and did not include appropriate attributes of ecological health as independent variables.

Nakhumwa and Hassan (2003) and Yirga and Hassan (2010) integrated a biomass effect component in dynamic soil nutrient depletion models. Jogo and Hassan (2010) estimated ecological production functions for the Limpopo wetland system in South Africa. In all these studies the authors used nutrient inputs and groundwater inputs into biomass production functions which in turn integrated into agricultural production functions. Other examples of production function studies include maintenance of biodiversity and carbon sequestration in tropical forests (Boscolo and Vincent, 2003); nutrient reduction in the Baltic Sea (Gren *et. al.*, 1997); pollination service of tropical forests for coffee production in Costa Rica (Ricketts *et. al.*, 2004); tropical watershed protection services (Kaiser and Roumasset, 2002); groundwater recharge supporting irrigation farming in Nigeria (Acharya and Barbier, 2000); marine reserves acting to enhance the 'insurance value' of protecting commercial fish species in Sicily (Mardle *et. al.*, 2004) and in the northeast cod fishery (Sumaila, 2002); and nutrient enrichment in the Black Sea affecting the balance between invasive and beneficial species (Knowler *et. al.*, 2001). Hassan (2003) used quantitative models of woody land resources in South Africa to estimate their value to rural inhabitants of the Fynbos Biome. The work of Matete and Hassan (2006), Allen and Loomis (2006) and Pattanayak and Butri (2005) also included examples of production function applications. Simonit and Perrings (2011) demonstrated the regulating service value of wetlands in Kenya on fish stocks in Lake Victoria.

### 2.5 Fisheries ecological production function models

Static production models to value how a change in estuarine habitat components and functions affects the market for commercially harvested fish have been studied by Lynne and co-workers (1981), Ellis and Fisher (1987), Freeman (1991), Sathirathai and Barbier (2001) and Barbier (2007). Barbier (2007) estimated a static habitat-fishery model in which harvested volume of fish (catch) of the commercial fishery is assumed to be a function of economic effort inputs and habitat area as a proxy for the supporting estuarine ecosystem attribute. This allowed specification of reduced form first order solution functions of a cost-minimizing fishery to determine equilibrium levels of fish harvest and measure the marginal impact of a change in estuarine system functionality. It was accordingly possible to compute changes in equilibrium harvest and price levels and the corresponding changes in consumer surplus associated with a change in estuarine functionality, for a given demand elasticity.

This specification assumed that the fishery was open access, which implied that any profits in the fishery would attract new entrants until all the profits disappeared, and in equilibrium, the welfare change in coastal wetland was in terms of its impact on consumer surplus only. The standard assumption for an open access fishery is that effort next period will adjust in response to the real profits made in the past period (Bjørndal and Conrad 1987). However, in fish stock production functions, the magnitude of the fish stock is a significant variable, and thus valuing changes in terms of the impacts on current harvest and market outcomes alone is flawed. In order to address this weakness, dynamic models of coastal habitat-fishery linkages, which incorporate the change in estuarine system functionality within a multi-period harvesting model of the fishery, are required (Barbier 2007).

Most attempts to value habitat-fishery linkages via a dynamic model that incorporates stock effects have assumed that the fishery affected by the habitat change is in a long-run equilibrium. Such a model has been applied, for example, in case studies of valuing habitat fishery linkages in Mexico (Barbier and Strand 1998), Thailand (Barbier *et. al.* 2002; Barbier, 2003) and the United States (Swallow 1994). Similar 'equilibrium' dynamic approaches have been used to model other coastal environmental changes, including the impacts of water quality on fisheries in the Chesapeake Bay (Kahn and Kemp 1985; McConnell and Strand 1989) and the effects of mangrove deforestation and shrimp larvae availability on aquaculture in Ecuador (Parks and Bonifaz 1997).

Valuing the change in estuarine ecosystem attributes in terms of its impact on the long-run equilibrium of the fishery raises additional methodological issues. First, the assumption of prevailing steady state conditions is strong, and may not be a realistic representation of harvesting and biological growth conditions in estuarine dependent species. Second, such an approach ignores both the convergence of stock and harvest to the steady state and the short-run dynamics associated with the impacts of the change in coastal habitat on the long-run equilibrium. The literature provide examples of pure fisheries models that assume that the dynamic system is not in equilibrium but is either on the approach to a steady state or is moving away from initial fixed conditions (Bjørndal and Conrad 1987; Homans and Wilen 1997). Barbier (2007) adopted this approach to the case of valuing a change in coastal wetland habitat (i.e. mangroves) in terms of the dynamic path of an open access fishery.

The dynamic specification is based on the assumption that growth of the stock of fish over time is a function of biological growth in the current period, net of any harvesting determined by the fish stock as well as fishing efforts in past periods. The influence of the estuarine ecosystem extent, composition, structure and functionality, is assumed to be positive, meaning that an increase and improvement in estuarine attributes would mean more carrying capacity for the fishery and thus greater biological growth.

Although the production function approach is theoretically sound, we have a poor understanding of the linkages between management actions and ecosystem functioning; their linkages to the supply of ecosystem services and linkages to the value of these services to humans (Barbier et. al. 2009). It follows from the above that the formulation of production functions that adequately describe likely consequences of changes in multiple variables of complex systems, requires a combination of expert knowledge and empirical data. This research intends to extend the production function approach to explicitly integrate effects of changes in biological diversity and other ecological factors.

## 2.6 Conclusion

The MEA presents a new approach to the analysis of the interface between ecology and economics (MEA 2005, MEA 2007). However, theory, methods and appropriate data are considered yet insufficiently developed to support a comprehensive valuation of ecosystem services, particularly, supporting and regulating services. Thus the outputs of the MEA (and TEEB) challenged economists to develop methods for defining the consequences of ecological change induced by current economic activity; methods for analyzing the distribution of possible outcomes of alternative activities and, the probabilities attached to those outcomes; and methods for developing appropriate mitigating or adaptive policies.

The literature review demonstrated that, although a large number of techniques have been developed for valuation of provisioning and cultural services, the valuation of regulating services is still in its infancy. The literature review further demonstrated that a production function approach provides an appropriate methodological framework for valuation of regulating and supporting services. Both static and dynamic formulations of fishery production function models have been applied to measure effects of changes in coastal and estuarine ecosystems' functionality and fish harvest and consequent impacts on economic welfare.

Most attempts to value habitat-fishery linkages via a dynamic model that incorporates stock effects have assumed that the fishery affected by the habitat change is in a long-run equilibrium. Valuing the change in estuarine ecosystem attributes in terms of its impact on the long-run equilibrium of the fishery raises additional methodological issues. The literature provides examples of pure fisheries models that assume that the dynamic system is not in equilibrium but is either on the approach to a steady state or is moving away from initial fixed conditions.

Dynamic specifications assume that growth of the stock of fish over time is a function of biological growth in the current period, net of any harvesting determined by the fish stock as well as fishing efforts in past periods. The influence of the supporting coastal and estuarine ecosystem extent, composition, structure and functionality, is assumed to be positive, meaning that an increase and improvement in estuarine attributes would mean more carrying capacity for the fishery and thus greater biological growth.

# 3. Chapter 3: Approach and methods of the study

## 3.1 Introduction

This research contributes to the badly needed effort to develop and apply improved methodologies to support the valuation of regulating and in some cases, supporting ecosystem services. The literature review demonstrated that a production function approach provides an appropriate methodological framework for valuation of regulating and supporting services. A production function-based methodology of this nature requires the consideration of several methods, techniques and principles.

Firstly, we are dealing with complex systems, which comprise many variables, complex linkages and feedbacks. These variables are likely to vary temporally, sometimes on daily (e.g. tidal or rainfall variation), monthly (e.g. river flow or tidal cycle variation), seasonally (e.g. seasonal rainfall variation), and other time frames (e.g. El Niño, global change related variation). They may also vary at a spatial scale, due to spatial variation in geographical, biological and other biotic and abiotic features (i.e. ecological factors). The methodology therefore requires, as a point of departure, a hypothesis of how the ecosystem that is being studied, works. Such a hypothesis would require an understanding of the key variables in the system and a description of how these variables interact. The MEA introduces such systems description as a "*chain of causality*".

Secondly, the economic-ecological production function methodology would have to integrate, in one analytical framework, a multiplicity of economic and ecological attribute indicators. Economic indicators would include economic output parameters, prices and cost of factors of production. Ecological indicators would describe all the variables that describe the ecological system, as envisaged above. This would therefore include variables that describe the extent, composition, structure and functionality of the system under study.

Thirdly, the production function methodology would have to allow for the analysis of the system in a way that the marginal impacts of potential changes or shocks in the system may be assessed, not only using the mean, but also using higher moments of the distribution.

The practical challenge to doing this lies in the application of econometric analysis, which allows for evidence-based multivariate analysis. Econometrics requires the application of time series data or cross-section data for assessing assumptions of economic theories, in this case resource economic theories, based on existing scientific knowledge. Data for econometric analysis is sourced either from primary (surveys) or secondary sources (records from third party monitoring or controlled experiments' results).

The key problem associated with economic analysis is the availability of suitable evidence. Two types of evidence are required here: evidence that supports the formulation of a hypothesis of how the socio-ecological system works, or, as the MEA envisaged it, the chains of causality; and thereafter suitable and sufficient data that measures the dependent and independent variables required to construct the production functions.

In the case of the system hypotheses, it is doubtful whether explicit ecological modeling has the ability to adequately capture the complexity of ecosystems and their social interactions into such a systems description. It is perhaps more feasible to adopt a pragmatic approach, based on tacit, local, expert knowledge, to develop a conceptual production function model of a particular ecosystem; which is thereafter supported by empirical evidence (Breen 2009). This would be akin to forming an expert-informed hypotheses of how a particular ecosystem works and thereafter empirically testing these hypotheses, using the production function approach and suitable available data.

Finding suitable and sufficient empirical data is likely the more challenging part of the evidence problem. In most physical and biological sciences, scientific experimentation is regarded as the only method through which to generate evidence about the relationship between two or more variables. Scientific experimentation may either collect new primary data through an appropriate sampling method (if budget allows), or directly apply the outputs of an appropriate scientific experiment conducted elsewhere. The scale and variability associated with complex systems makes it impractical to apply this evidence collection method to the research at hand. One way of overcoming this problem is through meta-analysis. This method however relies upon the availability of a large body of previously completed scientific experimentation or environmental monitoring studies on the relevant aspects of the same system. Another form of empirical data collection is data mining, a technique commonly associated with economic analysis.

This research attempts to adapt and apply an ecosystem services framework and use the concepts of composition, structure, and function of an ecosystem to formulate and estimate production functions specifying inks between ecological infrastructure and biodiversity to the economy. This requires a case study of a suitably bounded system, where empirical data may be collected through some means.

Thus, using existing scientific knowledge and data sourced from existing databases and studies, this research develops and empirical estimates ecological production functions to measure such relationships in the KZN fisheries along the east coast of South Africa as a case study. This is demonstrated for marine and estuarine systems separately.

### 3.2 Advantages of the production function approach

The advantages of such ecological production function approach, when applied within the MEA framework, are numerous:

- This approach takes as point of departure the tacit and other expert knowledge of the complex system under study. It relies on this knowledge to develop hypotheses about the system, in other words, to describe the chain of causality.
- It also provides an evidence-based analytical framework for examining relationships that link ecosystems and the economy.
- Through the use of suitable economic and ecological indicators, and an econometric approach, it enables integration of regulating (and supporting)

services, and, where relevant, provisioning and cultural services when they play an intermediate role, into the economic analysis problem.

- It allows of the use of a variety of empirical data, both time series and cross-sectional.
- In addition, it would enable the deeper analysis of the regulating value of ecosystems with the purpose of (after Perrings 2006):
    - *"Understanding the consequences of ecological change induced by current economic activity;*
    - *Understanding the distribution of possible outcomes of alternative activities and, where feasible, the probabilities attached to those outcomes; and*
    - *Developing appropriate mitigating or adaptive policies*."

### *3.3* Evidence considerations and empirics

Production function development is data intensive and requires consideration of the concept of "evidence".

Management of ecosystems, as in law, requires decisions to be made on best available evidence within a limited timeframe (Miller and Miller 2005). Evidence is information that supports (or undermines) a proposition, whether a hypothesis in science, a diagnosis in medicine, or a fact or point in question in a legal investigation (Miller and Miller 2005). Evidence is the consideration of both the data as well as how much and in what ways we can infer from that data with some degree of confidence. Evidence is popularly also referred to as the currency by which one fulfils the burden of proof.

In common law, evidence includes anything that can be used to determine or demonstrate the truth of an assertion. Evidence may include testimony, documentary evidence, physical evidence, digital evidence, scientific evidence, demonstrative evidence, eyewitness identification and genetic evidence.

Standards of proof are important. Well-defined evidentiary standards are applied in law, through which to judge a "standard of proof appropriate to the fact or point in question" (Miller and Miller 2005). The standard of proof is the level of proof required in a legal action to convince the court that a given proposition is true (i.e. to discharge the burden of proof). The standard of proof required depends on the circumstances of the proposition. These range from the lowest, the Precautionary Principle, to 'more likely than not' (the balance of probability) or 'clear and convincing' being a standard used in civil cases, to criminal standards of 'beyond a reasonable doubt' (Table 1).

**Table 1. Legal and scientific standards of proof (taken from Miller and Miller 2005). The precautionary principle is a standard of evidence that is to be applied in the absence of scientific evidence.**

| Type of evidence | Level of Evidence | Standard |
|---|---|---|
| Regulatory, Legal | | Precautionary Principle |
| Legal — Civil | * | More likely than not |
| Legal — Civil | ** | Clear and convincing |
| Legal — Criminal | *** | Beyond a reasonable doubt |
| Scientific | **** | Irrefutable |

Thus, the proof of an assertion, within the context of the degree of certainty required, depends on both the quantity and quality of evidence. Sufficiency of evidence in the case of a decision that needs to be made is thus dependent on the degree of certitude required. The standard of proof depends on the magnitude and likelihood of consequences to human well-being.

The Precautionary Principle is an interesting standard of proof with relevance here. It is derived from the 1990 Bergen Declaration, which states, "*Where there are threats of serious or irreversible damage, lack of full scientific certainty should not be used as a reason for postponing measures to prevent environmental degradation*" (Miller and Miller 2005). The precautionary principle is broadly analogous to 'probable cause' and it therefore requires a low standard of proof, and often forms the basis for environmental impact assessment decisions and authorisations. The Precautionary Principle obviously has important evidentiary implications for application of the production function approach in analysing complex systems.

Expert knowledge is a form of evidence that is required to develop hypotheses about how system work (the chain of causality). Other evidence-collation techniques, including meta-analysis and database mining would be required to generate data for measuring the dependent and independent variables of the production functions.

Meta-analysis of selected completed scientific studies typically follows the methodology defined by the Centre for Evidence-based Conservation (CEBC 2014). In this method, the results or data from a sufficient number of experimental studies, which have generated a mean and a variance, are pooled and analysed statistically. This pre-supposes that a suitable body of completed scientific studies on similar complex systems exist.

Data-mining involves the analysis of data from existing databases and other data embedded in reports or studies, as a source of evidence. The abundance of electronic databases is increasing rapidly, and vast data on land use, water quality, habitat quality, water flow and the like become increasingly available. The availability of such databases, combined with rapidly increasing computing power, enables simultaneous analyses of GIS data, water data and economic data, in ways that have not been possible before.

The evidence requirements outlined above requires a case study approach to be followed in order to collect sufficient evidence for the completion of this research.

The case selected is of the estuarine and marine ecosystems of the KZN east coast of South Africa. This system produces fish upon which commercial, subsistence and recreational fishers depend. It also plays an important functional biodiversity role in connecting terrestrial and estuarine system processes and components to marine-based species dynamics. The marine system is centred on the Thukela Bank, a coastal shelf that forms the centre of a line-fishing industry. The estuarine ecosystem consists of approximately 70 estuaries that have sub-tropical characteristics. Since the early 1980s, many researchers have conducted numerous studies on the physical components, processes and human use of these systems, and thus a rich set of databases exist upon which to base the current research.

## 3.4 Case study: marine and estuarine ecosystems in KZN

### *3.4.1. The study area*

KwaZulu-Natal is one of the 9 provinces of South Africa. It is located in the southeast of the country and has a long shoreline adjacent to the Indian Ocean. Its capital is Pietermaritzburg and its largest city is the port city of Durban. Another city is the industrial port city of Richards Bay, north of Durban. The climate of the coastal areas is humid and subtropical.

The coastline is characterised by many small towns, many of which are recreational hubs, and associated with estuaries. The KwaZulu-Natal coast is also renowned for its sardine run, which occurs when millions of sardines migrate from their spawning grounds in the south toward KwaZulu-Natal.

Marine ecosystems therefore play a fundamental part in the socio-economy of the province.

### *3.4.2. Marine system description*

The estuarine meta-system of the case study area includes a freshwater aquatic system, which connects, through a system of estuaries, to a marine system.

The east coast of South Africa forms a bight (a gentle inward curve of the shoreline) between Cape St Lucia and Durban, known as the KZN Bight (Figure 1). Seaward of the KwaZulu-Natal Bight lies the Thukela Bank, a section of the coastal shelf, very narrow north and south of the Bight, but widening along the Bight to a maximum width between Richards Bay and the Thukela River mouth. This coastline, with the continental shelf, and the estuaries and their catchments, which extend all the way to the continental divide on the Drakensberg Mountain Range, at about 3,000 m amsl, defines the character of the estuarine and marine ecosystem

The Agulhas Current (AC), one of the major western boundary ocean currents of the world, tends to follow the edge of the African continental shelf. The Coriolis force drives the waters of the AC south-west toward the coast.  The waters of the current are warmer than both the water inshore (over the shelf) and the deep sea. The AC flows rapidly in a more-or-less constant direction south-westwards and usually has a darker blue “oceanic” colour (compared to the greenish colour of the coastal water). Its surface is often choppy. Having traversed the tropical zones north and east of Madagascar, the waters of the current arrive on the KZN coast depleted of inorganic nutrients after phytoplankton uptake during their passage through the tropics. The AC dominates the oceanography of the Thukela Bank in several ways (CSIR 1998):

- First, Agulhas waters are depleted of inorganic nutrients, arriving at the shelf in “typically low” concentrations after phytoplankton uptake during their passage through the tropics;
- However, the Current induces sporadic upwelling of colder water, with the net effect of bringing somewhat higher concentrations of inorganic nutrients in the surface waters near the coast along the Bight, this effect being substantially greater than delivery of nutrients in rivers, and apparent all along the shelf of the Bight;
- There is thus periodic oscillation between low-nutrient surface waters, and somewhat richer upwelled waters, but generally the water is nutrient-poor; and
- The overall oceanographic effect is to create a somewhat confined nearshore current system, in which currents and turbulence over the shelf (the Thukela Bank) are wind-driven, except that occasional eddies, or gyres, veer off from the main Current to “pulse” southward over the shelf (Connell 2012).

Thus, the AC, coastal topography and wind together create a fairly complex system of predominantly wind-driven inshore currents and topographically driven “*semi-permanent eddies of expanded cooler inshore water, with predominantly north-going currents on their inshore edges*” (Connell 2012). Two such eddies exist, namely an eddy in the KwaZulu-Natal Bight just north of Durban extending up to Richards Bay (A in Figure 2) and another centred just south of Port St Johns (B in Figure 2). Another “*disrupting feature to inshore currents, is the passage of a ‘Natal pulse’, essentially a moving eddy of water trapped inside the AC, which is generated when a topographically generated eddy becomes unstable and breaks free to move down the coast (or a large inshore deflection of the Agulhas current becoming isolated), with a rapid clockwise rotation (when viewed from above), generating unusually* *powerful north-going currents on its inshore edge*" (Connell 2012). These semi-permanent eddies act as important stages for fish migrating up the coast as "*their inshore edges have predominantly north-going currents*" (Connell 2012).

The Thukela Cone is the major depositional feature on the coastal shelf of the Bight, formed by the transport of sediments from the steady cutting back of the basalts of what is now the Drakensberg Mountains, and their underlying sedimentary rocks. Flemming and Hay (1988) estimate the recent annual sediment supply to the shelf at about 14.2 tonnes per year, of which about 5.9 tonnes comes from the Thukela River, and 4.3 tonnes from the Mhlathuze River and its adjacent catchments. However, these are averages, and most silt arrives when rivers are in spate.

The coastal shelf sedimentary deposits are mainly sands though a relatively small area off the Thukela River mouth is muddy. This is within a so-called deposition zone, where gyres (see below) favour the accumulation of fine sediments.

The deposited sediments on the coastal shelf are reworked by wave action driven by winds near the coastline. Generally, the nearshore waters are turbulent, from wind-driven wave movement, preventing the deposition of muds, except in the few deposition zones that arise from eddying over the Thukela Bank (Connell 2012). Inshore, wind-driven waves create turbulence in the coastal waters, to a depth of 30 m or more over the shelf. This action scours the shore and seabed. Thus, the waters off the Bight are usually more turbid that those north and south, and seaward of the shelf.

The numerous rivers that enter the sea are very important for a number of reasons. The rivers discharge large quantities of sediment into the sea that settle to provide important habitat to benthic organisms. Moreover, terrestrial plant material washed down rivers, together with decaying seaweed, is macerated by wave action and provides a major input of nutrients to filter feeders such as mussels, oysters and ascidians (whose high biomass is certainly linked to the high diversity and abundance of carnivorous near-shore teleost fish (van der Elst 1988).

There are 20 rivers that flow via estuaries into the ocean along the KwaZulu-Natal Bight (see Figure 1). Of these, St Lucia, Richards Bay Harbour, the Mhlathuze

Estuary and Durban Bay are the largest in surface area. These estuaries are at the extremity of catchments subject to severe erosion. These estuaries are subject to cycles of sudden erosional deposition, followed by comparatively long periods during which the sediments settle and are formed and eroded by normal inflows.

### *3.4.3. Estuarine system description*

Various compositional and structural estuarine ecosystem attributes are of interest here and relate to important regulating services. The regulating services of interest include:

- Habitat type and extent, in particular with respect to hydrodynamics and primary production;
- Salinity regulation; with respect to species diversity; and
- Nutrient cycling, with respect to primary production.

Hydrodynamics and nutrient cycling are important because the ecosystem services yielded by estuaries depend in the first place on the flow of water and the materials it carries (Turpie, Lamberth 2010; Gillanders, Kingsford 2002; Van Ballegooyen *et.al.* 2005; Lamberth *et.al.* 2009). Both tidal flows and freshwater inflows are important. Freshwater inflows, especially during spates, deliver nutrients in the form of dissolved and suspended solids forms. Much of the suspended solids settle in the estuaries (Cooper *et.al.* 1999) while the rest passes into the main water body, where it precipitates as mud in the benthos, or, if organic, is carried out with the tides or decayed or comminuted, and then consumed by detritus feeders. Primary production in estuaries comes from photosynthetic microorganisms, mainly microalgae, in the shallow benthos; and phytoplankton in the water body and mangroves. The rich benthic filter- and particle-feeding communities of the tidal flats, especially the sub-tidal flats, are key to primary production. They provide energy and nutrition to the fry of sea-breeding and other fish in the shelter of the shallows and thus form the link in the local and regional fisheries chain. The benthic animals and the abundant small fish attract predators, such as fish and birds, the basis for recreational activities and other services (Kruger *et.al.* 2007).

Attributes of the ecological infrastructure of an estuary include variables describing type of habitat services provided; areas of tidal flats and mangroves, measures of degree of openness reflecting retention time and accessibility to juvenile fish; and measures of the volume of detritus available to estuaries from freshwater runoff.

Estuaries are often referred to as transitional waters, as a transition takes place between freshwater and marine environments. This transition is characterised by varying salinity levels within the fresh and seawater mixing zones. Within the mixing zone, salinity values vary, and this affects fish species diversity. Salt dissolved in water applies called osmotic pressure on the cell walls of living organisms, dehydrating it. Organisms living in sea water are equipped with buffering mechanisms allowing them to retain their body fluids in the presence of salt. In most cases this mechanism is adjusted to a particular dissolved salt level and yields a distinction between freshwater and marine species, into which most organisms of aquatic life fall. In turn very few species can withstand variable salinity, which has implications for the biodiversity of estuaries. Some estuarine species are adapted to varying salinity levels. For example, Euryhaline marine species (i.e. able to tolerate a wide range of salinity) usually breed at sea, with juveniles showing varying degrees of dependence on southern African estuaries. Species that can tolerate lower salinity levels can therefore also find refuge from more marine-based predator fish (Marbef 2014).

It follows from the above that the compositional element species diversity is also an important measure for consideration.

### *3.4.4. Fish production*

The KZN coast lies within a recognised area of fish endemism, namely the south-western Indian Ocean, which lies within the Indo-Pacific region of high fish species diversity. More than 73% of the approximately 200 fish species from over 150 families are primarily Indo-Pacific, 16% are endemic to the subtropical estuaries of the KZN coast, a number of which are classified as endangered or threatened by the IUCN Red List of Threatened Species (van der Elst 1988, Weerts 2002). The majority of these species are strongly associated with estuaries (Whitfield 1998 in Weerts 2002).

Reasons for the high endemism in this region are directly related to physico-chemical factors that influence fish dispersal, namely (from van der Elst 1988):

- Sea surface temperatures that lie predominantly between 20˚C and 25˚C, but fall below 20˚C fairly regularly;
- The limited (at this temperature) extent of reef-building corals, present north of Kosi Bay and in much of the rest of the Indo-Pacific;
- Surface salinity that is generally higher (>35 ppt) than in other parts of the Indo-Pacific;
- The water bodies and eddies that regularly remain localised further restrict the distribution of fish, likely through regulating nutrients.

The KZN fish fauna belong to an open ecosystem that has distinct seasonal variations in temperature and current that influence fish migration and spawning patterns giving rise to four categories (from van der Elst 1988; Weerts 2002):

- Resident fish that spawn locally and have local nurseries (many of which are estuaries);
- Pelagic summer migrants from tropical Indian Ocean waters that spawn in the tropics and have distant nurseries;
- Demersal (benthic feeders) or estuary summer migrants from south-eastern Mozambique that spawn locally and have local nurseries (many depend on the littoral and estuary environments for the recruitment of their juveniles);
- Winter migrants from Cape waters once the temperatures drop below 21˚C that are mostly endemic to South African waters, spawn locally and use the AC as a dispersal mechanism to transport eggs and larvae to the nutrient-rich Cape waters and nurseries.

### *3.4.5. Fishing activities*

The system produces fish stocks, which support a significant shore and boat recreational fishing industry, a small commercial line-fishing industry, and various subsistence fisheries (Turpie, Lamberth 2010).  A range of other ecosystem services, in addition to fish biomass, are also produced, but for practical considerations and availability of suitable data those have not been considered in this study.

The KZN province is a well-known recreational angling destination and various studies have quantified the economic size of the above fisheries, both in estuarine and marine ecosystems (Guastella and Nellmapius 1993, McGrath *et. al.* 1997, Brouwer *et. al*. 1997, Mann *et.al.* 2002, Lamberth and Turpie 2003, Kruger *et.al.* 2007, Beckley *et. al.* 2008, Crafford *et. al.* 2008). These studies report various economic indicators, use different valuation techniques and report for a variety of years. Based on a meta-analysis of these studies, the size of the total fisheries industries may vary between R900 million and R1,400 million per year (2013 prices), measured in terms of its contribution to Gross Domestic Product (GDP). The variation results from changes in catch and in input and output prices that vary from year to year. Interestingly, the largest component of this industry is recreational shore and boat fishing (in estuarine and marine systems), where the expenditure on fishing efforts and associated travel costs constitute key determining factors. From the meta-analysis it is estimated that the total number of angler outings in KZN could be approaching 2,000,000 per year, with an estimated total expenditure on recreational shore angling approaching R650 million in 2013. Another very significant component is subsistence fisheries, the contribution of which accrues in the informal sectors of the economy. However, no data is available on this activity.

The above scientific knowledge creates an important starting point for the development of ecological production functions, and thus the following section derives the structure of production functions, using this knowledge.

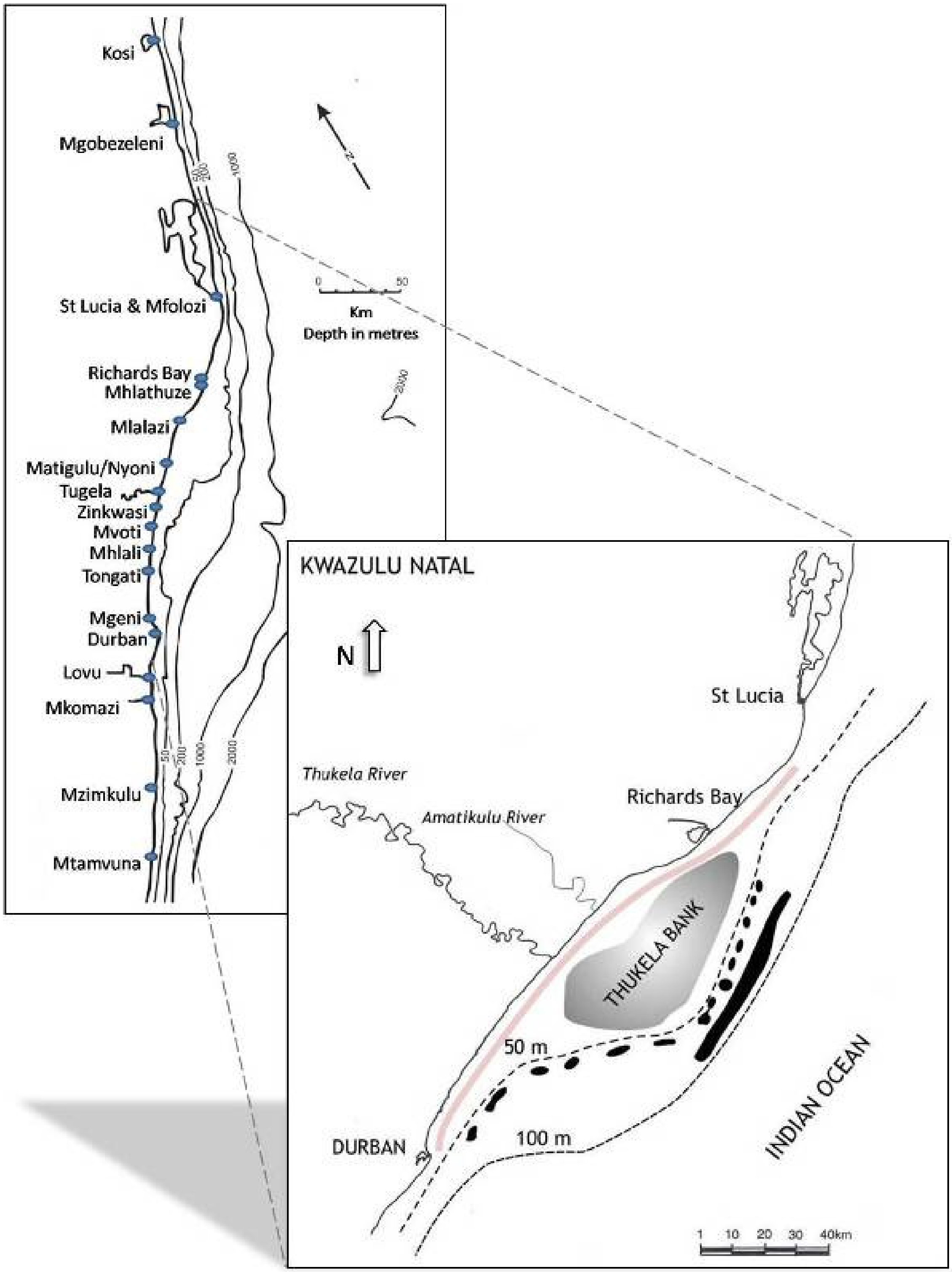


**Figure 1. KwaZulu-Natal coastline showing the location of large estuaries (>100 ha) with an inset of the KwaZulu-Natal Bight (shaded pink), Thukela Bank (shaded grey), reef areas (black) and -50 m isobaths (adapted from Forbes and Demetriades 2005).**

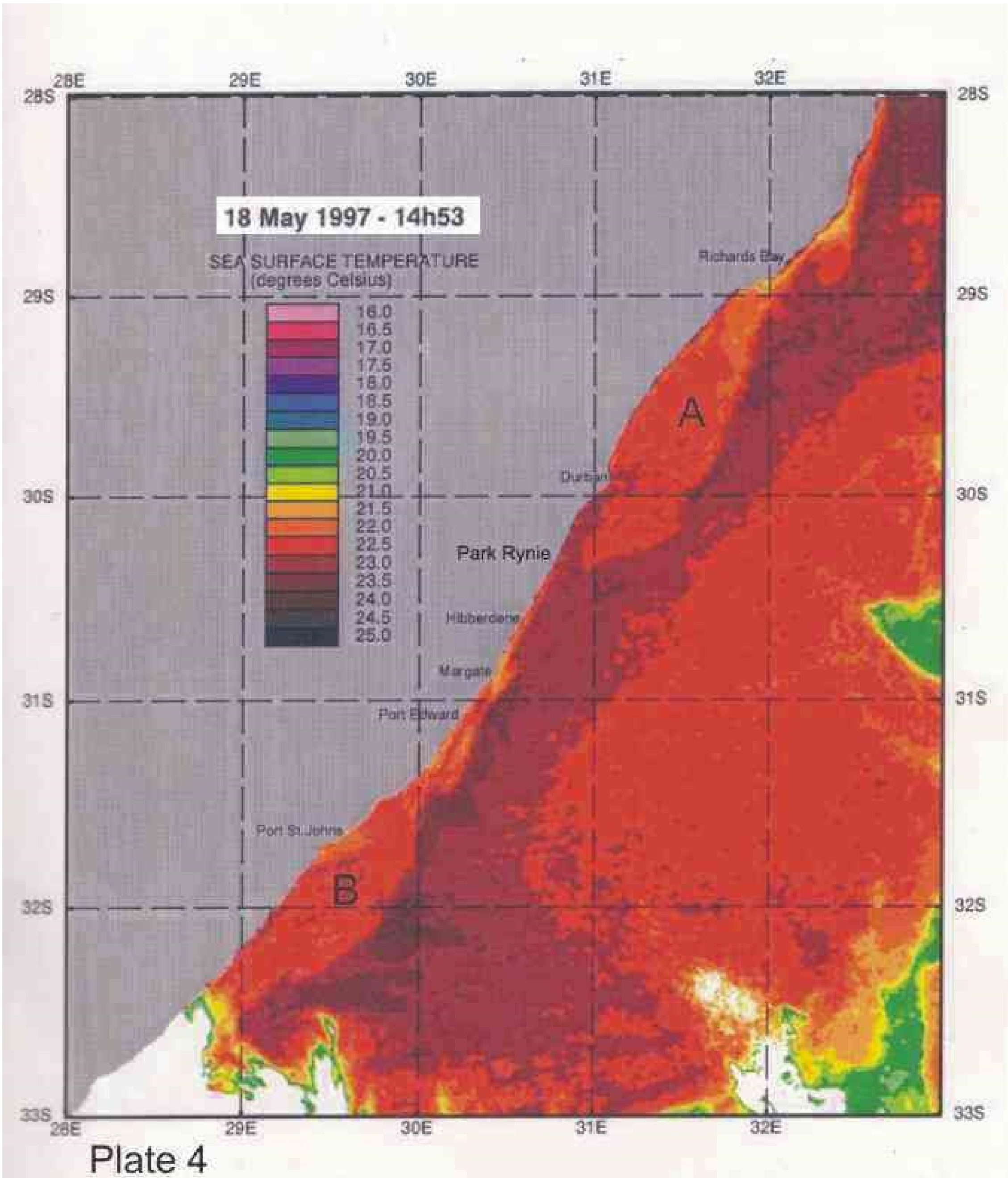


**Figure 2. Sea surface temperature along the KwaZulu-Natal coast illustrating two semi-permanent eddies of lower temperature centred in the KwaZulu-Natal Bight between Durban and Richards Bay (A) and just south of Port St Johns (B) (taken from Connell 2007). The sea over the Thukela Bank (A) is in turn distinguished by smaller-scale serial eddies over the shelf (see text).**

### 3.5 The fishery ecological production function analytical framework

The production function approach treats an ecological service, such as estuarine habitat (in this case a supporting service), as an input into the economic activity, and, like any other input, its value can be equated with its impact on the productivity of any marketed output. The following specification has been generally used to model the relationship between fish harvest $H$ and attributes of the underlying ecological infrastructure:

$$H = h(E_i, S_i) \qquad (1)$$

Where $E_i$ denotes economic effort (inputs) of a commercial fishery and $S_i$ denotes ecosystem attributes, which are independent of $H$ but assumed to have a direct influence on the fish catch, $H$[1]. A typical assumption of an open access fishery is that effort next period adjusts in response to profit levels made in the past period (Bjørndal and Conrad 1987). It is considered inappropriate, however, to model the dynamics of fish production systems based only on impacts on current harvest and market outcomes. This weakness was addressed by developing dynamic models incorporating the change in estuarine system functionality within a multi-period production model of the fishery. Examples exist of pure fisheries models that do not assume equilibrium but a state in which the dynamic system is either approaching a steady state or is moving away from initial conditions (Bjørndal and Conrad 1987; Homans and Wilen 1997).

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[1] Barbier (2007) employed a Cobb–Douglas approach to estimate a static habitat-fishery production function model, which allows specification of a cost function for a cost-minimizing fishery of the form. This specification enabled computing new equilibrium harvest and price levels and corresponding changes in consumer surplus associated with a change in estuarine functionality, for a given demand elasticity (refer to Appendix 1 for additional derivation steps for the Barbier (2007) application).

The dynamic modeling starts with defining $X_t$ as the stock of fish measured in biomass units, and where any net change in growth of this stock over time can be represented as:

$$X_t - X_{t-1} = F(X_{t-1}; S_{t-1}) - H(X_{t-1};E_{t-1}) \tag{2}$$

Thus, net expansion in the fish stock occurs as a result of biological growth in the current period, $F(X_{t-1}, S_{t-1})$, net of any harvesting, $H(X_{t-1}, E_{t-1})$, which is a function of the stock as well as fishing effort, $E_{t-1}$. The influence of the estuarine ecosystem extent and functionality, $S_{t-1}$ on growth of the fish stock is assumed to be positive, $\partial F/\partial S_{t-1} > 0$ (e.g. as an increase and improvement in estuarine supporting and regulating services quality will mean more carrying capacity for the fishery and thus greater biological growth).

If we assume $p(h)$ to be the market price of landed fish, $w$ the unit cost of effort and $\varphi > 0$ the adjustment coefficient, then the fishing effort adjustment equation will be:

$$E_t - E_{t-1} = \varphi.[p(H_{t-1}).H(X_{t-1};E_{t-1}) - wE_{t-1}]; \partial p(H_{t-1})/\partial H_{t-1} < 0 \tag{3}$$

This model was further developed employing:

(1) The logistic biological growth and other ecological factors function

$$F(X_{t-1}, S_{t-1}) = \mathrm{R}.X_{t-1}.[1 - X_{t-1}/K(S_{t-1})] \tag{4}$$

(2) The Schaefer (1954) production process

$$H_t = q.X_t.E_t, \tag{5}$$

Where $q$ is the ‘catchability’ coefficient, R is the intrinsic growth rate and $K(S_t)$ defines the impact of biomass and estuarine extent and functionality on the carrying capacity, $K$, of the fishery.

(3) An iso-elastic market demand function for harvested fish

$$P = p(h) = KH^{\eta}; \quad \text{where } \eta = 1/\varepsilon < 0 \tag{6}$$

Substituting these expressions into (2) and (3) above gives the following system of equations:

$$X_t = R.X_{t-1}.[1-(X_{t-1}/K(S_{t-1}))] - H_{t-1} + X_{t-1} \quad (7)$$

$$E_t = \varphi.R_{t-1} + (1+\varphi w).E_{t-1} \text{ and } R_{t-1} = KH_{t-1}{}^{1+\eta} \quad (8)$$

Homans and Wilen (1997) argue that since both $X_t$ and $E_t$ are predetermined, equations (7) and (8) can be estimated independently. Following Schnute (1977) we can define the catch per unit effort as $C_t = H_t/E_t = qX_t$. If $X_t$ is predetermined so is $C_t$. Substituting the expression for catch per unit effort in (7) produces[2]:

$$(C_t - C_{t-1})/C_{t-1} = R - (R/q\alpha).(C_{t-1}/K(S_{t-1})) - qE_{t-1} \quad (9)$$

Thus equations (8) and (9) can be estimated independently to determine the ecological and economic parameters of the model. For given initial effort, harvest and biomass data, both the effort and stock paths of the fishery can be determined for subsequent periods, and the consumer plus producer surplus can be estimated for each period (Barbier 2000 and 2007). Alternative effort and stock paths can then be determined as various ecological parameter changes in each period, and thus the resulting changes in consumer plus producer surplus in each period are the corresponding estimates of the welfare effects of the ecological parameter change.

### 3.6 Integration of ecological parameters as inputs into the production function

Equation 9 defines $S_t$ as an ecological input variable. In the literature, this variable has predominantly been treated as habitat area. For instance, Barbier (2000, 2003, 2007), used coastal wetland, or mangrove areas as measures of $S_t$. Habitat area alone however, is an inadequate indicator of the multiple regulating services supplied by coastal and marine ecosystems. Functioning of these systems is supported by a

[2] Refer to Barbier (2007) and Appendix 1 for more details on these derivations.

complex set of ecosystem processes, which regulate the production of provisioning and cultural services supplied by the system.

Following Noss (1990), biodiversity is characterized by composition, structure, and function within a particular ecosystem. Composition has to do with the identity and variety of elements in a collection, and includes species lists and measures of species diversity and genetic diversity. Structure is the physical organization or pattern of a system, from habitat complexity as measured within communities to the pattern of patches and other elements at a landscape scale. Function involves ecological and evolutionary processes, including for instance disturbances, and nutrient cycling.

Thus, key components of the estuarine and marine ecosystems include, for the most part, physical characteristics (ecological infrastructure) as well as processes that relate to the exchange of water, through terrestrial and marine inflows resulting from daily, seasonal and annual tidal, flood and storm events. As an example, productive estuaries generally have high nutrient content, high water body retention times and suitable substrate where nutrients can accumulate and thus feed the benthic and higher species within the system. Salinity variation also plays a major role in estuarine functionality.

Accordingly above model formulations need to be modified to account for such ecosystem attributes (e.g. species diversity, components and processes that regulate coastal and estuarine ecosystems functionality.)

Fish stock production is driven primarily by nutrient influx into the fish production system. Whitfield (1998) demonstrated this by proving the empirical relationship between detritus input into an estuarine system, and the stock of fish as measured by fish biomass. Furthermore, it is common cause that the more productive fisheries zones around the Southern African coast is driven primarily by oceanic nutrient upwelling. The eastern coast of Southern Africa is, however, located in a high nutrient-depleted section of the Indian Ocean and is thus believed to rely heavily on nutrient influx from terrestrial runoff (Connell 2010). If this hypothesis is true, nutrient influx into the system would be a sporadic event, tied closely to annual variations in climate, rainfall and, ultimately, runoff. Thus, the functionality of this system can be described by the relationship:

$$S_t = f(SN_t(g(SAR_t))) \qquad (10)$$

Where, as defined above, $S_t$ is an indicator of ecological attributes; $SN_t$ is an indicator of nutrient load in the fish production system; and $SAR_t$ denotes mean annual runoff.

However, nutrient and runoff data alone are insufficient to adequately describe ecological inputs into the production function. The nutrient load is supplied from approximately 8 million hectares of terrestrial catchment area along the KZN coast, and enters the fish production system through the ecosystem of sub-tropical 72 estuaries comprising approximately 43,000 hectares (ha) of estuarine area. Estuaries are important structural or physical components of the fish production system, for two reasons. Firstly, they serve as nutrient traps during the periods between flood events, and thus regulate nutrient supply into the system. This ecosystem service is thus characterised not only by the volume of nutrient influx into a particular estuary, but also by the water retention time inherent to every individual estuary. Water retention time is a factor of the physical attributes of the particular estuary, such as the dimension of the water body, the volume of daily tidal exchange and the physical complexity of the system which creates eddies and settlement areas within the estuary. Secondly, estuaries provide physical infrastructure, that are fundamental to the life cycle dynamics of nearly half the fish species caught commercially and for recreational purposes along the KZN coast. This physical infrastructure relates to various components of habitat where nutrients are trapped and thus enters the food chain through various levels of benthic species. Such habitats, primarily shallow sub-tidal sand and mud flats as well as mangrove swamps, thus serve a dual purpose in that they provide various levels of physical security to various fish and prawn species during the juvenile phase of their life cycles. Such species spawn on the Thukela Banks and/or the near-shore surf zones, from where the juveniles find their way into estuaries until maturity. Stable estuarine habitats provide both food and physical security from predators who are prevented from access either through physical encumbrances or salinity gradients (also refer to section 3.4.2). Thus, Equation 10 can be extended to include measures of the state of such structural and functional attributes:

$$S_t = f(SN_t(g(SAR_t)), STYPE_t, SHAB_t, SSAL_t) \tag{11}$$

Where $S_t$, $SN_t$ and $SAR_t$ are as defined before; $STYPE_t$ is an indicator of the physical characteristics of an estuary that measures both retention time and accessibility to juveniles (Whitfield, 1998); $SHAB_t$ is a vector of habitat types (either shallow sub-tidal sand and mud flats or mangroves) preferred by various fish or prawn species; and $SSAL_i$ is a measure of salinity and is an indicator of the ratio of seawater to fresh water in estuary *i*. Estuaries are prone to degradation due to various forms of human activity due to various forms of water pollution, physical destruction and other disturbances. As a result, various estuarine ecology studies have developed indicators of human disturbance through rating the condition of estuaries. The variable $SAL_t$ is thus included in Equation 11 to control for the effects of salinity variations.

Equation 11 so far internalises influences of key structural and functional elements of an estuarine ecosystem into the production function. However, the compositional element of the system need to also be taken into consideration. Evidence from various literature sources (refer to section 3.4) supports the hypothesis that a higher system resilience, and thus productivity, is achieved through higher levels of species diversity. Species-area studies further suggest that species diversity is significantly related to the available area of suitable quality habitat (Scheiner 2003), and thus Equation 11 is reformulated as:

$$S_t = f(SN_t(g(SAR_t)), SSPECIES_t\,(h(STYPE_t, SHAB_t, SSAL_t))) \tag{12}$$

Where $SSPECIES_t$ is an indicator of species abundance in the estuarine system.

### 3.7 Type and sources of the data

Data availability is often a significant limitation in economic studies (refer to section 3.3) and in this case, the approaches to solving Equations 8, 9 and 12 are limited by availability of suitable data. However, one of the objectives of this research is to explore the possibility of using data-mining of existing databases to develop empirical measures for the variables comprising Equations 8, 9 and 12.

At the outset it is difficult to find data on this system. However, with some effort, much data can be found on various aspects of the KZN estuarine and marine systems and several existing data-bases exist in published documents, official databases and grey literature. Since the early 1980s, many researchers have conducted studies on the physical components, processes and human use of these systems, and thus a rich database could be constructed to support the work done here.

This study ultimately had access to five different data sets.

The first two sets comprised cross-section data collected independently by two research teams from sub-sets of the east coast sub-tropical estuaries of South Africa are available. One of these datasets was compiled by Dr George Begg and was published in hard copy as *"The Estuaries of Natal Volumes 1 and 2"* (Begg 1978 and Begg 1984 respectively). This data was based on extensive environmental monitoring carried out on 72 estuaries during the late 1970s and early 1980s. The second source of cross-section data was available from Dr Trevor Harrison who compiled this information for the "*State of the Estuaries Report*" for the Department of Environmental Affairs in 2000 (Harrison *et.al.* 2000). The Harrison data evaluated 47 estuaries along the KZN coast. The combined Begg-Harrison (B&H) cross-section data recorded in excess of 120 measures of various estuary components and processes.

The purpose of these datasets were to establish baseline scientific information on the majority of estuarine ecosystems. The measures were collected from basic surveys conducted on these systems during the periods 1979-1982 and 1992-1999 respectively. This included ichthyofauna (fish), water quality, and geomorphological and aesthetic observations.

The above data sets comprised cross-section data and therefore limited the development of production functions to a static analysis of Equation 12. Detailed description of variables contained in these datasets and methods of their collection and measurement is given in Chapter 4. This cross-section dataset captured functional, compositional and physical infrastructure attributes of the estuarine ecosystem (e.g. species composition; nutrients supply and type; extent, and shoreline length and openness of the estuary; salinity) influencing productivity of estuarine-dependent fishery. This dataset however, did not contain information on economic effort and hence the empirical analysis of Chapter 4 could not account for the dynamic influences of economic factors.

Three additional datasets were available for empirical implementation of the dynamic formulation of the fishery model defined by Equations 8, 9 and 12. The first of these datasets provided time series records on commercial line fishing efforts and catch collected by the Department of Agriculture, Fishery and Forestry (DAFF 2012)[3], which allowed estimation of parameters of the economic factors effects on the studied KZN coast fishery. The DAFF dataset was combined with available time series records of fish egg abundance and fish egg biomass collected by Dr Allan Connell since 1987 (Connell, 2012). Annual spawning period, egg abundance and absolute egg count trends, together with egg and larval descriptions are provided in the Connell dataset. A third data set, time series data on Annual Runoff (AR) into the KZN coast, was also collected as part of the same fish egg data base.

The above three data sets comprised time series data and therefore enabled the development of dynamic production functions as described by Equation 8, 9 and 12. Detailed description of variables contained in these datasets and methods of their collection and measurement is given in Chapter 5. This time series dataset captured economic behaviour and functional and compositional attributes of the marine ecosystem (e.g. catch effort, harvest, nutrients supply) influencing productivity of the marine line-fish fishery. The resulting analysis of the dynamic formulation of the fishery production system is presented in Chapter 5.

[3] The data was obtained on request from the DAFF, extracted from a DAFF fish catch database.

## 3.8 Conclusion

The production functions developed in this study were formulated based on literature, outcomes of iterative systems analyses combined with Comparative Risk Assessment, yielding a systemic, conceptual description of various ecosystem attributes. Production functions estimate the indirect effects of changes in regulating services on an economic activity. Therefore, the physical changes in the ecological system, may be valued in terms of the corresponding change in the output of provisioning services. The formulation of production functions in this study followed a two-stage process. First, expert analysis, through the Comparative Risk Assessment (CRA), served to specify the independent variables of the various production functions. Second, available data was used in statistical models to predict ecosystem production.

The coastal fishery of KZN was selected as the case study area for implementing this study. The study area is located in the southeast of the country and has a long shoreline adjacent to the Indian Ocean. The coastline is characterised by many small towns, many of which are recreational hubs, and associated with estuaries. The KwaZulu-Natal coast is also renowned for its sardine run, which occurs when millions of sardines migrate from their spawning grounds in the south toward KwaZulu-Natal. Marine ecosystems therefore play a fundamental part in the socio-economy of the area.

The study had access to five different data sets to support the intended empirical investigation. The first two sets comprised cross-section data collected independently by two research teams conducting extensive environmental monitoring of key estuaries along the KZN coast. This datasets recorded in excess of 120 cross-section measures of various estuary components and processes and thus supported empirical specification and measurement of parameters of the static fishery production function model. This cross-section dataset captured functional, compositional and physical infrastructure attributes of the estuarine ecosystem (e.g. species composition; nutrients supply and type; extent, and shoreline length and openness of the estuary; salinity) influencing productivity of estuarine-dependent fishery.

Three additional datasets were available for empirical implementation of the dynamic formulation of the fishery model. The first of these datasets provided time series records on commercial line fishing efforts and catch that allowed estimation of parameters of the economic factors' effects on the studied KZN coast fishery. The said dataset was combined with available time series records of fish egg abundance and fish egg biomass collected by Dr Allan Connell since 1987 (Connell, 2012). A third time series data on AR into the KZN coast, was also collected as part of the same fish egg data base. These three time series data enabled the development and estimation of dynamic fishery production functions for the case study area.

# 4. Chapter 4: Economic-ecological production functions approach to measuring the relationship between estuarine ecosystem attributes and fish production

## 4.1 Overview

This chapter develops and estimates economic-ecological fishery production functions based on the compositional and structural biodiversity attributes of estuaries on fish production. The analysis uses cross-section data and static analysis to investigate the effects of habitat type, habitat extent, salinity variation and a static indicator of nutrient cycling on the diversity of fish species and the biomass of fish, as specified in Equation 12 above.

Section two of the chapter provides the context for the KZN coast estuarine-dependent fishery and attributes of the underlying ecosystem and advances hypotheses to be tested on the structural linkages between them. Section three develops the empirical model and describes types and sources of the data used to estimate the specified model parameters. Sections four and five present and discuss results of the empirical analyses and the final section derives conclusions and implications of the static model analysis.

## 4.2 The KZN estuarine fishery and estuarine ecosystem regulating services

The KZN province is a well-known recreational angling destination with a large dependence on estuarine ecosystems of which there exists more than 70 sub-tropical estuaries.

Recreational anglers undertake nearly 2,000,000 angler outings per year (Crafford *et. al.* 2008). Fish stock is therefore a key driver, and an important dependent variable in the analysis. Fish stock in turn is regulated by various compositional and structural estuarine ecosystem attributes. Important regulating services to be analysed here, following on from sections 3.4.2 and 3.6, include:

- Habitat type and extent, with respect to hydrodynamics and primary production;
- Salinity regulation; with respect to species diversity; and

- Nutrient cycling, with respect to primary production.

A key habitat indicator is the degree of openness of estuaries. This is measured as the percentage of time over a certain period that the estuary mouth is open to the sea, reflecting retention time and accessibility to juvenile fish.

Habitat and nutrient cycling are important because the ecosystem services yielded by estuaries depend on the flow of water and the nutrient. Freshwater inflows, especially during spates, deliver nutrients in the form of dissolved and suspended solids forms. Much of the suspended solids settle in estuarine habitat accessible to fish, where it precipitates as mud in the benthos, and then consumed by detritus feeders. The rich benthic filter- and particle-feeding communities of the tidal flats, especially the sub-tidal flats, are key to primary production. Thus, the extent of benthic habitat is of importance.

Measures of the volume of detritus available to estuaries from freshwater runoff are also important.

Estuarine transition zones are characterised by varying salinity levels within the fresh and seawater mixing zones. And this is important to fish species diversity.

It follows from the above that the compositional element species diversity is also an important measure for consideration.

The above described context and relationships between are formally represented by the fishery production model specified in Equation 12 of the analytical framework (Chapter 3). This model is empirically developed and estimated in the following sections.

### 4.3 Data sources and empirical model specification

The following set of production functions employ cross-section data to capture effects of estuarine attributes. The two cross section data sets described in section 3.7 was used here.

The combined Begg-Harrison (B&H) cross-section data recorded in excess of 120 measures of various estuary components and processes. Both B&H conducted extensive sampling and recorded the fish species abundance and fish biomassin the surveyed estuaries. Many other compositional and structural characteristics of the KZN estuaries were measured in the B&H cross-section data. Key among those are: (a) estuary depth and shoreline length used as proxies for the extent of very important tidal flat habitat; (b) the type of estuary reflecting degree of openness of the estuary; (c) measures of freshwater inflow into the estuary; (d) measures of salinity. These variables all have important roles in the production of estuarine ecosystem services. The degree of openness measures the connectivity of an estuary to the marine ecosystem. Highly connected estuaries (i.e. which are open for 12 months of the year) will be more productive than estuaries that are only temporarily open, and would be expected to have higher species abundance. The shoreline length (in kilometres) and area of shallow sub-tidal flats (in hectares) are both measures of habitat area. Begg does provide measures of terrestrial runoff, either directly as river inflow into estuaries, or indirectly as catchment areas which can yield runoff values. The B&H data also contain dissolved oxygen measurement in mg/litre which is indirectly proportional to nutrient content, as higher dissolved oxygen in these estuaries is commonly associated with lower nutrient load conditions (Chambers *et. al.* 2006). Higher dissolved oxygen is primarily a result of larger systems open and subject to marine flushing rather than nutrient concentrations. The salinity of the water body is measured in numerous ways and are reported.

A notable difference in the Begg and Harrison data sets relates to their fish sampling gear, which was very different. Begg used a small beam trawl, while Harrison used a variety of nets including seine nets and gillnets (personal communication Dr A Connell 2013). Therefore this research used only the Harrison data set for fish species abundance and biomass data, with data standardised to a catch per unit effort.

The analytical framework developed using the B&H cross-section data set allowed specification of the following system of ecological production function equations:

$$SBIOMF_i = \alpha_0 + \alpha_1 * SSPECIES_i + \alpha_2 STYPE_i + \mu_i \qquad (13)$$

$SSPECIES_i = \beta_0 + \beta_1 {*} SSAL_i + \beta_2 {*} SSHRLN_i + \beta_3 {*} STYPE_i + \beta_4 {*} SNUTRI_i + \varepsilon_i$ (14)

Where, $SBIOMF_i$ measures the total weight of fish biomass caught in grams per trawl sample in estuary i (and is therefore an indicator of fish stock); $SSPECIES_i$ is the number of fish species in the sample from estuary *i*. $SSAL_i$ is a measure of salinity and is an indicator of the ratio of seawater to fresh water in estuary *i*; $SSHRLN_i$ measures the length of estuary i shoreline in meters. $STYPE_i$ is an index that refers to the classification of estuary i as defined by the Whitfield physical classification of estuaries (Table 2); $SNUTRI_i$ is an index of the nutrient capacity of the estuarine system calculated by dividing the catchment area (in $km^2$) by the volume of the estuary water body (in $m^3$); $\alpha_j$ and $\beta_j$ are model parameters and $\mu_i$ and $\varepsilon_i$ are the residual error terms.

Equation 13 specifies the biological production of fish to be a function of the number of species present in the system, and the type of estuary. In Equation 14 the number of species present in the system (species abundance) is modelled to vary with differences in key ecosystem component and process variables.

**Table 2. Whitfield's (1992) physical classification of estuaries. The resultant Index is an important indicator of estuarine habitat type. The estuaries in the case study area contains various estuaries of all these types.**

| Type | Index | Tidal prism | Mixing process | Average salinity* |
|---|---|---|---|---|
| Estuarine bay | 5 | Large (>10 x 106 m3 ) | Tidal | 20 – 35 |
| Permanently open | 4 | Moderate (1-10 x 106 m3) | Tidal/riverine | 10 - >35 |
| River mouth | 3 | Small (<1 x 106 m3) | Riverine | <10 |
| Estuarine lake | 2 | Negligible (<0.1 x 106 m3) | Wind | 1 - > 35 |
| Temporarily closed | 1 | Absent | Wind | 1 - > 35 |
| * Total amount of dissolved solids in water in parts per thousand by weight (seawater = ~35 ) | | | | |

## 4.4 Empirical estimation results and analyses

The estimation results that follow show that fish biomass in estuaries are significantly influenced by species diversity ($SSPECIES_i$), estuarine type ($STYPE_i$), salinity ($SSAL_i$), habitat ($SSHRLN_i$), and nutrient load.

The first key variable of interest is species diversity. The fact that variable *SPECIES* appear on both sides of the system Equations 13 and 14 implies an endogeneity problem and hence non-suitability of Ordinary Least Squares (OLS) estimation. A Two-Stage Least Squares (2SLS) procedure was accordingly employed to estimate the parameters of this system, which unlike OLS yields consistent estimators of system parameters (Johnston 1984). Two functional form specifications have been tested: the linear and Cobb-Douglas (double-log) forms[4]. The linear function gave the best statistical fit as evidenced by the significance of the independent variables (P-values). Results of the 2SLS estimation of the linear system are presented in Table 3. Results of the double-log function are reported in Table 4.

The influences of all factors in the linear 2SLS model are of high statistical significance and all show the expected sign, i.e. direction of effect. Equation 13 relates the number of species ($SSPECIES_i$) and the estuarine type ($STYPE_i$) to fish biomass ($SBIOMF_i$), and these two variables explained 43% of the variation in fish biomass ($SBIOMF_i$).

As expected, catch size of fish biomass in an estuary increases with increasing number of fish species at more than 99.9% confidence limit. This result is of high importance as it provides strong scientific evidence that biodiversity, as measured by species abundance, is positively correlated with biomass, and thus the productivity of the estuarine system. Similarly, levels of fish biomass in an estuary is higher in larger estuaries with a higher degree of openness, as measured by the Whitfield physical classification index of estuaries ($STYPE_i$ was significant at a 99% confidence limit).

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[4] OLS regression requires that the regression equation be linear in the parameters. However, the equation may be either linear or nonlinear in the variables. Cobb-Douglas is a functional form which non-linear relationships.

**Table 3. 2SLS estimates of parameters of the system of Equations 13 and 14 for the Linear model. All the independent variables are significant and the model provides a very good explanation of the variability in the dependent variable (high $R^2$).**

| | Coefficient | Std. Error | t-Statistic | Prob. |
|---|---|---|---|---|
| $\alpha_0$ | 17,708 | 6209.448 | 2.851823 | 0.0055 |
| $\alpha_1$ | 661.6 | 139.7012 | 4.735973 | 0.0000 |
| $\alpha_2$ | 5,964 | 2301.972 | 2.590676 | 0.0113 |
| $\beta_0$ | 26.532 | 2.924246 | 9.073188 | 0.0000 |
| $\beta_1$ | 0.3129 | 0.079973 | 3.913315 | 0.0002 |
| $\beta_2$ | 0.000414 | 4.53E-05 | 9.150209 | 0.0000 |
| $\beta_3$ | 8.317 | 1.425839 | 5.833169 | 0.0000 |
| $\beta_4$ | 0.017685 | 0.007858 | 2.250593 | 0.0270 |
| | | | | |
| Determinant residual covariance2.59E+09 | | | | |
| | | | | |
| Equation 13: $SBIOMF_i = \alpha_0 + \alpha_1*SSPECIES_i + \alpha_2*STYPE_i$ | | | | |
| Adjusted R-squared | 0.4298 | | | |
| | | | | |
| Equation 14: $SPECIES_i = \beta_0 + \beta_1*SSAL_i + \beta_2*SSHRLN_i + \beta_3*STYPE_i + \beta_4*SNUTRI_i$ | | | | |
| Adjusted R-squared | 0.7332 | | | |

**Table 4. 2SLS method estimates of parameters of the system of Equations 13 and 14 for the Cobb-Douglas Model.**

| | | | | |
|---|---|---|---|---|
| System: BIOM_M_TSLS | | | | |
| Estimation Method: Two-Stage Least Squares – Cobb-Douglas | | | | |
| | Coefficient | Std. Error | t-Statistic | Prob. |
| $\alpha_0$ | 7.281629 | 0.674133 | 10.80147 | 0.0000 |
| $\alpha_1$ | 0.929339 | 0.197365 | 4.708722 | 0.0000 |
| $\alpha_2$ | -0.607428 | 0.326855 | -1.858404 | 0.0667 |
| $\beta_0$ | 0.279958 | 0.372497 | 0.751573 | 0.4544 |
| $\beta_1$ | 0.157397 | 0.032251 | 4.880376 | 0.0000 |
| $\beta_2$ | 0.294073 | 0.040109 | 7.331749 | 0.0000 |
| $\beta_3$ | -0.483041 | 0.135289 | -3.570434 | 0.0006 |
| $\beta_4$ | 0.005454 | 0.021222 | 0.256995 | 0.7978 |
| | | | | |
| Determinant residual covariance | | 0.026485 | | |
| | | | | |
| Equation 13: $\ln(SBIOMF_i) = \alpha_0 + \alpha_1*\ln(SSPECIES_i) + \alpha_2*\ln(STYPE_i)$ | | | | |
| Observations: 46 | | | | |
| R-squared | 0.441443 | Mean dependent var | | 9.517919 |
| Adjusted R-squared | 0.415464 | S.D. dependent var | | 0.848863 |
| S.E. of regression | 0.648998 | Sum squared resid | | 18.11154 |
| Prob(F-statistic) | 1.682823 | | | |
| | | | | |
| Equation 14: $\ln(SSPECIES_i) = \beta_0 + \beta_1*\ln(SSAL_i) + \beta_2*\ln(SSHRLN_i) + \beta_3*\ln(STYPE_i) + \beta_4*\ln(SNUTRI_i)$ | | | | |
| Observations: 45 | | | | |
| R-squared | 0.741723 | Mean dependent var | | 2.865228 |
| Adjusted R-squared | 0.715896 | S.D. dependent var | | 0.516789 |
| S.E. of regression | 0.275456 | Sum squared resid | | 3.035039 |
| Prob(F-statistic) | 1.274492 | | | |

Equation 14 measures the effects of estuaries ecosystem compositional and structural attributes on fish species abundance, and explained more than 70% of the variation in fish species. Shallow-sub-tidal habitat (*$SSHRLN_i$*), estuarine type (*$STYPE_i$*) and the nutrient capacity (*$SNUTRI_i$*) of the estuary were all highly statistically significant variables. This means that larger estuaries with larger shallow-sub-tidal flat habitats, higher degrees of openness and larger nutrient capacities would accommodate a richer diversity of fish species. Salinity showed a strong positive correlation with species diversity and was significant at the 99.9% confidence limit.

To derive marginal effects of above estuarine ecosystem services on fish catches one should bear in mind the direct and intermediate effects specified in equations system 13 and 14. These are calculated in Table 5 below based on parameter estimates given in Table 3. It is clear that the degree of openness and species diversity contribute highest values to fish productivity. These estimates can be used to derive shadow values (accounting prices) of the estuary intermediate and regulating ecosystem services at fish catch prices. They however, likely overestimate the value of the marginal contribution of these services. This is because this model, due to lack of appropriate corresponding data, did not account for the effect of economic efforts (inputs) on harvest. Thus the results can be used as some upper bound estimate.

**Table 5. Marginal impacts of estuaries ecological structure on fish catches in KZN.**

| Ecosystem attribute | Unit | Direct effect | Intermediate effect | Total effect |
|---|---|---|---|---|
| Species diversity (SSPECIES) | # species | 661 ($\alpha_1$) | | ($\alpha_1$) |
| Openness (STYPE) | Whitfield's Index | 5,964 ($\alpha_2$) | 8.317 ($\beta_3$) | ($\alpha_2$) + ($\alpha_1$)*($\beta_3$) |
| Shoreline length (SSHRLN) | Meters | | 0.0004 ($\beta_2$) | ($\alpha_1$)* ($\beta_2$) |
| Salinity (SSAL) | mS/cm | | 0.313 ($\beta_1$) | ($\alpha_1$)* ($\beta_1$) |
| Nutritional capacity (SNUTRI) | ha/m$^3$ | | 0.0176 ($\beta_4$) | ($\alpha_1$)* ($\beta_4$) |

## 4.5 Discussion

The combined analysis of the production and demand functions estimated above provide important insights into the role of estuarine regulating services in fish production. Analysis of the data mined from ecological monitoring studies of Begg (1978, 1984) and Harrison (2000) provided empirical evidence of the significant role that habitat, salinity regulation and nutrient cycling plays in supporting diversity of fish species. In turn, diversity of fish species and habitat plays a significant role in producing fish and thus securing estuarine fish stocks.

A very significant result is the empirical evidence that relates fish species diversity to fish stock. The estimation results for Equation 13 showed that higher fish species diversity produced larger fish stocks. This highly significant empirical relationship further enables the analysis of effects of changes in other estuarine attributes on economic activities dependent on fish stocks.

Various development hazards may put ecosystem attributes at risk. Table 6 below simulates the direct and indirect effect of changes in estuarine attributes, that may results from various development hazards, on fish species diversity, biomass and recreational fishing demand.

Estuarine habitat attributes (which can also be thought of as ecological infrastructure), in this case measured by Openness (*STYPE*) and Shoreline length (*SSHRLN*), support fish species diversity through providing a large variety or diversity of habitats, as well as an availability of habitat (space) where detritus may settle. Table 6 below indicates the effects a 1% change in Openness and Shoreline length for the sample of 72 estuaries. Openness has a pronounced effect with a 1% change in Openness causing a 0.62% change in species diversity. Estuarine fish species diversity is therefore highly sensitive to the degree of Openness of the estuaries and any decrease in the total degree of Openness could severely affect species diversity in a particular estuary. This in turn has a very large effect on Biomass (1.16%) and could therefore significantly affect fish production and the economic activities dependent upon it.

Shoreline length has a less pronounced, although still significant effect, with a 1% change in Shoreline length causing a 0.27% change in species diversity, and an indirect effect of 0.19% on Biomass. This indicates the importance of conserving and managing estuarine habitats such as mangroves and sub-tidal sand and mud flats.

Salinity significantly affects fish species diversity through creating a water body habitat for a larger variety of fresh and seawater fish. Euryhaline species also use estuarine water as refuge from marine predators. The variation in Salinity within the estuaries analysed (Coefficient of Variation (CV) = 84%) indicates a wide variation in salinity conditions within the estuarine meta-community. A 1% change in Salinity would cause a 0.21% change in species diversity, and an indirect effect of 0.15% on Biomass. This attribute indicates the importance of managing freshwater inflow into estuaries, combined with maintaining the degree of Openness.

The Begg-Harrison data set analysed here is cross-sectional and do not contain any time-series data. This limits the extent to which the impact of nutrient cycling can be analysed. The data nevertheless enabled the analysis of cross-sectional variations in nutrient capacity through measuring the ratio between catchment area and estuary water body volume. Thus, higher runoff from larger catchment areas, would deposit more detritus, relative to the estuary size. A 1% change in nutritional capacity would cause a 0.03% change in species diversity, and an indirect effect of 0.02% on Biomass. Although this effect is relatively small compared to the other estuarine attributes, it is nevertheless significant. A more thorough analysis of the dynamic effect of nutrient cycling follows in the next chapter.

This analysis demonstrates the importance to the economy of managing impacts on estuarine parameters, of conserving estuarine habitat, of the relationship between estuarine parameters, fish species diversity and fish stock. Such analysis also enables the valuation of changes in these parameters through estimating their effects on economic activity as measured by, for example, TCM and other methods.

**Table 6. This table simulates the direct and indirect effect of marginal changes in estuarine attributes on fish species diversity and biomass.**

| **Dependent variable** | **Sample variation:** Coefficient of Variation (CV) | **Effect of a 1% variation in the dependent variable on:** | |
|---|---|---|---|
| | | Species diversity (SSPEC) | Biomass(SBIOMF) |
| Openness (STYPE) | 36% | 0.62% | 1.16% |
| Shoreline length (SSHRLN) | 156% | 0.27% | 0.19% |
| Salinity (SSAL) | 84% | 0.21% | 0.15% |
| Nutritional capacity (SNUTRI) | 255% | 0.03% | 0.02% |

## 4.6 Conclusion

Cross section data and a static model were used to measure the relationship between fish production and compositional and functional elements of both estuarine and marine ecosystems. This demonstrated how shadow values (accounting prices) may be estimated for the KZN estuarine and marine ecosystems. This has been achieved through estimation of a system of ecological production functions employing the 2SLS regression analysis method. The estimated system gave highly significant statistical performance and generated parameter effects consistent with scientific knowledge. The results provided compelling evidence of the importance of estuarine ecological composition and structure on fish species diversity and fish production.

The results showed that higher fish species diversity produced larger fish stocks. This analysis demonstrated the importance to the economy of prudent management of estuarine ecological parameters, of conserving estuarine habitat, of the relationship between estuarine parameters, fish species diversity and fish stock biomass. Such analysis also enables the valuation of changes in these parameters through estimating their effects on economic activity as measured by various ecosystem valuation methods.

# 5. Chapter 5: Bio-economic fishery production model for analysis of the dynamic interactions between marine ecosystems and the fish production

## 5.1 Overview

Whereas the previous chapter investigated the effects of compositional and structural estuarine attributes, using cross-section data and a static modelling approach, this chapter investigates key functional estuarine attribute of nutrient cycling. It does this using time series data and therefore enables the development of a dynamic modelling approach. This chapter therefore develops and estimates an economic-ecological fish production function to model the combined effects of both ecosystem attributes and economic efforts (inputs) on fish biomass and harvest in the case study area. The chapter uses available time series data containing data on economic efforts, marine line-fish harvests and relevant marine ecosystem attributes to implement the dynamic specification of the fishery model of Equations 8, 9 and 12 of the analytical framework described in Chapter 3.

The following section discusses types and sources of the used data and develops the empirical model to be employed for implementing the dynamic fishery model. Section three presents and discusses results of the data sources and empirical analyses. The estimated dynamic bio-economic model parameters are then used to perform in-sample simulation (i.e. recover historical trend records for model validation) of changes of studied fishery stocks (section four) and scenario analyses of effects of possible changes in selected components of the estimated system (section five). Section six concludes the chapter and distils implications of the study.

## 5.2 The KZN line-fishery and supporting estuarine ecosystem attributes

The KZN commercial line-fishery consists of about 100 vessels and is responsible for about 5% of the total South African line fish catch. This boat-based line-fishery focuses on two major fishing areas. The first is the narrow zone of scattered reefs that extends along much of the coast and the second, the deeper reefs to the south of Durban and to the north of the Thukela River in a depth of 100 to 200 m. The majority of vessels used in the KZN commercial line-fishery are fibreglass, mono-hull or catamaran ski-boats ranging in length from 5 to 10m and powered by outboard motors. Vessels use echo sounders and Global Positioning System devices to locate reefs and fish. Most of these vessels are launched through the surf and there are only a few larger, harbour-based vessels, which operate from Durban or Richards Bay. There are no fish processing factories in the province and the entire catch is sold as fresh or frozen both locally and in the hinterland (Crafford *et. al.* 2008).

The KZN line-fishing industry is dependent upon large endemic reef fish such as *Seventy-four*, *Red Steenbras* and *Rockcods*; smaller sparids such as *Slinger*, *Santer* and *Trawl Soldier*. Shoaling migrants (*Geelbek*, *Dusky Kob* and *King Mackerel*) have become increasingly important to such an extent that years of good catches are characterized by strong migrations of these fish species, rather than increased catches of resident reef fish.

There is ample evidence in the literature that nutrient levels strongly influence fish stock (i.e. fish biomass and biological production). The KZN coastline is characterised as an oligotrophic, or nutrient deficient, marine ecosystem. Thus fresh water runoff from rivers is considered an important source of nutrient loading into estuaries and the marine environment (Lamberth, Turpie 2003; Turpie, Lamberth 2010; Whitfield 1998). This fresh water system therefore plays an important functional biodiversity role in connecting terrestrial and estuarine system processes and components to marine-based species dynamics.

### 5.3 Data sources and the empirical model specification

The system of Equations 8 and 9 specifies the dynamic path of the fishery as a function of changes in coastal ecosystem (CE) ecological attributes and consequent changes in fish production (harvest) and welfare given economic efforts applied at the time, i.e. independent of knowledge of (data on) X. Another data challenge relates to measuring the effects of CE attributes on the carrying capacity of the fishery K specified in Equation 7.

The transformation of Equation 8 allows for estimation of the dynamic fishery model parameters independent of data on X. While this also allows for an indirect derivation of an estimate of the ecological effects parameter α of Equation 8, it does not specify an actual functional relationship for the ecological effects. Coastal wetland extent measured as mangrove area was used in Barbier (2007) to determine CE effects on fish biomass through the system carrying capacity factor K in Equation 7 above. Our study had access to an alternative data set that allows estimation of ecological effects' parameters. The said data provides time series records of fish egg species abundance and absolute egg counts collected by Dr Allan Connell since 1987. This consists of more than two decades of records, in 215 separate species data sheets, of fishes with pelagic eggs on the inshore shelf (within 5km of the KZN coastline (Connell, 2012). Annual spawning period and egg species abundance and total egg count trends are provided, as well as egg and larval descriptions. This activity is part of an ongoing research effort to increase the annual trend graphics for each species and to gather barcodes of currently unidentified eggs and larvae, so that they will be identified when adult material has been sequenced (the fish egg database of the website: http://fisheggs-and-larvae.saiab.ac.za/). Fish eggs and larvae from each sample are collected by hand, counted and identified on a weekly basis. The number of eggs and larvae collected in each sample is used in our study as an index of fish egg species abundance (count of species) as well as a good proxy to measure of fish biomass X (count of total number of eggs). In particular, for the purposes of this study, the total number of fish eggs sampled, i.e. the total egg count (*STEC*) was used as an indicator of the density or biomass of fish eggs. In addition, the total number of species sampled, i.e. number fish egg species counted (*SSEC*) was used as an indicator of species abundance or diversity of fish eggs.

Nutrient influx is a key attribute of coastal ecosystem function influencing fish biomass productivity, particularly on the KZN coast, which is nutrient-depleted and heavily reliant on nutrient influx from terrestrial runoff (Connell, 2012). This is supported by strong empirical evidence from recent research suggesting that nutrient influx into this system is closely tied to annual variations in runoff (Whitfield, 1998; Lamberth and Turpie, 2003; Turpie and Lamberth, 2010). We can accordingly describe the relationship between the functional attributes of the KZN coast ecosystem and productivity of its fisheries by Equation 10:

$$S_t = f(SN_t(g(SAR_t))) \qquad (10)$$

Where $SN_t$ is an indicator of nutrient load in the fish production system and $SAR_t$ denotes annual runoff (*SAR*). Data on *SAR* into the KZN coast was also collected as part of the same fish egg data base described above which allowed estimation of the relation between nutrient influx, fish species abundance and fish biomass (using the fish egg measures as proxies).

Our empirical model of the CE dependent dynamic fish production system (Equations 7, 8 and 9) along the KZN coast is accordingly reduced to the following system:

$$E_t = \theta_1 H_{t-1} + \theta_2 E_{t-1} + \varepsilon_t \qquad (15)$$

$$(C_t - C_{t-1})/C_{t-1} = \beta_0 + \beta_1 (C_{t-1}/lnS_{t-1}) + \beta_2 E_{t-1} + \mu_t \qquad (16)$$

$$S_t = \gamma_0 + \gamma_1 SAR_t + \omega_t \qquad (17)$$

Where $E_t$ measures effort in crew hours per month (number of crew x number of hours fished) for vessel engaged in line-fishing in period t; $H_t$ measures total catch in Kilograms per month (kg/month) in period t, $(C_t - C_{t-1})/C_{t-1}$ was calculated from the catch per unit effort (kg/hour/month) data in periods t and t-1, respectively; $(C_{t-1}/lnS_{t-n})$ was calculated using catch per unit effort data in period t-n and our measure of $S_{t-n}$ (measures of fish eggs and larvae); $SAR_t$ measures mean annual runoff in million cubic meters; $\theta_1$, $\theta_2$, $\beta_0$, $\beta_1$, $\beta_2$, $\gamma_0$ and $\gamma_0$ are model parameters and $\varepsilon_t$, $\mu_t$, and $\omega_t$ are residual error terms. As mentioned earlier, values of parameters of the structure of the original dynamic fishery model can be recovered from estimates of the coefficients of above empirical system of equations (15-17) according to the following correspondences:

$\varphi P = \theta_1$; $(1-\varphi w) = \theta_2$; $r = \beta_0$; $(r/q\alpha) = -\beta_1$; $q = -\beta_2$; hence $\alpha = -(r/q\beta_1) = (\beta_0/\beta_2\beta_1)$

The system is estimated using data on commercial line-fishing efforts *E*, their catch *H* and catch per unit effort collected by the Department of Agriculture, Forestry and

Fisheries (DAFF 2012) combined with above described data on fish egg biomass and annual runoff data.

Equation 17 in the above analytical framework, describing the relationship between freshwater runoff and fish egg measures, was estimated using the above described nutrient influx data and counts of both fish egg species abundance (diversity) and fish egg stock (density) as indicators of condition, diversity and biomass of those fish species that spawn in the area. Whilst it is recognised that the condition, biomass and species abundance of the spawner biomass as well as seasonal variables will influence spawning intensity[5], counts of fish egg species is expected to be influenced by *AR* in the current year ($SAR_t$), as nutrient inflow indirectly promotes spawning (refer to Equations 13 and 14 and Tables 3 and 4 above). Nutrient inflow promotes primary productivity, which will increase the biomass of zooplankton grazers benefitting from the algal blooms. The zooplankton and algal mix form a good food source for grazing fish, such as anchovy, sardine and other small shoaling species. As these are the key spawners, one would thus expect a lag between high nutrient input (rainfall) and high egg production about a year later. One can expect a further lag before the predatory fish, feeding on these small shoaling species, will also contribute to the higher spawning levels. In addition, one can expect a relationship between fish egg species diversity and fish egg stock (biomass) (refer to Equations 13 and 14 and Tables 3 and 4 above).

### 5.4 Empirical model estimation results

This study estimated extended versions of the model in Equation 17 to account for lagged *SAR* effects on the alternative available index of the fish biomass $STEC_t$ (i.e. fish egg density). The following specification of the relationship was empirically tested:

$$STEC_t = c_0 + c_1 SAR_t + c_2 SAR_{t-1} + c_3 SAR_{t-2} \quad (18)$$

---

[5] Others have referred to the two attributes egg species abundance and egg count together, as “spawning intensity”.

The empirical relationship between runoff and fish biomass (or fish egg density) was estimated by testing linear and Cobb Douglas functional forms. The linear form function gave the best statistical performance, results of which are presented in Table 7.

The results show that nutrient input into the system in the current period t, combined with lagged effects of nutrient input over preceding two years (t-1 and t-2) explains 97% of the variation in $STEC_t$ (Table 7). However, effect of a one period lag ($SAR_{t-1}$) was not statistically significant and hence estimation results for only the statistically significant two periods lag are reported in Table 8. Both coefficients indicate a strong positive correlation between runoff and fish egg biomass or density (STEC). The lagged effect is consistent with the life cycle characteristics of fish, which reach spawning maturity after 12-24 months. It is to be noted that improved condition of the spawning adults in the population, resulting from improved nutrition, contributes to the increased spawning.

**Table 7. Effect of Runoff on fish production estimation results (Equation 18).**

| Dependent Variable: $STEC_t$ | | | | |
|---|---|---|---|---|
| | Coefficient | Std. Error | t-Statistic | Prob. |
| $SAR_t$ | 0.215091 | 0.089188 | 2.411666 | 0.0949 |
| $SAR_{t-1}$ | 8.150709 | 4.976418 | 1.637867 | 0.2000 |
| $SAR_{t-2}$ | 11.40421 | 4.191028 | 2.721102 | 0.0725 |
| C | -4805.102 | 1206.367 | -3.983117 | 0.0283 |
| | | | | |
| R-squared | 0.969937 | Mean dependent var | | 513.1857 |
| Adjusted R-squared | 0.939874 | S.D. dependent var | | 346.0199 |
| S.E. of regression | 84.84651 | Akaike info criterion | | 12.01512 |
| Sum squared resid | 21596.79 | Schwarz criterion | | 11.98422 |
| Log likelihood | -38.05293 | Hannan-Quinn criter. | | 11.63310 |
| F-statistic | 32.26321 | Durbin-Watson stat | | 2.179820 |
| Prob(F-statistic) | 0.008769 | | | |

**Table 8. Estimation results of SAR influences on fish production in the coast of KZN (Equation 18).**

| Dependent Variable: $STEC_t$ | | | | |
|---|---|---|---|---|
| | Coefficient | Std. Error | t-Statistic | Prob. |
| $SAR_t$ | 0.414260 | 0.074884 | 5.532028 | 0.0026 |
| $SAR_{t-2}$ | 11.28793 | 3.412231 | 3.308080 | 0.0213 |
| C | -2890.046 | 854.1805 | -3.383414 | 0.0196 |
| | | | | |
| R-squared | 0.927397 | Mean dependent var | | 594.5500 |
| Adjusted R-squared | 0.898356 | S.D. dependent var | | 394.4448 |
| S.E. of regression | 125.7556 | Akaike info criterion | | 12.78655 |
| Sum squared resid | 79072.32 | Schwarz criterion | | 12.81634 |
| Log likelihood | -48.14621 | Hannan-Quinn criter. | | 12.58563 |
| F-statistic | 31.93388 | Durbin-Watson stat | | 1.375046 |
| Prob(F-statistic) | 0.001420 | | | |

Ordinary least squares (OLS) regression procedure was then used to estimate the ecological and economic effects parameters of the above system of equations under the assumption that both $X_{t-1}$ and $E_{t-1}$ (hence $C_{t-1}$) are predetermined (Homans and Wilen, 1997). To allow for possible cross-equation correlations the study employed both single equation and systems of equations econometric estimation procedures to measure above model parameters. The Zellner (1962) Seemingly Unrelated Regression Equations (SURE) procedure estimates of the system of Equations 15-17 gave better statistical performance and hence the study reports and discusses hereunder (Table 9) the results of the SURE procedure.

Integrating Equation 17 into the dynamic systems model comprising Equations 15 and 16, as Table 9 displays, all variables showed very high statistical significance (1% and 5%).

Both the variables $SSEC_t$ and $STEC_t$, indicators of diversity and functionality respectively, were integrated into the system through employing a "log$SSEC_t$ + log $STEC_t$" identity in Equation 16. $SSEC_t$ represented the number fish egg species counted, i.e. the total number of species sampled, and which was used as an indicator of species abundance or diversity of fish eggs; whereas $STEC_t$ represented the total number of fish eggs sampled, i.e. the total egg count which was used as an indicator of the density or biomass of fish eggs.

The effect of "$SSEC_{t-2}$ + $STEC_{t-2}$" was statistically significant at 1% when estimated using the actual value instead of the log transformation but the sign of a key structural parameter (the measure of carrying capacity $\alpha$ in Equation 7) changed to an infeasible negative value. With the "log $SSEC_{t-2}$ + log $STEC_{t-2}$" transform the study was able to recover all structural parameters of the system with expected signs (Table 9). These results further indicated a 2-period (t-2) lagged effect, which corresponded with a 24 month life cycle to fish eggs reaching maturity.

These results demonstrate that both species abundance (i.e. diversity as measured by total number of species, SSEC) and biomass (i.e. density as measured by total count of eggs, STEC) play roles in the production system. It is further expected that a relationship exists between SSEC and STEC, as evidenced by Chapter 4 above. Equation 13, using the cross-sectional estuary data (refer to Table 4 above), demonstrated a significant relationship between SSPECIES and SBIOMF, indicators of estuarine fish species diversity and fish biomass (or density) respectively. Following from Equation 13, the relationship between SSEC and STEC was also tested:

$$STEC_{t-2} = \pi_1 * SSEC_{t-2} \qquad (19)$$

Equation 19 (Table 9) demonstrates a significant relationship (0.01%) between fish egg species abundance or diversity (SSEC) and fish egg biomass or density (STEC). This mirrors the results of Equations 13 and 14 (Tables 3 and 5). This demonstrates the complexity of the system and the existence of feedback loops within the system.

Results of the analysis of the relationship between runoff and fish egg biomass (density) show a strong effect of nutrient loading into the system in the current period $SAR_{t-2}$ ($\gamma_1$ in Table 9). Following from the exploratory analyses of this relationship reported above (Tables 7 and 8), a one period lag ($SAR_{t-3}$) however did not show statistical significance and hence estimation results for the statistically significant two periods lag ($SAR_{t-4}$) are reported ($\gamma_2$ in Table 9). Both coefficients indicate a strong positive correlation between runoff and fish egg biomass (density).

Estimation results indicate that data supports the theoretical assumptions and that in addition to historical catch and economic efforts, functional (nutrients input) and compositional (in this case measured by egg and larvae species diversity and biomass) attributes of the KZN estuarine and coastal ecosystems are also good key predictors of expected fishing catch and stocks and overall marine fishery system performance.

**Table 9. SURE estimation results**

Estimation Method: Seemingly Unrelated Regression
Sample: 2 324

| | Coefficient | Std. Error | t-Statistic | Prob. |
|---|---|---|---|---|
| $\theta_1$ | 0.200961 | 0.077909 | 2.579445 | 0.0100 |
| $\theta_2$ | 0.793936 | 0.065898 | 12.04802 | 0.0000 |
| $\beta_0$ | 0.193217 | 0.027099 | 7.129934 | 0.0000 |
| $\beta_1$ | -1.352582 | 0.426061 | -3.174616 | 0.0015 |
| $\beta_2$ | -3.47E-06 | 4.78E-07 | -7.265850 | 0.0000 |
| $\gamma_0$ | 58.87929 | 18.31693 | 3.214474 | 0.0013 |
| $\gamma_1$ | 0.251999 | 0.011833 | 21.29689 | 0.0000 |
| $\gamma_2$ | 0.074657 | 0.008312 | 8.981464 | 0.0000 |
| $\pi_1$ | 0.025026 | 0.000395 | 63.43195 | 0.0000 |
| Determinant residual covariance | | 4.95E+15 | | |

Equation: 15: $E_t = \theta_1 * H_{t-1} + \theta_2 * E_{t-1}$
Observations: 323

| | | | |
|---|---|---|---|
| R-squared | 0.598000 | Mean dependent var | 29911.27 |
| Adjusted R-squared | 0.596747 | S.D. dependent var | 14968.05 |
| S.E. of regression | 9505.042 | Sum squared resid | 2.90E+10 |
| Durbin-Watson stat | 2.517065 | | |

Equation16: $(c_t - c_{t-1})/c_{t-1} = \beta_0 + \beta_1 * (c_{t-1}/(\log STEC_{t-2} + \log SSEC_{t-2})) + \beta_{2*} E_{t-1}$
Observations: 252

| | | | |
|---|---|---|---|
| R-squared | 0.183234 | Mean dependent var | 0.018626 |
| Adjusted R-squared | 0.176674 | S.D. dependent var | 0.137990 |
| S.E. of regression | 0.125208 | Sum squared resid | 3.903600 |
| Durbin-Watson stat | 0.319912 | | |

Equation18: $STEC_{t-2} = \gamma_0 + \gamma_1 * SAR_{t-2} + \gamma_2 * SAR_{t-4}$
Observations: 252

| | | | |
|---|---|---|---|
| R-squared | 0.756159 | Mean dependent var | 513.8190 |
| Adjusted R-squared | 0.754200 | S.D. dependent var | 296.1913 |
| S.E. of regression | 146.8463 | Sum squared resid | 5369398. |
| Durbin-Watson stat | 0.164414 | | |

Equation 19: $STEC_{t-2} = \pi_1 * SSEC_{t-2}$
Observations: 252

| | | | |
|---|---|---|---|
| R-squared | 0.874545 | Mean dependent var | 1471.381 |
| Adjusted R-squared | 0.874545 | S.D. dependent var | 1561.046 |
| S.E. of regression | 552.9157 | Sum squared resid | 76734656 |
| Durbin-Watson stat | 0.099878 | | |

## 5.5 Discussion

This chapter adapted a bio-economic fishery model to measure the contribution of regulating and supporting services of the coastal ecosystems in KZN to the dynamics and productivity of its coastal fisheries.

Results of the econometric specification of the model parameters showed good statistical fit to the data and supported the study hypotheses on the importance and strong contribution of the tested ecological attributes (diversity and nutrients loading) to productivity and dynamics of the studied fishery system in addition to economic efforts and ecological factors.

The relationship between fish egg biomass and fish egg species diversity showed a strong, positive, direction correlation. These two measures together can be described as spawning intensity and it is clear that a high spawning intensity is associated with both high fish egg biomass and fish egg species diversity counts. It is also to be noted that improved condition of the spawning adults in the population, resulting from improved nutrition, contributes to the increased spawning (Crafford and Hassan, 2014). The relationship between river runoff and fish egg biomass (density) showed the strong effect of nutrient loading into the system in the current period as well as its lagged effect. The lagged effect results from good spawning intensity (i.e. high values of fish egg biomass and fish egg species count) in a 2-year period lag, which indicates that the larger number of spawning adults in the current period resulted not only from fish attracted to the nutrient input in the system in the current period, but also from the larger number of eggs in the lagged period that matured in the current period.

The inclusion of the fish egg indicators in equation 16 resulted in an improved biological production identity and demonstrates that increased nutrient inputs indirectly affects economic behaviour as increased spawning intensity (as measured by STEC and SSEC), positively increases changes in catch per unit effort, which in turn increases fish harvest. However, it is a recommendation from this study that equation 16 could be further improved.

Estimation results indicate that data supports the theoretical assumptions and that in addition to historical catch and economic efforts, functional (nutrients input and fish egg biomass) and compositional (in this case measured by egg and larvae species abundance) attributes of the KZN estuarine and coastal ecosystems are also good key predictors of expected fishing catch and stocks and overall marine fishery system performance.

These results can be used to estimate system parameters to generate estimates of stocks of this fishery over the years, which was lacking and hence provides basis for construction of resource accounts for this ecosystem and its assets. Parameter estimates were also used to simulate system performance through in-sample forecasting, which revealed very useful information for better policy and management of this resource. Parameter estimates generated were used to recover the structural parameters of the system described in Equations 15 to 17, which are reported in Table 10. Estimates of these parameters were used to simulate the structural model to compute estimates of the KZN coastal fishery stocks ($X_t$) given observed values of SAR (nutrient loading), the external driver of this system specification (determining the proxy of diversity measured in abundance of egg and larvae as well as fish egg biomass) and consequently the coastal fishery carrying capacity (K) and consequent system biological balances.

**Table 10. Structural parameters of the KZN coast fishery**

| Structural parameter | Symbol | Estimating formula | Value |
|---|---|---|---|
| Intrinsic growth rate | R | $\beta_0$ | 0.193217 |
| Catch ability coefficient | Q | $-\beta_2$ | 3.47E-06 |
| Estuaries ES attributes effect on carrying capacity (K) | $\alpha$ | $-(R/Qb_1)=-(b_0/b_2b_1)$ | 41,167 |
| Carrying capacity | $K_t$ | $\alpha$*log ($S_t$) | Vary by t |

**Table 11. Harvesting and empirical Maximum Sustainable Yield (MSY) for 5 cycles in the KZN fishery over the 20-year period 1990-2009.**

| Period | | Harvest (tons/month) | Empirical MSY (tons/month) |
|---|---|---|---|
| P1 | 1990-1994 | 21 | -2 |
| P2 | 1995-1999 | 15 | 33 |
| P3 | 1999-2002 | 41 | -5 |
| P4 | 2002-2004 | 28 | 75 |
| P5 | 2004-2009 | 35 | 31 |

Figure 3 and Table 11 depict the current situation of KZN coast fishery indicating periods of over-fishing and declining trend of fish stocks (1990-1994, 1999-2002, 2004-2009) and periods of stock recovery (1995-1999, 2002-2004).

Fishing pressure measured as harvest efforts, exceeded the maximum sustainable biological Yield (MSY) of the fishery severely during 1990-1994 and during 1999-2002, and to a lesser extent during 2004-2009, driving fish stock down on a declining trend (Figure 3). These declining stock periods are associated with elevated harvests combined with lower rainfall (and thus AR). The converse is true for the periods of stock recovery (1995-1999 and 2002-2004). When this data is analysed over the long run data period (1990-2009), harvest was 86%-89% of MSY, indicating a sustainably fished system.

This analysis demonstrates that this fishery system is resilient and has, over the past 20 years, been able to recover from short-term periods of stress resulting from both over-fishing and natural variations in nutrient deposition.

Performance of the system was also simulated under a change in AR scenario, assuming a 10% reduction of AR scenario with climate change (Figure 4). This was implemented by simulating a 10% reduction in the rainfall trend of the past 10 years (1999 – 2009) for a forecast period from 2011 to 2018. Impact of such reduction of nutrient inputs on the system works through egg and larvae abundance and biomass biology determining systems carrying capacity and productivity. The reduction in AR produced an impact of a 2.6% reduction in the system's carrying capacity lowering its MSY from about 32.5 to 29.5 Mt per month. The results also indicate that a disinvestment will take place from fisheries through reduced fishing effort as a result of reduction in harvest.

This indicates the importance of not only managing the system through managing fishing effort, but also monitoring nutrient inputs into the system and managing their flows and the allowable fishing efforts within the sustainable biological limits of the system to prevent possible collapse of this fishery.

## 5.6 Conclusion

This chapter used time-series data and a dynamic approach adapting a bio-economic fishery model to measure the contribution of regulating and supporting services of the coastal ecosystems in KZN to the dynamics and productivity of its coastal fisheries. A generic fishery bio-economic specification was extended to establish an explicit link between the ecological composition (biodiversity as measured by fish egg species abundance) and functional (nutrient supply and biomass) attributes of coastal ecosystems and productivity of the coastal fishery (i.e. harvest of the final service, fish biomass). In this dynamic case, the carrying capacity of the coastal fishery stock dynamics was specified to be a function of species density (egg abundance and composition/diversity). The same data set also contained records of runoff flows into the KZN coast, which was used as a proxy indicator of nutrients input into this CE and the major driver of species density, thus indirectly influencing carrying capacity and productivity of the fishery through the species composition variable. This data was combined with available time series on economic efforts and catches of the commercial line-fishing industry along the KZN coast to estimate the structural parameters of the developed dynamic fishery bio-economic model. In the dynamic case, the study used estimated system parameters to generate estimates of stocks of this fishery over the years, which was lacking and hence provides basis for construction of resource accounts for this ecosystem and its assets. In-sample simulation indicates that current long-run fishing efforts and harvest rates fall within the MSY parameters and that effort and harvests are sustainable. However, within a 20-year cycle, the results demonstrated several periods of declining fish stocks resulting from short periods of elevated harvests combined with lower rainfall (and thus AR), followed by periods of recovery.

The study has also simulated out-of-sample system performance under a future scenario predicting potential climate change influences on runoff to predict consequent impacts on the fishery system under study. The simulation of futures suggests that a potential 10% reduction in AR will increase the pressure on the fishery carrying capacity leading to faster declines in stocks of up to 4% by 2020, accompanied by a contraction in the fisheries industry of more than 50% as fishing effort is reduced due to reduced productivity levels.

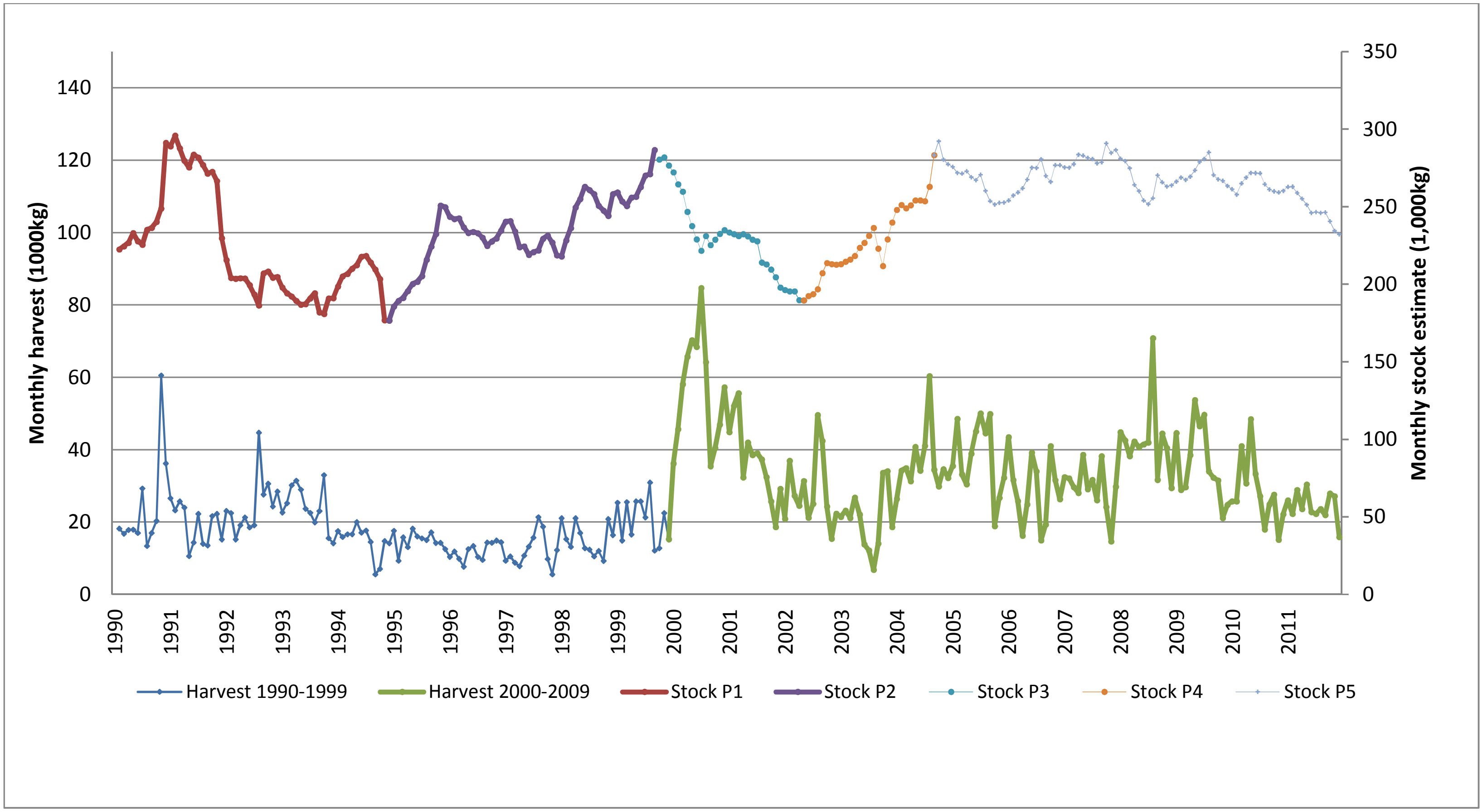


**Figure 3. Stock levels and harvest volumes of the KZN line-fishery for a 20 year period from 1990 to 2011. Declining trends of fish stocks (1990-1994, 1999-2002, 2004-2011) and periods of stock recovery (1995-1999, 2002-2004) are observed. Periods of stock decline are associated with elevated harvests combined with lower rainfall (and SAR). The converse is true for the periods of stock recovery (1995-1999 and (2002-2004).**

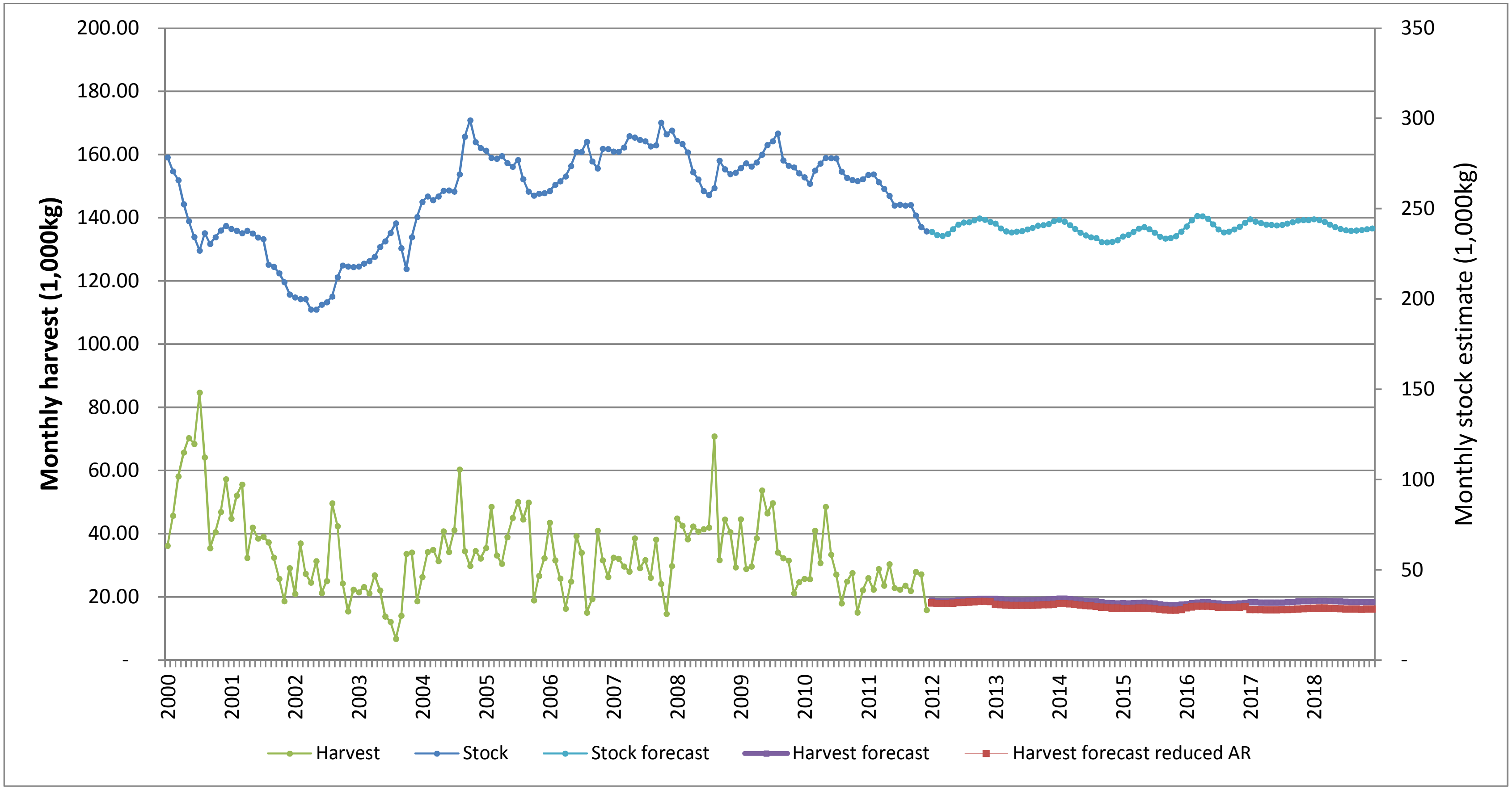


**Figure 4. Performance of the system was also simulated under a change in SAR scenario, anticipating 10% reduction of SAR scenario with climate change. The reduction in SAR produced an impact of a 2.6% reduction in the system's carrying capacity by 2020. The results further indicate that a disinvestment will take place from fisheries through reduced fishing effort as a result of reduced harvests.**

# 6. Chapter 6: Summary, conclusions and implications of the study

The MEA has been described as a radical new approach to the analysis of the interface of ecology and economics (MEA 2005, MEA 2007). However, the MEA did not attempt a comprehensive and systematic 'total' valuation of ecosystem services, because the judgment was that the theory, methods and data sources were insufficiently developed to support a credible effort of that sort (Kinzig *et. al.* 2007). Thus the outputs of the MEA (and TEEB) challenge economists to develop methods for defining the consequences of ecological change induced by current economic activity; methods for analyzing the distribution of possible outcomes of alternative activities and, the probabilities attached to those outcomes; and methods for developing appropriate mitigating or adaptive policies. This research is an attempt to respond to these challenges.

## 6.1 Using ecological production function approaches to analyse complex ecological-economic interactions

The study applied the MEA ecosystem services framework using the concepts of composition, structure, and function of an ecosystem to formulate and estimate production functions linking ecological infrastructure and biodiversity to the economy employing appropriate data and tools of empirical analysis. This required a case study of a suitably bounded system. The estuarine and marine ecosystems of KZN provided an appropriate case study opportunity, where empirical data collected through scientifically credible means was available for implementing the intended analyses. The work also demonstrated the importance of describing chains of causality, through expert-based development of hypotheses of how ecosystems function and what compositional and structural components are of importance.

The research applied the production function methodology to integrate, in one analytical framework, a multiplicity of economic and ecological attribute indicators. Economic indicators included economic output parameters (in this case fish catch), economic effort indicators (e.g. catch effort and travel cost) and various factors of production (e.g. labour and ecosystem inputs). Ecological indicators measured the ecosystem attributes that drive the ecological system including variables that describe the composition, diversity and functionality of biodiversity.

The econometric analysis applied multiple regression analysis of cross section and time series data for estimating economic-ecological production functions to test hypotheses advanced based on existing scientific knowledge about the construct of causality chains. The production function methodology allowed for the analysis of the system in a way that the marginal impacts of potential changes or shocks in the system may be assessed. Regulating services analysed here included various habitat services, salinity regulation and nutrient cycling. The significant effects of these services on fish species diversity and fish biomass or stock have been empirically investigated and measured.

The research thus demonstrated, based on conclusions drawn from the analyses reported in Chapters 4 and 5, how ecological production functions' analysis can support and confirm empirically expert-based system hypotheses. It also showed how useful is combining into one analytical framework components of the coupled social and ecological systems to explain how various attributes of ecosystems interact with economic activity in the production of benefits to human society employing appropriate ecological and economic data sets.

## 6.2 Dealing with data limitations

The study applied econometric analyses, a technique that allows for evidence-based analysis of observed data, based on existing scientific knowledge. The work demonstrated that existing scientific databases may contain useful evidence of relationships between ecological infrastructure and the economy, and that decisions need not always wait for the results of controlled experiments.

Data employed to implement the intended analyses were sourced from cross-section data mined from large-scale ecosystems assessments undertaken in the 1980's (hard copy format) and early 2000's (electronic format), in addition to time-series data on AR monitored by government agencies, time series data on fish eggs collected by an individual researcher, and time series data on fish catch and effort collected by the Department of Agriculture, Forestry and Fisheries (DAFF 2012).

By combining these data with existing scientific knowledge, this study developed and estimated empirical ecological production functions to measure such relationships in key KZN fisheries along the east coast of South Africa, for the estuarine and marine systems separately. The study has therefore demonstrated usefulness of existing databases for carrying such empirical investigations in search of evidence of critical structural linkages between ecological infrastructure and functional attributes (e.g. regulating services) and the economy.

### 6.3 New systems knowledge and insights into risks to ecosystems

The most important contribution of this work is the new knowledge on the empirical relationships between ecosystems attributes and fish production, which allowed for estimation of potential economic impacts of changes in regulating ecosystem attributes.

Chapter 4 used cross section data and a static model to measure the relationship between fish production and compositional and functional elements of both estuarine and marine ecosystems. It demonstrated how shadow values (accounting prices) may be estimated for the KZN estuarine and marine ecosystems. This has been achieved through estimation of a system of ecological production functions employing the SURE regression analysis method. The estimated system gave highly significant statistical performance and generated parameter effects consistent with scientific knowledge. The results provided compelling evidence of the importance of estuarine ecological composition and structure on fish species diversity and fish production.

The results showed that higher fish species diversity produced larger fish stocks. This analysis demonstrated the importance to the economy of prudent management of estuarine ecological parameters, of conserving estuarine habitat, of the relationship between estuarine parameters, fish species diversity and fish stock biomass. Such analysis also enables the valuation of changes in these parameters through estimating their effects on economic activity as measured by various ecosystem valuation methods.

Chapter 5 used time-series data and a dynamic approach adapting a bio-economic fishery model to measure the contribution of regulating and supporting services of the coastal ecosystems in KZN to the dynamics and productivity of its coastal fisheries. The adapted model is based on a well-known dynamic fishery bio-economics formulation that has been widely used to study fishery systems. In this study however, the generic fishery bio-economic specification was extended to establish an explicit link between the ecological composition (biodiversity as measured by fish egg species abundance) and functional (nutrient supply and biomass) attributes of coastal ecosystems and productivity of the coastal fishery (i.e. harvest of the final service, fish biomass). Other applications in the literature measured the contribution of other ecosystem attributes to fishery systems' dynamics and productivity (e.g. physical infrastructure such as habitat quality, extent of coastal wetland and mangrove area). The present study measured the contribution of other coastal ecosystems' attributes such as freshwater flows as the main source of nutrients for primary production supporting key compositional elements (biomass and diversity) and important underlying ecological processes influencing fish production.

In the dynamic case, the carrying capacity of the coastal fishery stock dynamics was specified to be a function of species density. The same data set also contained records of runoff flows into the KZN coast, which was used as a proxy indicator of nutrients input into this CE and the major driver of species density (biomass and diversity/composition), thus indirectly influencing carrying capacity and productivity of the fishery through the species composition variable. This data was combined with available time series on economic efforts and catches of the commercial line-fishing industry along the KZN coast to estimate the structural parameters of the developed dynamic fishery bio-economic model.

In both the static and the dynamic modelling, results of the econometric specification gave good statistical fit to the data and supported the study hypotheses on the importance and strong contribution of the tested ecological attributes (species diversity, habitat structure and extent, salinity and nutrient loading) for productivity and dynamics of the studied fishery system in addition to economic efforts and ecological factors.

In the dynamic case, the study used estimated system parameters to generate estimates of stocks of this fishery over the years, which was lacking and hence provides basis for construction of resource accounts for this ecosystem and its assets. Parameter estimates were also used to simulate system performance through in-sample forecasting, which revealed very useful information for better policy and management of this resource. For example the in-sample simulation indicates that current long-run fishing efforts and harvest rates fall within the MSY parameters and that effort and harvests are sustainable. However, within a 20-year cycle, the results demonstrated several periods of declining fish stocks resulting from short periods of elevated harvests combined with lower rainfall (and thus AR), followed by periods of recovery.

This study provided further confirmation of the strong linkage between nutrient levels (using SAR as a proxy) and fish egg abundance and thus enables the investigation of runoff and rainfall related climate change effects on KZN fisheries. The study has also simulated out-of-sample system performance under a future scenario predicting potential climate change influences on runoff to predict consequent impacts on the fishery system under study. The simulation of futures suggests that a potential 10% reduction in AR will increase the pressure on the fishery carrying capacity leading to faster declines in stocks of up to 4% by 2020, accompanied by a contraction in the fisheries industry of more than 50% as fishing effort is reduced due to reduced productivity levels. Hazards to runoff, other than climate change, may have similar effects and would have to be considered in all major coastal development projects.

### 6.4 The “insurance value” of ecosystems

This study provided important insights into ecosystem resilience, through the empirical evidence provided on the risk buffering effect of species diversity on fish abundance. Resilience is primarily determined by the amount of disturbance the system can absorb and still remains in the same state; the degree to which the system is capable of self-organisation versus organisation forced by external factors; and the degree to which the system can build and increase the capacity for learning and adaptation (Carpenter et. al. 2001).

This study proved empirically that fish stock biomass is a function of fish species diversity. Fish species diversity in turn is a function of a variety of functional biodiversity attributes such as habitat, nutrient cycling and salinity. Thus, maintenance of a wider portfolio of options in estuarine ecosystems is likely to enhance the capacity of the system to respond to shocks and stresses and can thus be interpreted as contributing an “insurance” benefit to human well-being.

### 6.5 Implications for coastal management and policy

Ultimately, the above results are of great importance to estuarine management, harbour development and planning, and various other coastal sustainable development strategies, research agendas and policy formulation.

The work demonstrated the importance of monitoring and managing the key attributes of ecosystems that play an intermediate role the production of final consumption services. The research further demonstrates applications of production functions, in this case study, for planning coastal and terrestrial developments (such as harbours and aquatic impoundments), issuance of fisheries permits, ecosystem monitoring and forecasting the effects of global change.

The results hold several important policy implications as it demonstrates, at the hand of empirical evidence, how ecological degradation and change in estuarine and marine ecosystems may indirectly affect a valuable industry. The results provide statistical evidence of the importance of ecological infrastructure and biodiversity and the fish production systems along the KZN coast. As tension between conservation of biodiversity and economic development is expected to increase in future, in all types of ecosystems, the need for evidence-based policy decisions will become greater.

This research confirms the importance of evidence-based policy analysis for better decision-making and policy design. Where large development needs may increase tension between conservation of biodiversity and economic development, extreme precautionary and risk-averse environmental policy is often hard to accept. Decision-makers therefore need better knowledge and clear understanding of how economic and ecological systems interact with sufficient evidence of impacts and appropriate mitigation options and policy interventions.

To this end one key recommendation is to increase investment in systems ecology research to improve our understanding of the complex interplay between social and ecological systems. The production functions specified and tested in this research was premised on systems descriptions performed by estuarine and marine ecologists with deep understanding of systems ecology.

The independent variables used in the production functions further serve as useful indicators of ecosystems change and their impacts. In this case, the research demonstrated the usefulness of monitoring critical a set of attributes such as harvests, species diversity, SAR, extent and type of estuarine habitat parameters, salinity and nutrient levels in order to measure the productivity of estuarine and marine fisheries. Ecosystem service valuation is data intensive, and this is likely to remain a problem into the future. However, this research demonstrated that existing databases may provide useful evidence of key structural relationships between ecological infrastructure (particularly regulating ES) and the economy, and that decisions need not always wait for the results of controlled experiments. This serves to affirm the critical role and benefits of instituting continued investment in systematic data collection and monitoring of such systems. This study therefore strongly supports continued investment in long-term scientific monitoring programmes of fish catches, rainfall and studies such as those performed by scientists such as Begg and Harrison and the invaluable research efforts performed by Dr Allan Connell in this regard.

Another important recommendation relates to the modification of conventional fisheries models' approaches to estimation of fish stocks and management and control of fishing efforts and permits. Conventional fishery models use historical catch records and catch effort data to estimate fish stock sizes and control fishing permit conditions. The dynamic modelling work has demonstrated the usefulness of including environmental variables as additional predictors of fish stocks, in addition to historical catch records and catch effort data. Results of the static modelling work, provided empirical evidence that the cost of recreational angling has a significant impact on the number of trips undertaken and therefore controlling the cost of fish licences may present policy makers with a useful policy instrument, if required. This new knowledge thus introduces the possibility of using environmental variables such as rainfall, nutrient deposition, salinity gradient and various habitat attributes as additional predictors of fish stocks. It is noted however, that this accentuated response to terrestrial based nutrients only applies to fish stocks that are located in areas with significant terrestrial runoff into otherwise oligotrophic waters.

Overall, the study demonstrates the importance of responsible coastal development initiatives that are sensitive to underlying coastal and marine ecosystems' functions. River impoundments, water pollution, harbour development and other coastal developments are all factors that influence fish production and thus the economy. For instance, the research demonstrated the intermediate roles of the ecological structure of estuaries, especially their degree of openness, and the extent of shallow sub-tidal sand and mudflats, which need to be taken into account in planning coastal development projects for mitigating effects on such critical ecosystem services. The concept of mitigation is well defined by the United States Environmental Protection Agency (EPA), which proposes a very useful three-step approach for defining ecosystem impacts, through a "mitigation sequence". The first step is to avoid (adverse impacts to ecosystems are to be avoided) and no impact shall be allowed if practicable alternative(s) with less adverse impact exists. The second step is to minimize (if impacts cannot be avoided), appropriate and practicable steps to minimize adverse impacts must be taken. The third step is to compensate (appropriate and practicable compensatory mitigation is required) for unavoidable adverse impacts which remain. The amount and quality of compensatory mitigation may not substitute for avoiding and minimizing impacts (EPA 2012). Only after all these measures have been exhausted should monetary compensation be considered. It is thus clear that evidence-based, production function analysis of regulating services have the potential to support decision-makers in being responsible in ecosystem management.

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# 8. Appendix 1: Derivation of Equations

## 8.1 Barbier's (2007) Cobb–Douglas derivation for estimating a static habitat-fishery production function model

The production function approach treats an ecological service, such as estuarine habitat (in this case a supporting service), as an input into the economic activity, and like any other input, its value can be equated with its impact on the productivity of any marketed output. Barbier (2007) thus denoted the production function, if H is the harvest of the fishery, as:

$H = h(E_i, S_i)$ (1)

Where $E_i$ are standard inputs of a commercial fishery and where $S_i$ denotes estuarine area, which is independent of H and which may therefore have a direct influence on the marketed fish catch, H. Although this denotation internalises estuarine habitat as an input into the production function, estuarine area is an inadequate indicator of the ecosystem services supplied by estuaries. Estuarine functioning is characterised by a complex set of ecosystem components and processes, which, together, regulate the production of provisioning and cultural services supplied by an estuarine system. The components are, for the most part, physical characteristics of the estuary. The processes relate mainly to the exchange of water, through terrestrial and marine inflows resulting from daily, seasonal and annual tidal, flood and storm events. Productive estuaries generally have high nutrient content, high water body retention times and suitable substrate where nutrients can accumulate and thus feed the benthic and higher species within the system. Equation 1 can thus be rewritten as:

$H = h(E_i, S_i)$ (1.1)

Where $S_i$ are indicators of the species, components and processes that regulate estuarine system functionality.

A standard assumption in most static habitat-fishery models is that production function (1.1) takes the Cobb–Douglas form, $H = AE^aS^b$, where A is a constant, $E$ is an aggregate measure of total effort in the off-shore fishery and $S$ is an aggregate measure of the species, components and processes that regulate estuarine system functionality. The derivation proceeds as follows:

Max (p.H – w.E)

of which duality: Min $L = w.E + \lambda(H - A.E^{a}S^{b})$

$\delta L/\delta E = w - \lambda.A.a.E^{a-1}.S^{b} = 0$

$\delta L/\delta \lambda = H - A.E^{a}.S^{b} = 0$

- ⇨ $E^{a} = H/AS^{b}$
- ⇨ $E = (H/AS^{b})^{(1/a)}$
- ⇨ $\quad = A^{-1/a}.H^{1/a}.S^{-b/a}$
- ⇨ $C^{*} = w.\ A^{-1/a}.H^{1/a}.S^{-b/a}$

Then:

$P = C^{*}/H$ and $P = k.H^{n}$

- ⇨ $C^{*} = P.H = k.H^{n+1} = k.H^{n}.H$
- ⇨ $k.H^{n}.H = w.A^{-1/a}.H^{1/a}.S^{-b/a}$
- ⇨ $(H^{n}.H)/H^{1/a} = w/k.A^{-1/a}.S^{-b/a}$
- ⇨ And $(H^{n}.H)/H^{1/a} = H^{(n+1-1/a)}$
- ⇨ $H = (w/k)^{a/\beta}.A^{-1/\beta}.S^{-b/\beta}$

  where: $n+1-(1/a) = (a(n+1)-1)/a = \beta/a$

  and thus: $\beta = a(n+1)-1$

It follows that the optimal cost function of a cost-minimizing fishery is:

$C^{*} = C(H,w,S) = wA^{-1/a}h^{1/a}S^{-b/a}$

Where $w$ is the unit cost of effort. After Barbier (2007), assuming an iso-elastic market demand function, $P = p(H) = kH^{n}$, $\eta = 1/\varepsilon < 0$ , the market equilibrium for catch of the open access fishery occurs where the total revenues of the fishery just equals cost (or price equals average cost), i.e. $P = C^{*}/H$, which in this model becomes:

$kH^{\eta} = wA^{-1/a}H^{1-a/a}S^{-b/a}$

which can be rearranged to yield the equilibrium level of fish harvest:

$H = [w/K]^{a/\pi}.A^{-1/\pi}\ S^{-b/\pi},\ \pi=(1-\eta)a-1$ (1.2)

It follows from (1.2) that the marginal impact of a change in estuarine system functionality is:

$dh/dS = (-b/\pi).\ [w/k]^{a/\pi}\ .A^{-1/\pi}\ \ S^{-(b+\pi)/\pi}$ (1.3)

The change in consumer surplus, *CS*, resulting from a change in equilibrium harvest levels (from $H_0$ to $H_1$) is:

$\Delta CS = {}_{H0}\int^{H1} p(H).dh - [p^1H^1 - p^0H^0] = \{k[(H^1)^{\eta+1}-(H^0)^{\eta+1}]\}/(\eta+1) - k[(H^1)^{\eta+1}-(H^0)^{\eta+1}]$

$= -\eta[p^1H^1 - p^0H^0]/(\eta+1)$ (1.4)

By utilizing (1.3) and (1.4) it is possible to estimate the new equilibrium harvest and price levels and thus the corresponding changes in consumer surplus associated with a change in estuarine functionality, for a given demand elasticity. This function assumes that the fishery is open access. Any profits in the fishery will attract new entrants until all the profits disappear, and in equilibrium, the welfare change in coastal wetland is in terms of its impact on consumer surplus only. The standard assumption for an open access fishery is that effort next period will adjust in response to the real profits made in the past period (Bjørndal and Conrad 1987).

However, in fish stock production functions, the magnitude of the fish stock is a significant variable, and thus valuing changes in terms of the impacts on current harvest and market outcomes alone is flawed.

In order to address this weakness, dynamic models of coastal habitat-fishery linkages, which incorporate the change in estuarine system functionality within a multi-period harvesting model of the fishery, are required (Barbier 2007).

The dynamic modeling starts with defining $X_t$ as the stock of fish measured in biomass units, and where any net change in growth of this stock over time can be represented as:

$X_t - X_{t-1} = f(X_{t-1}; S_{t-1}) - h(X_{t-1}; E_{t-1});\ \delta f/\delta X^2_{t-1}>0;\ \delta f/\delta S_{t-1}>0$ (2)

Thus, net expansion in the fish stock occurs as a result of biological growth in the current period, $f(X_{t-1}, S_{t-1})$, net of any harvesting, $h(X_{t-1}, E_{t-1})$, which is a function of the stock as well as fishing effort, $E_{t-1}$. The influence of the estuarine ecosystem extent and functionality, $S_{t-1}$, as a nursery habitat on growth of the fish stock is assumed to be positive, $\partial f/\partial S_{t-1} > 0$, as an increase and improvement in estuarine habitat will mean more carrying capacity for the fishery and thus greater biological growth.

Let $p(h)$ represent landed fish price per unit harvested, $w$ the unit cost of effort and $\varphi > 0$ the adjustment coefficient, then the fishing effort adjustment equation is:

$$E_t - E_{t-1} = \varphi[p(H_{t-1})h(X_{t-1};E_{t-1}) - wE_{t-1}];\ \delta p(H_{t-1})/\delta H_{t-1} < 0 \qquad (3)$$

Assume a conventional bio-economic fishery model with biological growth characterized by a logistic function, $F(X_{t-1}, S_{t-1}) = RX_{t-1}[1 - X_{t-1}/K(S_{t-1})]$, and harvesting by a Schaefer production process, $H_t = qX_tE_t$, where $q$ is a 'catchability' coefficient, R is the intrinsic growth rate and $K(S_t) = \alpha \ln S_t$, is the impact of biomass and estuarine extent and functionality on carrying capacity, $K$, of the fishery. The market demand function for harvested fish is again assumed to be iso-elastic, i.e. $P = p(h) = KH^{\eta}$, $\eta = 1/\varepsilon < 0$. Substituting these expressions into (2) and (3) yields:

$$X_t = r.X_{t-1}.[1-(X_{t-1}/K(S_{t-1}))] - h_{t-1} + X_{t-1} \qquad (7)$$

$$E_t = \varphi R_{t-1} + (1+\varphi w)E_{t-1}\ ;\ R_{t-1} = kh_{t-1}^{1+\eta} \qquad (8)$$

Both $X_t$ and $E_t$ are predetermined, and so (3.1) and (8) can be estimated independently (see Homans and Wilen 1997). Following Schnute (1977), we can define the catch per unit effort as $c_t = H_t/E_t = qX_t$. If $X_t$ is predetermined so is $c_t$. Thus, following Schnute (1977), we can define the catch per unit effort as $c_t = H_t/E_t = qX_t$.

⇨ $X_t = C_t/q$

Which, substituted in (7), gives:

$$C_t/q = R.(C_{t-1}/q).(1-(C_{t-1}/q)/K(S_{t-1})) - H_{t-1} + C_{t-1}/q$$

which can be simplified to give:

$$(C_t - C_{t-1})/C_{t-1} = R - (R/q\alpha).(C_{t-1}/K(S_{t-1})) - qE_{t-1} \qquad (9)$$

Thus Equations (8) and (9) can be estimated independently to determine the biological and economic parameters of the model.