A Decision Tree Framework for Assessing Status of Exploited Marine Ecosystems Under Changing Environmental Conditions

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Thesis presented for the degree of

Doctor of Philosophy

in the Department of Biological Sciences

University of Cape Town

August 2017

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Declaration

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a) Lockerbie, E.M., Shannon, L.J and Jarre, A. 2016. "The Use of Ecological, Fishing and Environmental Indicators in Support of Decision Making in Southern Benguela Fisheries." *Ecological Indicators*, 69: 473-487.

b) Lockerbie, E.M., Coll, M., Shannon, L.J. and Jarre, A. (2017) "The Use of Indicators for Decision Support in Northwestern Mediterranean Sea Fisheries." *Journal of Marine System*, 174: 64-77

c) Lockerbie, E.M., Lynam, C.P., Shannon, L.J. and Jarre, A., (under review) "The Use of a Decision Tree Framework to Support the Ecosystem Approach to Fisheries Management in the North Sea" *ICES Journal of Marine Science*

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Acknowledgements

I am exceedingly grateful to both my supervisors at the University of Cape Town, Dr Lynne Shannon and Associate Professor Astrid Jarre. I would not have gotten through this PhD without you both and can't begin to thank you enough. You have given me unfailing emotional support and I have helped me build up confidence in both myself and my work. Another huge thank you to my wonderful lab. You are the most special people and have made even the hardest days more easy to get through.

And, finally, to my family. My parents, who have done everything they could to support me from halfway around the world. Without the proof reading provided by my Dad the standard of my work would certainly have suffered. Lastly to Wesley Campher. You have been my rock for the last three years, thank you.

List of Acronyms

AMO	Atlantic Multidecadal Oscillation
BCLME	Benguela Current Large Marine Ecosystem
CPR	Continuous Plankton Recorder
EAF	Ecosystem Approach to Fisheries
EBM	Ecosystem Based Management
GES	Good Environmental Status
GFCM	General Fisheries Commission for the Mediterranean
ICES	International Council for the Exploration of the Sea
IEA	Integrated Ecosystem Assessment
IUU catch	Illegal, Unreported and Unregulated Catch
IVI	Intrinsic Vulnerability Index
LPUE	Landings per Unit Effort
MCDA	Multicriteria Decision Analysis
MSFD	Marine Strategy Framework Directive
MTI	Marine Trophic Index
NAO	North Atlantic Oscillation
NRF	National Research Foundation
NW	Northwestern
OHI	Ocean Health Index
PICES	North Pacific Marine Science Organisation
SAHP system	South Atlantic High Pressure System
SST	Sea Surface Temperature
TL	Trophic Level
TLmc	Trophic Level of Modelled Community
TLsc	Trophic Level of Surveyed Community
WeMO	Western Mediterranean Oscillation

Abstract

The removal of marine species through fishing has impacted marine ecosystems for thousands of years. The pressure of exploitation on marine ecosystems has now reached a point at which there is serious concern over ecosystem well-being on a global scale. There has, therefore, been a global move towards an ecosystem approach to fisheries management. The objective of this study was to develop a decision tree framework to assess the status of exploited marine ecosystems, which could be successfully applied to numerous ecosystems and guide decision support under changing conditions.

This work was based on that of the IndiSeas project, which makes use of indicators designed to detect the impacts of fishing on marine ecosystem around the world. A suite of indicators, selected from those utilised in the IndiSeas project, was divided into ecological and fishing pressure indicators. Ecosystem specific suites of environmental indicators were also included, allowing the framework to ascertain the impacts of environmental variability on ecosystem components. This is an important addition as currently many assessments of the impacts of fisheries do not account for the influence of the environment. The framework was developed for the Southern Benguela ecosystem and then applied, with minor adjustments to account for ecosystem-specific characteristics, to the South Catalan Sea and North Sea.

Indicator time series were analysed making use of linear regressions, resulting in the assignment of a score between one and five, depending on the direction and significance of trends. Data series were divided into distinct periods based on known environmental changes or shifts within ecosystems. Careful consideration was given as to whether fishing and environmental indicator trends could explain the observed trends in ecological indicators. A method of score adjustment was then developed to account for the impacts of both fishing

and environmental variability on ecological indicators. Correlations were conducted to detect potential redundancies of ecological indicators and weightings were applied to decrease the contribution of correlated indicators to overall ecosystem trends. However, as correlations differed between indicators and amongst ecosystems, it was necessary to adjust the applied weightings for individual ecosystems.

Results for the Southern Benguela classified the ecosystem as *neither improving nor deteriorating* during Period 1 (1978-1993) and Period 2 (1994-2003). During Period 3 (2004-2010) the ecosystem was classified as *possibly improving*. The South Catalan Sea was classified as *possibly deteriorating* during Period 1 (1978-1990) and *neither improving nor deteriorating* during Period 2 (1991-2010). The North Sea ecosystem was classified as *neither improving nor deteriorating* during Period 1 (1983-1992). During the second (1993-2003) and third (2004-2010) periods the ecosystem was categorised as *possibly improving*.

When assessing fisheries impacts at an ecosystem scale there are typically high levels of uncertainty. However, this thesis concluded that the development of a scoring and weighting system, alongside the addition of environmental drivers and the inclusion of expert knowledge throughout the applications of this framework, has allowed the developed decision tree framework to successfully categorise the three ecosystems. It is anticipated that the knowledge that this framework will add to current methods of generating advice for fisheries management will aid future decision support within these ecosystems.

Chapter One

Introduction

1.1. Introduction

Marine ecosystems have been influenced by anthropogenic forcing for thousands of years (e.g. Jackson *et al.*, 2001; Griffiths *et al.*, 2004; Lotze and Milewski, 2004), and pressure has grown to an extent that there is serious concern over the future health of ecosystems on a global scale (e.g. Jackson *et al.*, 2001; Coll *et al.*, 2008; Halpern *et al.*, 2008). Marine ecosystems represent an important source of goods and services, such as food provisioning, raw materials, climate regulation and nutrient cycling (Beaumont *et al.*, 2007). Consequently the need for sustainable management of fisheries has been recognised internationally in terms of food security, conservation of biodiversity and also the preservation livelihoods and the very existence of coastal communities (Ommer *et al.*, 2009). Ecosystems supporting the world's fisheries are now often degraded, and fish populations within them are frequently fully or over exploited (FAO, 2004; Worm *et al.*, 2009; FAO, 2011). To date, fisheries management has, in general, been ineffective (Pickitch *et al.*, 2004). The focus has been on maximising catches and managing fisheries based only on assessed target species or stocks, treating exploited species as though they are isolated from predators, prey and the environment. However, it is now widely accepted that the impacts of fishing on entire ecosystems should be considered in order for management strategies to be effective (FAO, 2003; Garcia and Cochrane, 2005; Garcia, 2010). There has, therefore, been a move towards ecosystem-based management of fisheries. This aims to protect ecosystems as a whole, avoiding degradation of ecosystems and considering several factors, including; the impacts of

fisheries on both target and non-target species, trophic interactions between species, habitat destruction, and the socio-economic aspects of fisheries (FAO, 2003; Pickitch *et al.*, 2004).

Alongside the effects of fishing on marine ecosystems, the impacts of climate change and long-term variability are becoming increasingly important as marine climate change is occurring at an alarming rate in many ecosystems around the world (e.g. Hoegh-Guldberg *et al.*, 2007; Hoegh-Gulderberg and Bruno, 2010; Schofield *et al.*, 2010). Variability is common within the ocean climate system and results in phenomena such as the El Niño Southern Oscillation, the North Atlantic Oscillation, the Pacific Decadal Oscillation and the Western Mediterranean Oscillation (Alheit and Bakun, 2010). Alongside natural variability, anthropogenic forcing also influences marine ecosystems. One of the most studied drivers is the increase in global temperatures. Temperatures around the world have risen at a rate of around 0.2°C per decade over the past 30 years as a result of increasing concentrations of atmospheric greenhouse gasses (Hansen *et al.*, 2006). The oceans are responsible for absorbing this excess energy, resulting in an increase in the heat content of the upper ocean (IPCC, 2007; Levitus *et al.*, 2009).

Climate change and variability are key factors in understanding the past and predicting the future of marine biodiversity. Species richness in many ecosystems is strongly linked to environmental variables (Macpherson, 2002; Cheung *et al.*, 2009) and such variables also have the potential to impact commercial fisheries (e.g. Roessig *et al.*, 2004; Worm *et al.*, 2006). Yet despite numerous studies on the impacts of long-term climatic variability on marine ecosystems (Walther *et al.*, 2002; Harley *et al.*, 2006; Hoegh-Gulderberg and Bruno, 2010), it is still often not considered in the assessment of fisheries. To effectively manage fisheries, it is important to consider the combined impacts of fishing and environmental forcing at all levels of the ecosystem.

1.2. The Impacts of Fishing on Marine Ecosystems

Humans directly impact marine ecosystems in a variety of ways, including the removal of organisms through fishing, the building of marine structures (e.g. oil rigs and aquaculture installations), dredging and pollution, to name just a few. However, for the purpose of this thesis the removal of selected sizes and species of fish from ecosystems via fishing activity is the focus.

Fishing can both directly and indirectly impact marine ecosystems, including increased mortality, alterations to competitive interactions, decreased prey/predators and reduced productivity, to name just a few. It is unsurprising, therefore, that fishing has the potential to influence entire ecosystems, altering both structure and functioning (Pauly *et al.*, 1998; Worm *et al.*, 2006). Within all marine areas there has been a general decline in the trophic level of landed fish. Trophic level identifies the position of an organism within food webs, through the identification of the source of energy for each organism (Lindeman, 1942; Odum and Heald, 1975). In practical application in fisheries ecology these values range from 1 (as defined for plants and detritus) to 5, although this is rare even in large fishes (Pauly and Palomares, 2000). A global decline in mean trophic level from 3.3. to 3.1 was observed between the 1950s and 1994, and a further steady decline has been observed since the 1970s (Pauly *et al.*, 1998). The removal of key species, both target and non-target, from an ecosystem can impact the numerous other species with which they interact. While removing large numbers of individuals from target stocks has clear consequences for the stocks themselves through increased mortality, there are also numerous consequences for dependent predator and prey species (Pickitch *et al.*, 2004). The depletion of stocks can have serious consequences for marine ecosystems, such as trophic cascades (e.g. Frank *et al.*, 2005; Daskalov *et al.*, 2007), species extinctions (e.g. Casey and Myers, 1998; Dulvy *et al.*, 2003) and ecosystem regime shifts (e.g. Jackson *et al.*, 2001; Möllmann and Diekmann, 2012)

The exploitation of both high and low trophic level species can alter marine ecosystems (Jennings and Kaiser, 1998; Smith *et al.*, 2011). Low trophic level species are usually found in high abundance, forming dense schools, and some of these species are highly targeted by fisheries. For example, small pelagic forage fish account for over 30% of global landings (Alder *et al.*, 2008). These small pelagic fish play an important role in many ecosystems, particularly eastern boundary upwelling systems, providing the major means of energy transfer from plankton up to top predators and are suggested to exert a wasp-waist control on some ecosystems (Cury *et al.*, 2000; Cury *et al.*, 2003). In wasp-waist control a small number of highly abundant species can influence the abundance of both higher and lower trophic level species.

Studies have found widespread impacts of the removal of low trophic level species, including negative impacts on marine mammals, sea birds and commercially important species (Cury *et al.*, 2011; Smith *et al.*, 2011; McCauley *et al.*, 2015). However, the extent of the impacts of removing low trophic level species has been observed to vary greatly between different marine ecosystems, and can depend on trophic interactions within ecosystems (Smith *et al.*, 2011).

There has been a strong decline in the abundance of large predators within marine ecosystems, including mammals, large teleost fish and sharks. Disturbingly, it has been estimate that around 90% of large predatory fish have now been lost from marine ecosystems (Baum *et al.*, 2003; Myers and Worm, 2003; Myers *et al.*, 2007). The removal of high trophic level predatory species can result in a phenomenon known as 'fishing down the food web' (see Pauly *et al.*, 1998). In this phenomenon a reduction in fisheries for high trophic level species, due to overexploitation of stocks, results in a subsequent increase in fisheries targeting lower trophic level species. The removal of high trophic level species can also result in a release of lower trophic level species from predation pressure, allowing the abundance of

these species to increase rapidly. Such overexploitation of ecosystems has been observed to cause shifts in ecosystem state. Non-target species may also be heavily impacted by fisheries, particularly through bycatch, discards and incidental catch, which are not always managed (e.g. Alverson, 1994; Lewison *et al.*, 2004; Bellido *et al.*, 2011).

The size-selective nature of fishing typically results in the removal of large individuals and species from ecosystems (Zhou *et al.*, 2010). Hence, fishing pressure has influenced body size and size distributions within populations of marine organisms (e.g. Shin *et al.*, 2005). Size-selective fishing techniques have resulted in a decrease in the maximum size of landed fish on a global scale over the last century (Hsieh *et al.*, 2010; Planque *et al.*, 2010).

Fishing can also have serious consequences for marine habitats. Several types of fishing gear directly influence marine habitats, with impacts including scraping, scouring and the resuspension of the substrate. The extent of such impacts varies depending on the type of habitat that is being fished. For example, the impacts of fishing on coral reefs may be much more conspicuous than the impacts of fishing gear on sediments with short-lived or temporary inhabitants (Jennings and Kaiser, 1998). Alongside this, habitats can be negatively impacted if fishing pressure influences those organisms responsible for habitat structure. One of the most marked examples of this can be observed in coral reefs, where fishing can drastically change community structure. Within reef systems fishing pressure can result in a change from a system dominated by herbivorous species (such as parrot fish which are often targeted) to one dominated by corallivorous species (such as sea urchins which are rarely targeted) (Jennings and Kaiser, 1998), can result in reefs being eroded faster than they can be accreted (Hughes *et al.*, 2003).

Another important element to consider when studying the impacts of fishing on marine ecosystems is that of food security. This is an international concern, principally in regions

such as South Asia and Sub-Saharan Africa (Garcia and Rosenberg, 2010), as fisheries

provide an important source of protein in low-income and rural areas. This has resulted in the

issue of sustainable exploitation receiving substantial public attention in recent decades. The

challenge of maintaining and restoring the resource potential of fisheries is further

complicated when considered in the context of changing marine ecosystems (Garcia and

Rosenberg, 2010). Alongside this, overfishing can have massive impacts on economies, with

the World Bank and the FAO estimating that excess fishing could cost the world around $50

billion a year in net economic losses (World Bank & FAO, 2009). By failing to preserve the

state of the world's oceans we risk the loss not only of the revenue from fisheries, but also of

historical sources of sustenance and employment (Srinivasan *et al.*, 2010).

1.3. The Impacts of Climate Variability on Marine Ecosystems

The oceans form a major component of the global climate system (e.g. Barnett *et al.*, 2001).

Although in many ecosystems fishing is considered the greatest threat to marine organisms, it

is important to note that fishing and climate variability interact in a number of ways and

therefore should not be treated separately (e.g. Brander, 2007; Brander, 2009). Ocean climate

variability often results in significant changes to both the physical and ecological dynamics of

the marine environment (Brander, 2007). In the context of fisheries there is concern that

future changing climates will have significant consequences for fisheries production (Cheung

et al., 2016). Climate can, therefore, exert additional pressure on marine organisms, acting in

conjunction with pressures such as exploitation, pollution, and introduced species.

Current knowledge on the impacts of climate change on marine ecosystems is much less than

that on their terrestrial counterparts. In general this is due to the complex nature of marine

ecosystems and the relative difficulty in collecting appropriate measurements within the

marine environment (Hoegh-Gulderberg and Bruno, 2010). The impact of environmental

forcing further complicates the ability to understand complex biological processes and

consequently a better understanding of such forcing is becoming increasingly important (e.g.

Stenseth *et al.*, 2002; Blanchard *et al.*, 2012; Jochum *et al.*, 2012; Jarre *et al.*, 2015).

Environmental changes are known to increase uncertainties in assessing and predicting the

state of fish stocks, and the influences of such changes can be detected at all levels of

biological organisation (Brander, 2009; Fréon *et al.*, 2009). Many relationships between

biological processes and environmental variability are indirect and may involve many

variables or a large number of biological components within ecosystems (Brander, 2009;

Checkley Jr *et al.*, 2009). Therefore, given the multitude of possible direct and indirect

processes which can potentially influence biological components, it can be an immense task

to identify those actually responsible for observed changes. Understanding the impacts of

climate on total fisheries production can be particularly challenging at ecosystem scales,

where physiological preferences, life histories and interspecific interactions can vary and are

often too poorly understood to make accurate predictions (Blanchard *et al.*, 2012). Current

knowledge on all components within ecosystems is limited. This further impedes the ability

of fisheries scientists and managers to predict potential changes in fisheries production

resulting from climate variability (Blanchard *et al.*, 2012; Jochum *et al.*, 2012). For example,

suboptimal environmental conditions can result in decreased foraging, growth and fecundity,

as well as influencing migratory behaviour (Barton and Barton, 1987; Donaldson, 1990). All

of these aspects have possible consequences for fisheries production and management.

Marine ecosystems change on a variety of time scales, with much variation resulting from

atmospheric and climate forcing and related processes (Lehodey *et al.*, 2006). For the purpose

of this study, due to the length of periods for which data are available for the considered

ecosystems, climate variability rather than long-term climate change is analysed. As multiple

ecosystems are considered, each of which is driven by different environmental conditions, environmental indicators must be specifically selected based on knowledge of specific environmental drivers within each particular region.

The complex, and sometimes synergistic, effects of exploitation and environmental variability that propagate through food webs frequently contribute to failures in fisheries management (Hilborn *et al.*, 2003). The idea that the impacts of fisheries and climate can be separated is still common and studies often do not consider the fact that exploitation can change the way in which fish populations respond to environmental forcing (Planque *et al.*, 2010). The interactions between exploitation and climatic impacts could be further complicated by the fact that individuals and populations may act to somewhat modulate environmental variability (Planque *et al.*, 2010; Travers-Trolet *et al.*, 2014).

While examples of the impacts of climate on fish populations have been well documented, the impacts of climate variability can also be observed in fishing practices (Planque *et al.*, 2010). An example of this can be seen in the cod fisheries of Labrador and Newfoundland throughout the 1980s and 1990s. During this time persistent cold water temperatures led to slow growth and poor condition in cod, known as 'skinny cod' (Drinkwater, 2002), a change that was believed to be a result of poor prey availability in the cold conditions. This led to an illegal process known as 'high-grading', through which poor condition fish were dumped overboard in favour of retaining large individuals (Kulka, 1997). Average ages of fish thrown overboard declined over this period resulting in many dumped fish being first time spawners. In this case a combination of the direct impacts of climate change, which resulted in slow growth and poor condition of cod, alongside the effects of fishing (in this case high-grading) which resulted in a truncated age structure, exerted additional stress to an already stressed stock (Drinkwater, 2002).

The interactions of fishing pressure and climate change can also act to alter the buffering capacity of populations. Longevity, reproductive strategy and overlapping generations are some of the adaptations employed by fish communities to cope with environmental variability (Murphy, 1967; Longhurst, 2002; Secor, 2007). Fish populations with a greater number of year classes are therefore capable of surviving longer periods of adverse conditions and low recruitment. In contrast to this, a population with fewer cohorts may be at greater risk of collapsing before favourable conditions return (Fromentin and Fonteneau, 2001).

The selective removal, usually of large individuals, through the actions of fishing can result in changes to age structure, such as truncation. This may act to increase coupling between climate fluctuations and recruitment variations (Hsieh *et al.*, 2006; Hsieh *et al.*, 2010). The effects on populations can be substantial, as larger and older individuals are generally more fertile and are known to produce larger eggs and more viable larvae (Cardinale and Arrhenius, 2000; Berkeley *et al.*, 2004b). Larger, older females are also known to exhibit longer spawning periods than smaller, younger individuals (e.g. Secor, 2007) and generally spawn over larger areas. When truncation of a population's age structure occurs, it can increase the dependence of eggs and larvae on a specific set of environmental conditions, thus tightening the link between recruitment and the environment. Examples of such scenarios have been observed in several cod stocks, where environment-recruitment relationships seem to have tightened under low stock biomasses (Brander, 2005).

 It is likely that, under current fishing pressure trends, many ecosystems will become more vulnerable to both internal and external forcing (Hsieh *et al.*, 2006; Tittensor *et al.*, 2009). A greater variety of responses to climate variability may therefore become visible in ecosystems around the world. While many species may suffer under future conditions, some species will unsurprisingly benefit and potentially thrive (e.g. Richardson *et al.*, 2009). The complexity of

the interactions between marine organisms/ecosystems and climate forcing makes it unrealistic to separate climate-induced and fishing-induced effects, but rather calls for future studies to consider their interactions and combined effects (Brander, 2007). In this study it was therefore decided to include the impacts of the most important indicators of environmental variability for each ecosystem when analysing ecological indicator trends with respect to fishing pressure.

1.4. An Ecosystem Approach to Fisheries and the IndiSeas Project

The primary goal of most current fisheries management strategies is to ensure fishing is maintained at a level allowing fisheries to produce sustained annual yields (FAO, 2011). Effective management should reduce the number of overexploited stocks, whilst increasing the sustainable exploitation of non-exploited species in order to maintain the contribution of fisheries to global food sustainability. Despite significant efforts, concern of the unsustainable nature of marine fisheries has been growing since the 1980s (Garcia, 2010). Typically, methods of fisheries management have been based on the assessments of only targeted stocks. However, it is now commonly accepted that such management methods are inadequate. The majority of management failures have been attributed to ineffective governance based on inadequately communicated or insufficient science, that fails to take in account multi-species, ecosystem effects and climatic impacts (Cury *et al.*, 2005).

With the addition of specifications to the Code of Conduct for Responsible Fisheries (see FAO, 1995), the Ecosystem Approach to Fisheries (EAF) was formalised in 2001 (FAO, 2003). Several other terminologies exist which also relate to the management of fisheries and ecosystems, including ecosystem-based fisheries management (EBFM), an ecosystem approach to fisheries management (EAFM) and ecosystem based management (EBM). While

these concepts are closely related, there are some key differences between them (see Garcia, 2003). However, the EAF, which is not limited to management and could also cover development, food safety and planning, has been selected for use in this study due to the international recognition it has received. For example, at the 2001 Reykjavik conference the importance of an EAF was acknowledged by the 47 countries which participated, with all but two countries issuing declarations to make an effort to "reinforce responsible and sustainable fisheries in the marine ecosystem" (FAO, 2001; Cochrane *et al.*, 2004). As the aim of the development of this framework is its successful application to marine ecosystems around the worlds, it is highly important that concepts included in the approach are recognised on a global scale. The EAF also incorporates the human component, as well as the various interactions between humans and natural sub-systems. As we increase our understanding of these linkages they will be included future modifications of this framework.

The FAO (2003) defines the EAF as an approach which "strives to balance diverse societal objectives, by taking account of the knowledge and uncertainties about biotic, abiotic and human components of an ecosystem and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries". Yet the application of such an approach is not without difficulties. Possibly the greatest problem associated with implementing an EAF is the formulation of agreements between scientists and the many stakeholders involved within the fishing industry on what strategic action to take. Potential conflicts include resistance of existing rights holders to reduced quotas, conflicts between various fishing sectors, and conflicts with other marine sectors (e.g. mining) (Cochrane *et al.*, 2004).

An EAF emphasises the interactions between fisheries, target resources and other ecosystem components. It is necessary to acknowledge the impacts of these interactions on target resources and the fact that alterations to the ecosystem can influence its restoration and

productivity (Cochrane *et al.*, 2004). Several other complications must also be considered when attempting to formulate and apply an EAF. Flows within ecosystems can be controlled in a number of ways: by top predator feeding behaviour (top-down control), by primary producers (bottom-up control) or by numerically abundant mid trophic level species (wasp-waist control) (Cury *et al.*, 2003). It is also possible for flows to change as a result of both fishing and environmental variability. Alongside this, ecosystem structure, function and species composition are also known to change both seasonally and annually (Garcia, 2003), further complicating the application of an EAF.

Progress towards the implementation of an EAF has been aided by the selection of applicable indicators which are capable of communicating ecosystem changes. Trends in these indicators can be used to assess the state of marine ecosystems, and hence assist in the management of marine resources. The IndiSeas project (Shin and Shannon, 2010; www.indiseas.org) is aimed at working towards implementing an EAF in marine ecosystems around the world, making use of suites of robust indicators. The working group was established in 2005 under the auspices of the Eur-Ocean European Network of Excellence to look at "EAF Indicators: a comparative approach across ecosystems". The goal of IndiSeas was not to create new indicators but rather to select the most representative, practically achievable and meaningful set of indicators from those previously proposed. These indicators are used to measure and tract the ecosystem effects of fishing. A number of ecological indicators were assembled, examined and reviewed (Shin and Shannon, 2010; Shin *et al*., 2010a; Shin *et al*., 2012), a process which involved several selection criteria and extensive discussion. The selection criteria were inspired by Rice and Rochet (2005) and four selection criteria were adapted for use in the IndiSeas project: (i) that indicators have a theoretical basis/ecological significance, (ii) that indicators be sensitive in order to track ecosystem

change, (iii) that indicators be measured/estimated on a regular basis and (iv) the indicator and its link to fishing must be widely and intuitively understood.

An important feature of selected indicators is the survey-based nature of the majority of data, making the data fishery-independent. This contrasts to similar studies in which meta-analyses have been based largely on model-derived or catch-based indicators (e.g. Alder and Pauly, 2008; Halpern *et al.*, 2008; Pitcher *et al.*, 2009). The IndiSeas Project has relied strongly on multi-institutional collaboration, permitting the sharing of scientific data and, in particular, scientific diagnoses based on local expertise on each ecosystem studied (Shin *et al.*, 2010b). The inclusion of knowledge from ecosystem experts is one of the key features of the IndiSeas project. Regional experts have been included in the IndiSeas project since its inception, and were involved in the selection of indicators. These experts provide local data, inform about any biases in collection and can better interpret specific, local indicator trends (Shin *et al.*, 2010b; Shin *et al.*, 2012). This is particularly important in the IndiSeas project as the inclusion of expert knowledge ensures that global comparisons of marine ecosystems are robust and meaningful (Shin *et al.*, 2012).

One of the key aims of the IndiSeas project is to make use of indicators in a comparative context, evaluating changes across numerous ecosystems over recent decades (Shin and Shannon, 2010). Such an approach provides an opportunity to draw generalizations about ecosystem responses, allowing a better understanding of the broader ecosystem perspective and thereby potentially aiding the implementation of an EAF (Blanchard *et al.*, 2010). Several studies included in the IndiSeas project have therefore adopted a comparative approach (e.g. Coll *et al.*, 2009; Blanchard *et al.*, 2010; Bundy *et al.*, 2010). These studies evaluated the responses of 19 ecosystems, including upwelling, high latitude, temperate and tropical ecosystems (Shin and Shannon, 2010). Comparative studies may be based on single species (e.g. Brander, 1994; Drinkwater, 2005), whole ecosystems (e.g. Moloney *et al.*, 2005;

Bundy *et al.*, 2009) or ecosystem indicators (e.g. Pitcher *et al.*, 2009; Shannon *et al.*, 2009b). Comparisons of similar ecosystems (upwelling and comparable) have been carried out by Shannon *et al.* (2010) and rankings according to ecosystem type by Coll *et al.* (2010). The line of thinking behind the comparative approach is that comparisons amongst a wide range of ecosystems can operate as *ad hoc* replicate responses, highlighting common features, key differences and important species responses, as well as giving insight into the variety of pressures influencing ecosystem processes (Blanchard *et al.*, 2010).

The comparisons of ecosystems with varying levels of exploitation can help determine the relative state of each ecosystem (e.g. Blanchard *et al.*, 2010; Bundy *et al.*, 2010; Coll *et al.*, 2010; Shin *et al.*, 2010a). The more ecosystems included in a comparative analysis, covering a wide range of indicators values, the more significant the comparative analysis will be. Thus far the IndiSeas project has analysed and compared 27 marine ecosystems around the world (Coll *et al.*, 2016). This has aided the selection of a robust suite of indicators that are measurable over a wide range of differing conditions (Shin *et al.*, 2012). This approach could allow scientists and stake holders to learn from the mistakes made in degraded ecosystems and also to identify early warning signals, manifesting through declining indicators (Shin and Shannon, 2010). It was for this reason that the decision tree framework developed in this thesis was aimed at using IndiSeas indicators and being applicable to a range of ecosystems throughout the world's oceans.

The present research thus follows on from the work conducted by the IndiSeas project, making use of a suite of ecological and fishing indicators which have been identified as applicable to many marine ecosystems. IndiSeas indicators have been formulated to respond to the impacts of fishing pressure, both increases and decreases. However, it has been suggested (e.g. Walther *et al.*, 2002; Islam and Tanaka, 2004; Bundy *et al.*, 2010, Table 5) that these indicators will also respond to other drivers, such as the environment. To account

for the impacts of environmental variability, a suite of environmental indicators has therefore been added to account for the influence of these drivers on ecosystem components.

1.5. Selected IndiSeas Indicators

The first phase of the IndiSeas project selected a suite of eight indicators, including six trend and two state indicators (see Table 3 in Shin *et al.*, 2010b), that were assessed across nineteen marine ecosystems (Shin *et al.*, 2010a; Shin *et al.*, 2012). These included mean length, mean lifespan, biomass of surveyed species, proportion of predators, inverse fishing pressure and the trophic level of landings. During the second phase of the IndiSeas project a further five ecological indicators were added (Coll *et al.*, 2016); trophic level of the modelled community, trophic level of the surveyed community, landings, the marine trophic index (MTI) and the intrinsic vulnerability index (IVI). Trophic level indicators (trophic level of modelled and surveyed communities and MTI) were added as they capture changes in trophic structure of marine ecosystems (Shannon *et al.*, 2014a). Selected indicators, along with the calculations used to calculate them (equations 1.1-1.10) can be seen below.

1.5.1. Mean Length

Larger fish within a community are generally more fecund and produce more viable eggs than smaller individuals (Longhurst, 1999). These fish are generally the major target of fisheries, and therefore fishing may act to decrease the mean length of fish within a community. The mean length indicator can therefore be used to identify the impacts of fishing on mean length of fish within the ecosystem. The size-selective nature of fishing gear, the high value of large species and the accumulation of the impacts of fishing in large species

over time would all contribute to the expected decrease of this indicator under increased fishing pressure. Change in size structure has the potential to impact entire communities and predator-prey interactions, altering the ecosystem as a whole (Shin *et al.*, 2005).

Calculated as: Mean length $= \sum_i L_i/N \ (cm)$ (1.1)

1.5.2. Mean Lifespan

This indicator acts as a proxy for turnover rate of species and communities within ecosystems, and can reflect their buffering capacities. This indicator is designed to reflect the stability of an ecosystem (Shin *et al.*, 2010b). Lifespan is fixed within species, therefore changes in this indicator reflect the biomass of species with different turnover rates within the ecosystem. In this case changes in mean lifespan are used to assess changes in species compositions within ecosystems. To correctly interpret this indicator it must be considered that changes in species compositions and fluctuations in recruitment may influence trends (Shin *et al.*, 2005).

Calculated as: Mean lifespan $= \sum_S (age_{max} B_S)/\sum_S B_S$ (1.2)

1.5.3. Biomass of Surveyed Species

When exploitation is generalised through the food web a decrease in survey biomass would be expected under increased fishing pressure. However, as species are fished and their biomass is reduced, the biomass of other species may increase due to release from predation pressure or reduced competition. Some caution must therefore be taken in the interpretation of trends observed in this indicator. For example, an increase in biomass, which would be expected under decreasing fishing pressure, may not necessarily reflect a recovery of fished

species. An increase in biomass of small fish can occur when larger fishes are removed and predation pressure is released. However, ecosystem conditions such as habitat and predator-prey interactions will determine to what extent species replacements can occur (Shin *et al.*, 2010b). The removal of predators can be expected to result in an increase in lower trophic level species, resulting in changes to ecosystem productivity. This in turn will have consequences for fisheries as decreased biomass of some species within an ecosystem will influence the productivity of dependent fisheries (Rochet and Trenkel, 2003). This indicator is calculated to trend in the same direction as other IndiSeas indicators, i.e. decreasing trend with increasing fishing pressure.

Calculated as: $Survey\ biomass = B(t)$ (1.3)

1.5.4. Proportion of Predatory Fish

This indicator is a measure of functional diversity within ecosystems. Predatory fish consist of species which are either piscivorous or feed on invertebrates' larger than 2cm. The proportion of predators will impact ecosystem functioning as these species act to regulate the abundance of lower trophic level species as well as dampening the impacts of environmental variability within the ecosystem. A key aim of an EAF should be to restore the abundance of predatory fish within ecosystems (Daskalov, 2008).

Calculated as: Proportion of predatory fish = biomass of predator fish/biomass surveyed (1.4)

1.5.5. Inverse Fishing Pressure

This is a measure of the inverse level of exploitation or total fishing pressure on the ecosystem. Inverse fishing can be considered to measure resource potential (Shin *et al.*,

2010b), identifying the portion of the ecosystem dedicated to fishing. This indicator is inversed in order for trends in this indicator to decrease under increased fishing pressure, in this way hypothetically varying in the same direction as other indicators under fishing pressure. Care must be taken when interpreting this indicator as changes in biomass are not solely related to fishing pressure.

Calculated as: Inverse fishing pressure = B/Y retained species (1.5)

1.5.6. Trophic Level of Landings

The trophic level of landings measures the average trophic level of exploited species within an ecosystem and represents the trophic position of the whole catch. It would be expected to decrease as fisheries typically target larger species, resulting in 'fishing down the food web' (Pauly *et al.*, 1998). This has the potential to change both ecosystem structure and functioning by shortening the length of food chains and releasing lower trophic level species from predation pressure. As the average value of TL per species is considered here this indicator should give an idea of species composition of the catch in terms of trophic positioning.

Calculated as: $TL\ landings = \sum_S (TL_S Y_S)/Y$ (1.6)

1.5.7. Trophic Level of Modelled and Surveyed Communities

Changes in trophic level (TL) occur as a result of the selective removal of fish species. Trophic level of the modelled community spans the entire ecosystem model, excluding zooplankton and primary producers (Coll *et al.*, 2016). Modelled values of biomass are derived from Ecosim models fitted to time series (Shannon *et al.*, 2014a). Ecosim models can

be generated to represent ecosystems and the trophic flows between components (Christensen and Walters, 2004; Shannon *et al.*, 2014a). These models aim to cover entire communities within ecosystems and therefore, due the differences in species included in each of these indicators, it is not uncommon to see differing results when analysing trends in indicators based on different sources (i.e. model-, catch-, survey-based) (Shannon *et al.*, 2014a). Trophic level of the surveyed community is calculated based on all surveyed species, both exploited and non-exploited. The trophic level of the modelled community contains a greater number of species than those that can be collected in surveys, and therefore may pick up trends that the surveyed indicator cannot.

Calculated as: $\text{TLmc} = \sum_{i=1}^{n} B_{Mi} \cdot TL_i / B_{MT}$ (1.7)

$$\text{TLsc} = \sum_{i=1}^{n} B_i \cdot TL_i / B_T \tag{1.8}$$

1.5.8. Landings

Knowledge on trends in landings across ecosystems gives insight into the levels of exploitation exerted on species within ecosystems. This is an indicator of the impacts of fishing pressure on marine ecosystems. While this indicator was not selected for use in the IndiSeas project, Link *et al.* (2010) highlighted the usefulness of including landings as an indicators. Landings are expected to decrease under scenarios of increased fishing pressure as ecosystems become degraded.

1.5.9. Marine Trophic Index

The MTI measures the mean trophic level of all landed species. By identifying the mean trophic level, it is possible to determine the position of fisheries within the food web. This

indicator has a cut off of 3.25 and can therefore be used to identify the relative abundance of more threatened, higher trophic level species within ecosystems (Pauly and Watson, 2005). Identifying trends in the MTI can indicate the mean position of exploited species (with trophic levels > 3.25) within the food web of an ecosystem (Fey-Hofstede and Meesters, 2007). This indicator has been suggested as a more suitable indicator of ocean health than the trophic level of landings, which may also capture the impacts of the environment rather than just fishing (Pauly and Watson, 2005; Butchart *et al.*, 2010; Shannon *et al.*, 2014a).

Calculated as: $\sum_{i=1}^{n} Y_{TL \geq 3.25} \cdot TL_i / Y_{L(TL_i \geq 3.25)}$ (1.9)

1.5.10. Intrinsic Vulnerability Index

The Intrinsic Vulnerability Index (IVI) of species is based on ecological characteristics and life history traits. Generally larger species with higher longevity, higher age at maturity and lower growth rates are more vulnerable to fishing (Cheung *et al.*, 2007), and therefore would have a higher score on the IVI. A decrease in the IVI would, therefore, be expected under increased fishing pressure as larger, more vulnerable species are fished out and lower trophic species are then targeted (Cheung *et al.*, 2007). A decrease in the IVI may therefore suggest a decline in large species within an ecosystem. The IVI in this case is calculated based on all landed species and weighted by the contribution of each species to total landed catch.

Calculated as: $\text{IVI} = \dfrac{\sum_S (age_{max} B_S)}{\sum_S B_S}$ (1.10)

1.5.11. Environmental Indicators

While fishing is considered to be the biggest driver in many marine ecosystems, in some ecosystems environmental drivers can have greater impacts (Mackinson *et al.*, 2009).

Consideration of the impacts of environmental factors was addressed in the second phase of the IndiSeas project. Two simple indicators were selected during this phase, sea surface temperature (SST) and chlorophyll a, due to the importance of these indicators in all marine ecosystems. However, Shin *et al.* (2012) acknowledged that system-specific environmental and climate indicators as well as the identification of regime shifts would be useful in the analysis of ecosystem trends.

For the purpose of this thesis, therefore, environmental indicators were selected based on the known environmental drivers of each particular ecosystem. The inclusion of system-specific environmental indicators aids assessments by allowing appropriate management targets to be set, accounting for the influence of environmental drivers. However, as identified by the IndiSeas project, some environmental indicators such as SST, are applicable across all ecosystems and so this indicator has been included in all case studies. The environmental drivers of each individual ecosystem are considered, allowing the selection of appropriate indicators of environmental variability for each ecosystem. For each case study included in this thesis, the selected environmental indicators utilised are discussed. Data sets were divided into distinct time periods based on known or detected regime shifts.

While definitions of regime shifts in marine ecosystems vary, here we consider a regime shift to be a sudden, drastic shift in the structure and functioning of an ecosystem (Cury and Shannon, 2004). While environmental regime shifts have been clearly observed in some ecosystems, such as the Southern Benguela ecosystem (Howard *et al.*, 2007; Blamey *et al.*, 2012), this may not be the case in all ecosystems. Therefore, the division of data sets, particularly based just on shifts in environmental data, may not be appropriate in all ecosystem case studies.

1.6. Decision Trees

Decision trees and expert systems that incorporate indicators can be used to evaluate the impacts of fishing pressure and ocean climate variability on ecosystems. Decision trees are a form of multicriteria decision analysis (MCDA) and are essentially models which portray a complex decision problem as a tree (e.g. Starfield and Bleloch, 1986; Bundy *et al.*, 2010). MCDA has a wide application in fisheries and resource management for the integration of multiple indicators (Jarre *et al.*, 2008). This is particularly useful for assimilating different types of information and merging different objectives among stakeholders with differing interests (Keeney and Raiffa, 1979). Decision trees can be used to create expert systems, essentially computerised decision trees, that provide users with a logical framework in which to access synthesised information (Osman, 2010). Expert systems generally contain a high level of expertise in a form which makes it accessible to a wider range of users (Jarre *et al.*, 2006). This makes it possible to guide users (in this case stakeholders and fisheries managers) through a complex decision-making process, providing explanations at each step. Expert systems should therefore provide "an effective means of communication between scientists and end users" (Starfield and Louw, 1986) and may aid the process of informing management groups about current knowledge and the potential future states of ecosystems (Jarre *et al.*, 2006).

Decision trees have been widely used in decision making, informing managers in numerous sectors. Classically, each branch of the decision tree would end with a specific objective (Keeney and Raiffa, 1976). Several methods have been applied to move between branches and leaves, including rule-based (e.g. Bundy *et al*. 2010) and fuzzy logic (e.g. Jarre *et al.*, 2008). Outside the marine sector decision trees have been applied to countless ecosystem types and management areas. This includes the management of water quality (e.g. Hart *et al.*, 1999; Atkins *et al.*, 2007), management and assessments of terrestrial ecosystems (Laliberte

et al., 2007; Tien Bui *et al.*, 2012), extinction risk studies (Jones *et al.*, 2006; Sullivan *et al.*, 2006) and conservation planning (Regan *et al.*, 2005; Morrison *et al.*, 2012).

Within a fisheries context, decision trees have been adopted in several studies around the world, aimed at assessing changes within exploited marine communities (e.g. Rochet *et al.*, 2005; Bastardie *et al.*, 2013), involvement of stakeholders within management (Soma, 2003) and even bycatch (Dunn *et al.*, 2011). Typically, these studies assess ecosystems in terms of improving, not improving and deteriorating. However, this thesis has developed a score-based approach, allowing a greater number of categories that can be assigned to indicators, therefore increasing sensitivity of the analysis to indicator trends. Alongside this, in contrast previous decision tree studies, another set of branches has been added to the framework in this thesis to allow consideration of the impacts of environmental variability within the ecosystem. This step is currently lacking within the majority of EAF and decision tree studies, and should therefore prove an important addition.

In the context of the IndiSeas project, the first data-derived indicator-based decision trees framework was created by Bundy *et al.* (2010). The original six indicators from phase one of the project were utilised and a 'rule-based' approach was adopted to assess and classify the nineteen ecosystems initially included in the IndiSeas project. Bundy *et al.* (2010) investigated the adoption of "one strike and you are out" or "two strikes and you are out" rules. These rules, based on the original set of IndiSeas indicators, classified ecosystems as deteriorating when one or two indicators, respectively, were observed to show declining trends. A positive classification of the ecosystem could only be given if no indicators showed declining trends and two or more indicators were observed to significantly increase.

Further decision trees for the Southern Benguela were created utilising model-based indicators in Shannon *et al.* (2014b). Trends in ecological indicators generated from food web

models for specific time periods in the Southern Benguela were analysed. Three decision trees were generated; a pelagic-caught fish community decision tree, a demersal-caught fish community decision tree and an ecosystem-level decision tree. A similar rule-based approach to that developed in Bundy *et al*. (2010) was adopted, classifying the communities as deteriorating if one declining indicator trends was observed (Shannon *et al.*, 2014b). The ecosystem could not be definitively classified if the demersal and pelagic communities showed opposing trends.

This thesis acts to build on these previously developed decision trees (see Table 1.1). Experts will be consulted from the outset, and will advise when analysing trends in each ecosystem, allowing the incorporation of their knowledge alongside that gained from extensive literature reviews and from the data.

Table 1.1: Comparison of methods employed in previous decision trees with the decision tree developed in this thesis.

Previous Decision Trees	Current Decision Tree
Categories assigned to indicators: • Improving • Not improving • Deteriorating	Categories assigned to indicators: • Improving • Possibly improving • Neither improving nor deteriorating • Possibly deteriorating • Deteriorating
Rule-based or fuzzy logic approach	Score-based approach
Analysis of entire data sets	Division of data sets into time periods when appropriate. This allows regime shifts to be accounted for when analysing indicator trends.
Assess the impact of fishing pressure on indicators	Assess the impacts of fishing pressure and environmental variability on indicators
Trends assessed through analysis of data	Inclusion of knowledge from ecosystem experts when assessing indicator trends
Possible redundancies of indicators not identified	Correlations conducted to identify possible redundancies of indicators
Analysis of ecosystem indicators	Indicators separated into ecological and fishing pressure indicators
Ecosystem classified as deteriorating if one or more indicators showed negative trend	All indicator trends taken into account before ecosystem is classified
Fishing and ecological indicators not separated	Fishing pressure indicators combined into one indicators of overall fishing pressure

The inclusion of knowledge from experts is not a new concept, and has not only been used by the IndiSeas project. For example, the FAO has utilised expert knowledge to help validate individual country scores for the UN Code of Conduct for Responsible Fisheries (e.g. FAO, 2002; Pitcher *et al.*, 2006).

There are several advantages associated with including ecosystem experts throughout the development and application of the decision tree. For example, data are provided directly from ecosystem experts, and are hence validated by the experts. The inclusion of experts from various ecosystems into one study also encourages multi-institutional collaborations. However, possible the most important motivation behind the inclusion of ecosystem experts is that these experts are aware of external influences that may be affecting indicators (e.g. social drivers) as well as to sampling specificities (Bundy *et al.*, 2012). Therefore, the inclusion of expert knowledge will permit the necessary insight to correctly interpreted indicators trends and to understand the causes of observed trends. The assimilation of in-depth scientific knowledge into relatively simple frameworks should make it possible to fully understand the ecosystems that will be considered and correctly assess ecosystem state and trends.

In contrast to Bundy *et al.* (2010) and Shannon *et al.* (2014b) decision trees frameworks, the framework that is adopted in this thesis will use a score-based approach, assessing the significance and direction of indicator trends to classify ecosystems. A process of sequential analysis has been developed, including the determination of indicator trends, application of the scoring system, implementation of score adjustment and weighting systems, and determination of an overall ecosystem score (see Figure 1.1.). Trends in ecological, fishing and environmental indicators have been assimilated into the constructed decision trees, making use of the extensive information of reasoning behind trends. This will allow sequential analysis of the impacts of fishing pressure and environmental drivers on ecological

indicators, and overall trends across each ecosystem to be identified. Scoring and weighting systems have been developed to account for the impacts of these drivers, as well as to account for the potential redundancies of correlated indicators. Linear regressions were utilised to detect significant indicator trends. Due to the concern over autocorrelation associated with linear regressions, the Breusch-Gadfrey test for non-linearity was conducted on a sample of indicators across all time periods and across all ecosystems. No significant autocorrelations were detected, and therefore it can be assumed that autocorrelation did not affect observed indicator trends. A goal of this thesis is to identify to whether a generic decision tree framework can be established and to what extent ecosystem specific characteristics must be included in the process of framework development.

The presentation of large amounts of information pertaining to each ecosystem in a way which can be easily understood by stakeholders and mangers is expected to increase general understanding of the state of marine ecosystems in an ecologically meaningful way. The desired outcome of this thesis research is the development of a means of informing fisheries managers and stakeholders of the impacts of both fishing pressure and climatic variability on ecosystems, thus adding to the knowledge base for future fisheries management.

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┌─────────────────────────────────┐
│   Divide data set into Time Periods  │
│          (if appropriate)             │
└─────────────────────────────────┘
                 ↓
┌─────────────────────────────────┐
│      Determine indicator trends      │
└─────────────────────────────────┘
                 ↓
┌─────────────────────────────────┐
│      Attribute scores to indicators    │
└─────────────────────────────────┘
                 ↓
┌─────────────────────────────────┐
│       Combine fishing indicators into    │
│   "Overall Fishing Pressure Indicator"  │
└─────────────────────────────────┘
                 ↓
┌─────────────────────────────────┐
│   Determine impacts of fishing and the   │
│   environment on ecological indicators   │
└─────────────────────────────────┘
                 ↓
┌─────────────────────────────────┐
│            Adjust scores              │
└─────────────────────────────────┘
                 ↓
┌─────────────────────────────────┐
│       Weight correlated indicators     │
└─────────────────────────────────┘
                 ↓
┌─────────────────────────────────┐
│     Attribute overall ecosystem score   │
└─────────────────────────────────┘
```
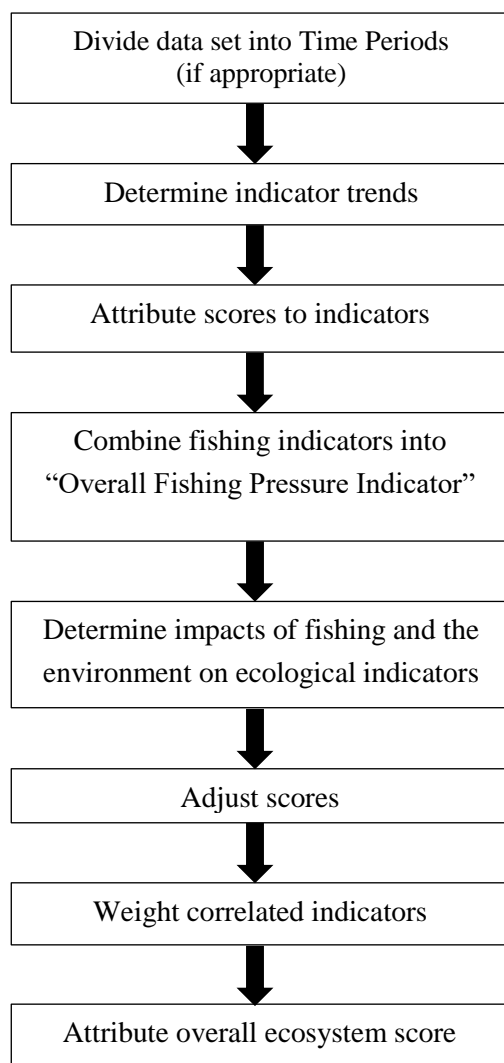
Figure 1.1: The step-by-step process of applying the developed framework to a marine ecosystem.
Firstly, it must be determined whether the data series must be divided into distinct periods based on known environmental shifts. Trends are then determined using linear regressions, and scores are attributed to indicators based on the direction and significance of observed trends. Fishing indicators are combined into one indicator of overall fishing pressure to simplify the decision tree framework. The impacts of fishing pressure and environmental variability are determined, and scores are adjusted to account for these impacts. A weighting is applied to correlated indicators to account for potential redundancies. Then finally the ecosystem is attributed an "overall ecosystem score".

1.7. Other Approaches to Ecosystem Assessments

It is important to consider that the IndiSeas project is not alone in the desire to use indicators to develop an ecosystem approach to fisheries. There are several approaches that have already been developed to aid the assessment of marine ecosystems around the world. One such

approach is the European-based Marine Strategy Framework Directive (MSFD) which aims at determining criteria and methodological standards to allow consistency in the approach of evaluating the achievements of Good Environmental Status (GES). This framework was designed to include the integration of environmental considerations into all relevant policy areas as well as the addition of marine protected areas. Similarly to the approach of the decision tree framework developed in IndiSeas, the process aims at applying an ecosystem based approach in order to manage human activities and enable the sustainable use of marine goods and services. Eleven descriptors, and associated indicators, have been suggested to provide information on ecosystem components (Borja *et al.*, 2010). The application of the MSFD has been complicated by the large number of member states, which may have different priorities and management methods. However, it is envisaged that its successful implementation will result in comparable results across ecosystems.

The International Council for the Exploration of the Sea (ICES) has also acknowledged the need for ecosystem-based management (EBM), and this is now the primary focus of ICES in their efforts to rebuild marine ecosystems (ICES, 2013). Building on examples from the USA, such as the California Current and the Gulf of Mexico (Samhouri *et al.*, 2013), ICES has adopted an approach proposed by Levin *et al.* (2009), known as integrated ecosystem assessments (IEAs). This aims at informing decisions in EBM at multiple scales and across multiple sectors (Levin *et al.*, 2009). ICES considers the challenge in the successful application of EBM to be "to link the incremental changes in selected indicators to a target end-state to the societal costs and benefits of achieving the end-state" (Murawski, 2007; Dickey-Collas, 2014). The ICES Strategic Plan aims to use IEAs in regional seas, creating a fundamental link between ecosystem science and the advice needed to successfully apply the ecosystem approach (ICES, 2013). The IEA framework is designed to guide the process of synthesising and analysing relevant scientific information supporting an ecosystem approach

in any system. It highlights the need to synthesise the available data, or model outputs where data-limited systems are concerned (Levin *et al.*, 2009), and to integrate the diverse physical, biological and socio-economic data into coherent assessments.

Also within the Northern Hemisphere, work has been conducted by the North Pacific Marine Science Organisation (PICES) on co-ordinating international research projects from its member states. In order to provide management advice, meaningful indicators are required that adequately reflect marine ecosystem states. In 2009 the Science Program on Forecasting and Understanding Trends, Uncertainty and Responses of North Pacific Marine Ecosystems (FUTURE) was created (PICES, 2008). This program aims at synthesising and disseminating knowledge provided by multiple research programmes and includes the development of a working group to assess the 'development of ecosystem indicators to characterize ecosystem responses to multiple stressors' (Boldt *et al.*, 2014). PICES has recently developed an implementation plan for the FUTURE program between 2016-2020 (PICES, 2016). A new challenge in implanting FUTURE is to develop products such as periodic ecosystem assessments and forecasts on ecosystem status. A key feature of both FUTURE and the IndiSeas methodology is the movement from previous programmes directed solely at research and enhanced scientific understanding towards an increase in synthetic knowledge that can aid ecosystem health, and hence productivity, in the future.

Recently an attempt has been made to develop a framework that can be applied to all coastal ecosystems. The Ocean Health Index focusses on synthesising individual components of health and benefits to enhance communication and directly compare management objectives (Halpern *et al.*, 2012). This index provides a robust tool that is widely applicable in the analysis of ocean health, with regard to well-accepted societal goals (Halpern *et al.*, 2012). However, it has been acknowledged that global-scale analyses, while useful for global comparisons, are typically inaccurate at local scales. Recently the Ocean Health Index has,

therefore, also been applied at smaller scales, where policy and management decisions are made (see http://www.oceanhealthindex.org/ohi-plus). The application of the framework developed in this thesis, which will likely need to be specifically adjusted for each individual ecosystem, may bridge the gaps created in global-scale studies, and complement those studies that are now being conducted on more local scales.

The framework developed in this thesis relates to several of the above-described methods, ensuring its relevance within current methodologies adopted around the world.

1.8. Selected Ecosystems

The decision tree framework is first developed for the Southern Benguela ecosystem. This ecosystem is a subsystem of one of the world's four eastern boundary upwelling ecosystems, located off the south west coast of Africa and separated from the northern subsystem by the upwelling cell at Lüderitz. Upwelling in this ecosystem is primarily a wind driven process, and therefore change in wind leads to changes in upwelling intensity, in turn affecting primary production (Blamey *et al.*, 2012). As in other upwelling ecosystems small pelagic fish are an important component, and therefore such ecosystems are often driven by 'wasp-waist control' (Cury *et al.*, 2000). This occurs when a small number of species which are dominant within an ecosystem (in the case of the Southern Benguela, sardine and anchovy) influence the abundance of both higher and lower trophic level species within the ecosystem (Cury *et al.*, 2000). Fisheries in the region have been well established for the last 50 to 100 years, with bottom-trawling beginning in the early 1900s and purse seine fisheries developing as a result of the increasing demand for tinned fish following the collapse of the Californian sardine and World War II (van der Lingen *et al.*, 2006; Jarre *et al.*, 2013).

As a participant in the World Summit on Sustainable Development, South Africa is bound to implement an EAF. The goal is for an EAF to compliment the traditional management of target species. Within South Africa the Department of Agriculture, Forestry and Fisheries (DAFF) is primarily responsible for managing fisheries. The fisheries branch is responsible for fisheries research, monitoring and the generation of management advice. An EAF scientific working group has been established as a DAFF advisory group, addressing an EAF at a national level. This group was established as a multiple stakeholder scientific forum, drawing together government, universities, fishing industry representatives, conservation non-governmental organisations and civil society groups. Together these representatives are able to generate scientific research and advice, advancing EAF implementation in South Africa.

The South Catalan Sea in the NW Mediterranean provides a second case study for this thesis. Of those ecosystems considered in the IndiSeas project the South Catalan Sea has been found to have the greatest similarities with upwelling ecosystems. The Southern Benguela and NW Mediterranean have been observed to be most similar in terms of biomasses, flows and consumption (Coll *et al.*, 2006a). As in the Southern Benguela sardine and anchovy make up a large and important component of the ecosystem and are also believed to exert a wasp-waist control on the ecosystem (Cury *et al.*, 2000; Coll *et al.*, 2008). The NW Mediterranean is one of the most intensely impacted areas in the Mediterranean basin, largely due to high population densities in coastal regions and the intense fishing pressure exerted by both Spanish and French fleets (Coll *et al.*, 2012; Micheli *et al.*, 2013a).

This ecosystem presents somewhat more of a challenge when attempting to implement an EAF. Socio-political complexities and the varying management priorities of the many countries bordering the basin, even when the same species is being exploited (Sartor *et al.*, 2014), are likely to complicate the application of the decision tree framework. The impacts of

climatic variability may also be greater on this ecosystem compared to the Southern Benguela. Due to the enclosed nature of the Mediterranean basin the region has been more impacted by climate change than many other regions of the ocean. An example of this can be seen in the expansion of species typically found in the warmer more southerly regions of the basin into the colder more northerly regions as sea surface temperatures steadily increase (e.g. Sabates *et al.*, 2006). In contrast to the Southern Benguela, the Mediterranean basin is largely oligotrophic in nature. Within the South Catalan Sea nutrients are largely introduced though intermittent wind events, the breakdown of the thermocline and river runoff (Estrada, 1996). However, despite the above-mentioned differences it is anticipated that the similarities between the two ecosystems would allow the developed framework to be successfully applied to this ecosystem without too many alterations.

The final case study is the North Sea ecosystem. The North Sea is a temperate sea, strongly influenced by exchanges with Atlantic water masses, and is one of the most heavily exploited and studied shelf seas in the world. Several human activities impact the ecosystem, ranging from fishing, oil and gas production, wind farms, and sediment extraction (Eastwood *et al.*, 2007; Emeis *et al.*, 2015). The area is characterized by irregular changes in productivity, including changes in phytoplankton, zooplankton, and both demersal and pelagic fishes. The historically high levels of fishing have reduced the biomasses of many species in the North Sea, including cod, sturgeon and some elasmobranches (ICES, 2016). Fishing pressure in the region has, however, been reduced since the 2002 Common Fisheries Policy reforms, which should result in increases in large fish species (Gray and Hatchard, 2003). Within this ecosystem shifts in spatial distributions of marine species have been attributed to both climate variability and pressure from exploitation.

Management in the Mediterranean is complex as it is executed by both national and regional entities, as well as several international organisations such as the General Fisheries

Commission for the Mediterranean (GFCM) and the fisheries department of the UN Food and Agriculture Organisation (FAO) (Coll *et al.*, 2014b). All European Union member states must collect data for the Data Collection Framework, aimed at providing the basic data needed to evaluate the state of fisheries resources and support scientific advice regarding the Common Fisheries Policy (Coll *et al.*, 2014b). Conservation in the Mediterranean is, unsurprisingly, complicated by the high diversity of political and cultural systems, as well as varying legal jurisdictions (Micheli *et al.*, 2013b). Management and conservation strategies currently include over 100 marine protected areas, specially protected area and the marine strategy framework directive.

This ecosystem presents a distinctive challenge to the previous ecosystems due to the stringent management methods that have been implemented in the recent past. It is likely that indicator trends reflect the influence of management action, rather than the influence of fishing pressure. This would need to be considered when interpreting indicator trends, as in some cases little or no trend in fishing pressure may go hand in hand with reduced landings, reflecting the positive influence on stocks and the ecosystem in general when effective management is put in place (e.g. decreases in landings).

In order to fully understand observed indicator trends, it is important to consider the state of the ecosystem at the beginning of the time series. All ecosystems included in this thesis were considered to be in an "impacted" state at the beginning of the time series. This was based on knowledge from ecosystem experts on whether (i) there were fully or overexploited stocks at the beginning of the time series, (ii) industrialised or destructive fishing practices were already in practice within the ecosystem and (iii) whether fishing had impacted communities or the ecosystem (such as habitat destruction, loss of top predators, impact on bycatch species etc) (Bundy *et al.*, 2010).

Management in the North Sea ecosystem falls under the Marine Strategy Framework Directive (MSFD). The MSFD requires all EU member states to take measures to achieve Good Environmental Status in their seas by 2020. Within the UK the requirements of the MSFD were incorporated into national legislation through the Marine Strategy Regulations 2010. A key requirement of the MSFD is that member states work together, therefore for the UK regional coordination has concentrated on other Member States within the North East Atlantic region. The OSPAR Regional Sea Convention has been key in this coordination processes, guiding international cooperation for protecting marine environments in the North East Atlantic. Recently the OSPAR Intermediate Assessment has been released, outlining progress made since the 2010 assessments (see https://oap.ospar.org/en/ospar-assessments/intermediate-assessment-2017/).

1.9. Implications and Aims

Understanding how to sustainably manage the world's marine capture fisheries has become increasingly important over recent decades. To help address this issue the research question tackled in this thesis is whether a generic decision tree can be developed, making use of trends in ecological, fishing and environmental indicators, to successfully categorise marine ecosystems around the world. It is anticipated that the developed framework will aid decision support of fisheries management within ecosystems.

The assessment of indicator trends allows the determination of overall ecosystem status, aiding in decision support for fisheries managers and stakeholders within the ecosystems to which they are applied. The framework includes consideration of the influences of both fishing pressure and environmental drivers on ecological indicators, and hence ecosystem components, through the adjustment of indicator scores. This aids the successful categorisation of ecosystem state and trends. The potential redundancy of some ecological

indicators is also considered to ensure the results of the analysis are not biased. A scoring and weighting system developed as part of the analysis accounts for such influences on indicator trends. Once successfully applied to the three case studies, the framework described in this thesis should be applicable to numerous marine ecosystems around the world, when employed in conjunction with knowledge from ecosystem experts to ensure trends are correctly interpreted.

It is envisioned that the creation of a largely generalised decision tree framework, which can be adjusted marginally in order to be applicable to a large number of ecosystems, could allow the comparison of the state and trends of exploited marine ecosystems around the world. The use of a comparative assessment of ecosystems should aid the implementation of an EAF and, as previously mentioned, could help stakeholders and managers learn from already degraded ecosystems and identify early warning signs (Shin and Shannon, 2010).

1.10. Thesis Structure

A brief description of the structure of the thesis is given below:

Chapter 2: **The Use of Ecological, Fishing and Environmental Indicators in Support of Decision Making in Southern Benguela Fisheries**

This chapter outlines the development of the decision tree framework in the context of the Southern Benguela ecosystem. Trends in the selected ecological and fishing indicators from the IndiSeas project are determined along with trends in selected environmental indicators. Data are analysed within distinct periods, accounting for known regime shifts that have occurred within the region. The influences of fishing pressure and environmental drivers on ecological indicator trends are identified, making use of relevant literature alongside advice

from ecosystem experts. A 'score-based' approach was developed to assess overall ecosystem trends. This included a score-adjustment system to account for the impacts of fishing pressure and environmental variability on ecological indicators. Correlations between ecological indicators were determined to account for possible redundancies of indicators, and weightings were applied to reduce the contribution of correlated indicators to the 'overall ecosystem score'. Following assessment of the trends in all indicators the state of the ecosystem could then be assessed and the ecosystem can be categorised. The Southern Benguela was categorised as neither improving nor deteriorating in Period 1 and 2, and as possibly improving in Period 3.

Chapter 3: The Use of Indicators for Decision Support in Northwestern Mediterranean Sea Fisheries

Making use of the same IndiSeas-based suite of ecological and fishing indicators, with the addition of the relevant environmental drivers, the framework developed in the previous chapter was applied to the South Catalan Sea in the Northwestern Mediterranean. Due to the similarities between this ecosystem and the Southern Benguela it was expected that the developed decision tree framework would be generally applicable. The long history of fishing in the Mediterranean, alongside known discrepancies in reported catch data somewhat complicate the interpretation of indicator trends, making ecosystem expert advice essential when understanding overall ecosystem trends. Again, scores attributed to ecological indicators were adjusted to account for the impacts of fishing pressure and environmental variability, as well as the appropriate weightings applied to correlated indicators. Following score adjustments and weighting the ecosystem could be assessed and receive a 'final ecosystem score' allowing the ecosystem state to be classified. The South Catalan Sea was

classified as possible deteriorating in Period 1, and neither improving nor deteriorating in Period 2, when the issue of illegal, unreported and unregulated catch is accounted for.

Chapter 4: The Use of a Decision Tree Framework to Support the Ecosystem Approach to Fisheries Management in the North Sea

The North Sea ecosystem makes up the final case study for this thesis. This ecosystem was somewhat different to the systems considered in Chapters 2 and 3. Stringent management methods have been put in place and so it was expected that management of marine resources would influence ecological and fishing pressure indicator trends. The framework described in Chapter 2 was again be applied to this ecosystem, although further adjustments had to be made due to the strong influence of fisheries management on indicator trends. Due to numerous correlations observed between ecological indicators over the third period correlations between indicators were not taken into account in this case study. Following adjustments of ecological indicators for the impacts of fishing pressure and environmental variability the status of this system was classified and categorised. The North Sea was categorised as neither improving nor deteriorating in Period 1, and as possibly improving in Periods 2 and 3.

Chapter 5: Synthesis and Conclusions

This chapter provides a summary of the entire thesis, discussing the main findings from each of the case studies as well as what was learnt from each individual ecosystem. The complexities of ecosystems and their consideration throughout the development of decision trees is considered, as well as the different considerations needed in individual ecosystems. The similarities of this method to other methods of assessing marine ecosystems currently

being developed and applied are also considered, and the developed framework is placed within the current global context. This allows the usefulness of the current framework to be viewed in a real-world context. The potential issues of ecosystem complexities, limitations of the method and finally the possibilities for future research are also reviewed.

Chapter Two

The Use of Ecological, Fishing and Environmental Indicators in Support of Decision Making in Southern Benguela Fisheries

This chapter has been published as 'Lockerbie, E., Shannon, L. and Jarre, A. (2016). "The use of ecological, fishing and environmental indicators in support of decision making in southern Benguela fisheries." *Ecological Indicators,* **69**: 473-487'. This paper has undergone minor edits for inclusion as a chapter in this thesis to ensure consistency.

2.1. Introduction

2.1.1. The IndiSeas Project and an Ecosystem Approach to Fisheries

Current methods of marine fisheries management are generally considered inadequate, which has resulted in a strong global move towards an ecosystem approach to fisheries (EAF). The concept of an EAF is not new and over the past two decades an interest in an ecological approach to fisheries management has emerged (Cochrane and Starfield, 1992; Garcia and Cochrane, 2005) with the EAF being formalised in 2001(FAO, 2003). In response to this the IndiSeas working group was established in 2005 under the auspices of the Eur-Ocean European Network of Excellence, to examine the use of "EAF Indicators: a comparative approach across ecosystems" (www.indiseas.org). IndiSeas implements comparative analyses of ecosystem indicators from several of the world's marine ecosystems in order to better understand the impacts of fishing pressure and provide decision support for fisheries management (Shin and Shannon, 2010; Shin *et al*., 2012). There are currently numerous

indicators available for marine ecosystems, therefore rather than creating new indicators, the IndiSeas project makes use of strict selection criteria (see Rochet and Trenkel, 2003; Rice and Rochet, 2005) in order to select the most representative and meaningful indicators from those already proposed (Shin *et al.*, 2010b; Coll *et al.*, 2016).

While fishing is generally considered one of the greatest threats to marine fish populations, in some ecosystems other factors such as climate and pollution may inflict even greater pressure (Derraik, 2002; Hoegh-Gulderberg and Bruno, 2010). Fishing and climate variability are known to interact in a number of ways, and therefore cannot be treated separately (e.g. Brander, 2007; Shannon *et al.*, 2010; Jarre *et al.*, 2015). However, despite this few EAF studies of the Southern Benguela include the impacts of climatic variability on the ecosystem quantitatively. This study will therefore follow on from previous IndiSeas work on the Southern Benguela (e.g. Bundy *et al.*, 2010; Shannon *et al.*, 2010), making use of a suite of previously determined ecological and fishing indicators with the addition of a suite of environmental indicators to better aid decision support within Southern Benguela fisheries.

2.1.2. The Southern Benguela Ecosystem

The Benguela Current Large Marine Ecosystem (BCLME) is one of the world's four eastern boundary upwelling ecosystems. Situated off the south west coast of Africa it stretches from the northern border of Angola to East London on South Africa's south coast. The distinctive bathymetry, hydrography, chemistry and trophodynamics make the Benguela one of the most biologically productive areas in the world (Shannon, 2006). The BCLME can be divided into four subsystems; the Angola subtropical, northern Benguela upwelling, Southern Benguela upwelling, and the Agulhas Bank systems (Jarre *et al.*, 2015). The latter two sub systems,

which adjoin the west and south coasts of South Africa respectively, hence forth referred to

as the Southern Benguela ecosystem, are the focus of this study (Figure 2.1).
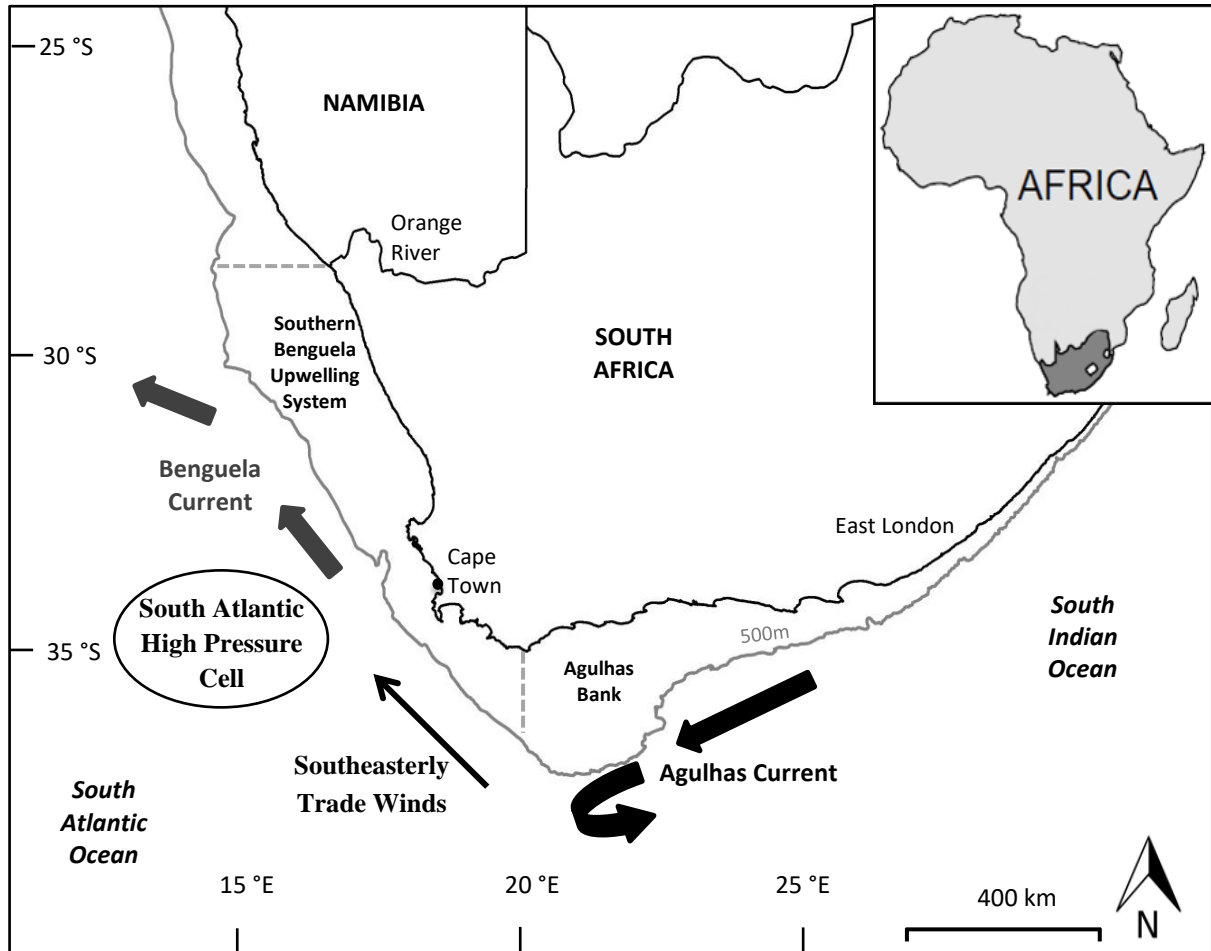
Figure 2.1: A map of the Southern Benguela indicating major currents, the dominant wind system and the approximate location of the South Atlantic High Pressure cell (adjusted from Blamey *et al*. 2015).

There is a long history of fishing in the Southern Benguela, with fisheries being established

for over 100 years. Bottom trawling was initiated in the early 1900s (e.g. Atkinson *et al.*,

2011) and the demand for canned fish resulting from World War II initiated the development

of purse seine fisheries (Griffiths *et al.*, 2004). Like other upwelling systems the Benguela is

characterised by a large and important small pelagic fish component, and is dominated by

sardine (*Sardinops sagax*) and anchovy (*Engraulis encrasicolus*). Due to their high

abundance, the fishery for small pelagics is one of the largest and most productive in the Southern Benguela, accounting for over 75% of total pelagic catches since the 1950s (van der Lingen *et al.*, 2006) as well as over half the marine catch within South Africa's exclusive economic zone (Baust *et al.*, 2015). These small pelagics exert a wasp-waist control on the ecosystem, influencing the abundance of both lower and higher trophic levels (Cury *et al.*, 2000). Fisheries in the Southern Benguela include those for small pelagics (e.g. anchovy, sardine, round herring and sardinella), mid water species (e.g. Cape horse mackerel), demersal fish (e.g. hakes), west coast rock lobster. Several fished species within the BCLME are currently considered to be over exploited, including hakes, sardine and horse mackerel (van der Lingen *et al.*, 2006).

In the Southern Benguela upwelling ecosystem environmental variability is known to strongly impact many ecosystem components (Shannon *et al.*, 2010). The upwelling nature of the Southern Benguela ecosystem, along with the dominance of small pelagic fish species, makes the region particularly susceptible to climatic variability. As a consequence of the dominance of short-lived species, fish production is largely dependent on recruitment, which is controlled by enrichment, concentration and retention processes (Bakun, 1998). These are in turn controlled by environmental factors. It is therefore essential to consider the impacts of environmental drivers within the ecosystem in order to aid fisheries management (Shannon *et al.*, 2010).

2.1.3. The Impacts of Climate on the Southern Benguela

Marine ecosystems change on a variety of timescales, from seasonal up to centennial and even longer, with much variation caused by atmospheric and climatic forcing and related processes (Lehodey *et al.*, 2006). Climatic variability may give rise to important changes in

marine fish populations and, consequently, changes within fisheries, as has been documented for the Benguela ecosystems (Hutchings *et al.*, 2012). Given that environmental variability has been shown to change over time in the Southern Benguela (see Blamey *et al.*, 2012), for the purpose of this study climate variability rather than directional climate change will be used to identify the impacts of the environment on fisheries and ecosystem components. Following extensive literature reviews on how climatic variability impacts the Southern Benguela, and with the advice of ecosystem experts, four important environmental drivers were identified; the position of the South Atlantic High Pressure System, upwelling, sea surface temperature (SST) and chlorophyll concentration. The South Atlantic High Pressure System, also known as the South Atlantic Anticyclone, defines the dominant wind system over the southern Atlantic. This system consists of mid-latitude westerlies, equatorward winds along the west coast of southern Africa, and south-easterly trade winds (Lübbecke *et al.*, 2010). These wind patterns give rise to southerly wind stress near the west coast of Africa whilst to the south of Africa there is a general westerly flow with changes in wind direction associated with mid-latitude cyclones which travel in an east to west direction (Shillington *et al.*, 2006). This high pressure cell shifts seasonally and its interaction with continental lows and the associated cloud band convergence zone results in upwelling-favourable winds on the west coast of South Africa (Hutchings *et al.*, 2009), with consequences for nutrients and primary production. It is essential that the variability of upwelling is considered when attempting to understand the impacts of environmental drivers in the Southern Benguela. The high levels of nutrients transported into the surface layer during upwelling events, which are then available for use by phytoplankton, result in the high levels of productivity observed in upwelling systems. Coastal upwelling off the west coast of South Africa is seasonal, occurring mostly in the summer months (October–March). The pulsing pressure field and consequent southeasterly winds off South Africa's west coast, which are characterised by five

to ten day variabilities, result in variations in upwelling, with a range of two to twelve days and a typical upwelling period of approximately six days (Roy *et al.*, 2001; Crichton *et al.*, 2013).

Temperature plays a central role in biological processes in the oceans due to its impacts on fundamental physiological processes as well as on distribution of marine species. Although it remains unknown how long-term natural variability in temperature will continue to change with fluctuating climatic conditions, it is clear that with increasing heat content of the oceans there will be a strong impact on strength and behaviour of the world's major ocean current systems (Bindoff *et al.*, 2007), resulting in global ecosystem-wide changes.

Chlorophyll concentration can be used as a proxy for primary production, and therefore through the use of ocean colour satellite data it is possible to get an indication of surface phytoplankton levels in the Southern Benguela. However, surface chlorophyll concentrations have been found to be a better indicator of phytoplankton biomass on the west coast than on the south coast of South Africa. The availability of phytoplankton can limit the productivity of fish populations and alter their distribution. For example, in the Southern Benguela, the main limitation on copepod growth is localised food availability, i.e. phytoplankton variability (Richardson and Verheye, 1999). Many of the highly abundant small pelagic fishes in the Southern Benguela rely on copepods as their principal food supply (Cury *et al.*, 2000; van der Lingen *et al.*, 2006a) and therefore these populations would also be influenced by the levels of phytoplankton in the water column.

2.1.4. Decision Support

Indicators support the decision-making process involved in fisheries management in a number of ways, including their description of pressures impacting an ecosystem and

communication of complex interactions to a non-specialist audience (Garcia *et al.*, 2000; Rice, 2000; Rochet and Trenkel, 2003). While many studies use specified reference points in order to aid decision making (Link *et al.*, 2002; Sainsbury and Sumaila, 2003a), here we use the direction of indicator trends. The observed trends in indicators can be placed into decision trees, which can then in turn be computerised in order to create "expert systems" to guide the decision making process while providing supporting information (Jarre *et al.*, 2006a). Decision trees allow for sequential determination of the impacts of pressures such as fishing and ocean climate variability on ecosystems. The high level of expertise involved in expert systems makes it possible to guide users through the decision-making process, providing explanations at each step. Expert systems should therefore provide "an effective means of communication between scientists and end users" and may aid the process of informing management groups about current knowledge and the potential future states of ecosystems (Starfield and Louw, 1986; McGregor, 2015).

While several indicator studies have used a 'rule-based approach' to determining ecosystem classifications (e.g. Bundy *et al.*, 2010) here a score based approach is adopted (following the qualitative modelling philosophy e.g. in Starfield *et al*., 1986; and the line of thinking in Shannon *et al*., 2009; Coll *et al*., 2010), with scores being allocated to indicators based on the significance and direction of their trend. Incorporating our decision tree in a formalised expert system will facilitate the use of trends in ecological, fishing and environmental indicators in terms of assisting the decision-making process when adopting an EAF in the Southern Benguela.

2.2. Materials and Methods

2.2.1. Ecological and Fishing Pressure Indicators

No single indicator is capable of providing management with all the information necessary to implement an EAF, primarily due to the complex nature of ecosystems. Therefore, a suite of ecological and fishing indicators were established in order to evaluate the status of marine ecosystems within a comparative framework (e.g. Shannon *et al*., 2010; Shin and Shannon, 2010; Shin *et al*., 2010a; Coll *et al*., 2012). The indicators used in this study were selected from previous IndiSeas work, with the addition of environmental indicators to incorporate the climatic aspect of ecosystem change. To support an EAF, ecological and fishing indicators must be able to track the state and trends of components and attributes that may be adversely impacted by fishing (Jennings, 2005). The main criteria used when selecting indicators were that they be ecologically significant, sensitive and measurable (Rice and Rochet, 2005). Following careful consideration by the IndiSeas Working Group, an expanded a suite of six ecological and five fishing indicators were selected (Coll *et al.*, 2016) (Table 2.1; see Chapter 1, Section 1.5. for full description and motivation behind indicator selection). These indicators incorporate size-based, species-/life-history based, trophodynamic, pressure-based and simple biomass-related indicators. An important feature of the IndiSeas indicators is that the majority are derived from fisheries independent surveys and therefore may better reflect changes in the actual community, while catch-based indicators depend on confounding factors such as fleet dynamics or under-reporting (Shannon *et al.*, 2014a).

Table 2.1: All IndiSeas derived indicators, including data sources, management objectives and details of indicators (adapted from Coll *et al.*, 2016). Asterisks indicate those indicators which were combined into an indicator of "overall fishing pressure". Abundance and catch data provided by the Department of Agriculture, Forestry and Fisheries, South Africa and trophic level information determined from Fishbase (www.fishbase.org).

INDICATOR	DATA SOURCE	MANAGEMENT OBJECTIVE	DETAILS
MEAN FISH LENGTH	Fisheries independent surveys	Ecosystem functioning	Allows tracking of the direct effects of fishing on a community (Shin *et al.*, 2005). Quantifies the relative abundance of large and small individuals.
MEAN LIFESPAN	Fisheries independent surveys	Maintaining ecosystem stability and resistance to perturbations	Proxy for the mean turnover rate of species and communities. Considered to be a measure of ecosystem stability and resistance to perturbations.
SURVEY BIOMASS	Fisheries independent surveys	Resource potential	Represents biomass of all surveyed species. This is a complex indicator as changes can result from changes in productivity/growth of certain species as well as environmental changes (Bundy *et al.*, 2010).
PROPORTION OF PREDATORS	Fisheries independent surveys	Conservation of biodiversity	Role of predators in ecosystem is essential as they act as dampeners on the whole food web (Sala, 2006). Depletion of predators can lead to trophic cascades (Frank *et al.*, 2006; Daskalov *et al.*, 2007).
TROPHIC LEVEL OF SURVEYED COMMUNITY	Fisheries independent surveys	Ecosystem functioning	Provides the trophic position of organisms sampled in research surveys (Rochet and Trenkel, 2003).
TROPHIC LEVEL OF MODELLED COMMUNITY	Model data (excluding zooplankton and primary producers)	Ecosystem functioning	Aims to cover the full community, not just those species sampled in surveys. Calculated using Ecopath with Ecosim models (see Shannon *et al.*, 2009b).
INVERSE FISHING PRESSURE*	Commercial landings and fisheries-independent surveys	Maintaining ecosystem stability and resistance to perturbations	Measure of resource potential because it reflects the part of the community production dedicated to fishing, calculated as 1/ (landings/biomass). Inverted so that it would decrease under increased fishing pressure to follow trends of other indicators.
LANDINGS*	Commercial landings	Resource potential	Provides knowledge of exploited marine species.
MARINE TROPHIC INDEX*	Commercial landings and estimates of trophic level (empirical and fishbase)	Conservation of biodiversity	Measures the change in mean trophic level of fisheries landings. Calculated from catch composition data collected by the FAO (FAO, 2004). This indicator is cut-off at a trophic level of 3.25.
TROPHIC LEVEL OF LANDINGS*	Commercial landings and estimates of trophic level (empirical and fishbase)	Conservation of biodiversity	Measures the weighted mean TL of species exploited by the fishery, representing the position level of the whole catch.
INTRINSIC VULNERABILITY INDEX OF LANDINGS*	Commercial landings	Ecosystem functioning	Based on life history and ecology characteristics (including maximum length, age at first maturity, longevity, natural mortality, fecundity, spatial behaviour and geographic range – for full details see Cheung *et al.*, 2007). Mean IVI is based on IVIs of all landed species, weighted by contribution of each species to the landed catch. This indicator relates only to the fish community. Note: data necessary to calculate this indicator were only available for period 3.

Indicator data for the Southern Benguela were available from 1978 to 2010, however the decision was made to split the data into three distinct periods. The periods were chosen according to changing regimes occurring in the Southern Benguela system, including changes in upwelling, wind stress and temperature (Howard *et al.*, 2007; Blamey *et al.*, 2012). Based on the regime shifts detected by Blamey *et al.* (2012), the periods considered were Period 1: 1978–1993 (increased upwelling variability), Period 2: 1994–2003 (increased mean upwelling and upwelling favourable winds) and Period 3: 2004–2010 (negative shifts in mean upwelling, upwelling variability, southerly and easterly winds), see Figure 6 in Blamey *et al.* (2012).

2.2.2. Environmental Indicators

Environmental indicators can track environmental change and climatic variability. It is likely that environmental variability will influence the impacts of fishing on an ecosystem, as well as the extent to which ecological indicators capture the effects of fishing alone. Environmental indicators should allow an understanding of the impacts of external drivers in order to be able to mitigate their impacts and adapt management appropriately. In order to be able to track climate variability it was necessary to select environmental indicators for which data are generally available over a long period (Shin *et al.*, 2010b). In this case, data available for selected environmental indicators, with the exception of chlorophyll, were available over the same periods as data for ecological and fishing indicators. Additionally, the underlying processes whereby variability of these indicators influences biological processes in the Southern Benguela have been previously considered (Blamey *et al.*, 2015).

Movement in the position of the South Atlantic High Pressure System has been observed to correspond with shifts in the winds off Lüderitz at the border between South Africa and

Namibia, which may account for observed changes in SST and upwelling in some time series (e.g. Howard *et al.*, 2007; Blamey *et al.*, 2012). Data on the position of this cell were provided by Dr J. Agenbag (see Jarre *et al.*, 2015). Its position was calculated from sea level data downloaded from NOAA, which were monthly values from the area 5–35° S, 30° W–10 ° E. The resulting 40 × 30 array was smoothed using a 3 × 3 moving average procedure. The centre of the high pressure cell was then identified by finding the highest pressure and computing the average latitude and longitude for cells with this pressure. The change in both latitude (northward/southward movement) and longitude (onshore/offshore movement) of the centre of the system can then be used as an environmental indicator of the impacts of such geographical shifts on the Southern Benguela.

Coastal upwelling off the west coast of South Africa is seasonal, occurring mostly in the summer months (October–March). Three main upwelling centres occur at the Cape Peninsula (34° S), Cape Columbine (33° S) and the Namaqua shelf (30° S) (Shannon *et al.*, 1984). However, for the purpose of this study upwelling indices were taken from Blamey *et al*. (2012) and calculated "using geostrophic wind data, derived from monthly sea level pressure (according to Ekman's theory for wind-induced mass transport)". Calculations were performed online at http://las.pfeg.noaa.gov/las6 5/servlets/dataset (see Blamey *et al*., 2012) for the sites as follows: Hondeklip Bay (34° 16'S, 18° 50'E), Cape Columbine (32° 50'S, 17° 50'E), Cape Point (30° 18'S, 17° 14'E), Cape Hangklip (34° 23'S, 18° 50'E) and Cape Agulhas (34° 49'S, 20° 0.6'E).

Changes in SST are one of the most well recorded climatic variations in global oceans. The SST data used in this study make up part of the IndiSeas dataset for the Southern Benguela. Values were derived from the standard 4 km Pathfinder v5.2 AVHRR satellite sensor data. It is important to note that only surface ocean temperatures are available through the use of satellite data, and although SST is a useful tool for looking at temperature change in the

oceans, it gives a very limited view of what is happening below the surface (Williamson, 2013). However, on longer time scales and for the South African west coast in particular, these series of SST are meaningful indicators of the strength of upwelling (Lamont, 2011; Jarre *et al.*, 2015). Current coverage of in situ temperature data for the Southern Benguela is not substantial enough to be used in an ecosystem-wide study, due to insufficient subsurface data collection from ships and buoys over the entire ecosystem.

Chlorophyll data were derived from ocean colour satellites, and therefore as for SST, are a measure only of surface chlorophyll levels. Shipboard data available for chlorophyll concentrations are currently too patchy to be of any use when looking at the Southern Benguela ecosystem as a whole, and while algorithms to estimate chlorophyll levels deeper in the water column have been developed for the open ocean, their use for coastal and upwelling regions is still not entirely understood (e.g. Volpe *et al.*, 2007; Frolov *et al.*, 2012; Williamson, 2013). SeaWiFS (September 1997–June 2002) and MODIS-AQUA data (July 2002 onwards) were used to identify surface chlorophyll concentrations (Djavidnia *et al.*, 2010; Jarre *et al.*, 2015). Surface chlorophyll levels in five different regions within the Southern Benguela were used, namely the Orange River (South African border with Namibia) to the Namaqua shelf (28.2 -31.3°S), Cape Columbine to St Helena Bay (31.3-32.7°S), the Southern West Coast (32.7-34.3°S), the Western Agulhas Bank (WAB, 18.4-20°E) and the combined Central and Eastern Agulhas Bank (CAB/EAB, 20-29°E), (Jarre *et al*., 2015; see Figure 2.2). Surface chlorophyll concentration data were only available from 1997 to 2010, covering part of Period 2 and the whole of Period 3, with no data available for Period 1, therefore chlorophyll concentration cannot be used to identify environmental variability during this period.

Due to length of periods used in this study no assumptions can be made about ocean productivity. Henson *et al.* (2010) identified that at least 15 years of data, ranging up to 50–

60 years depending on the region, are necessary in order to identify productivity trends. Similarly, Lamont (2011) found that while seasonal variability in chlorophyll is evident within the Southern Benguela ecosystem there is no clear long-term trend. Therefore, chlorophyll concentrations will be used purely as an indicator of phytoplankton biomass (standing stock) in the ecosystem.
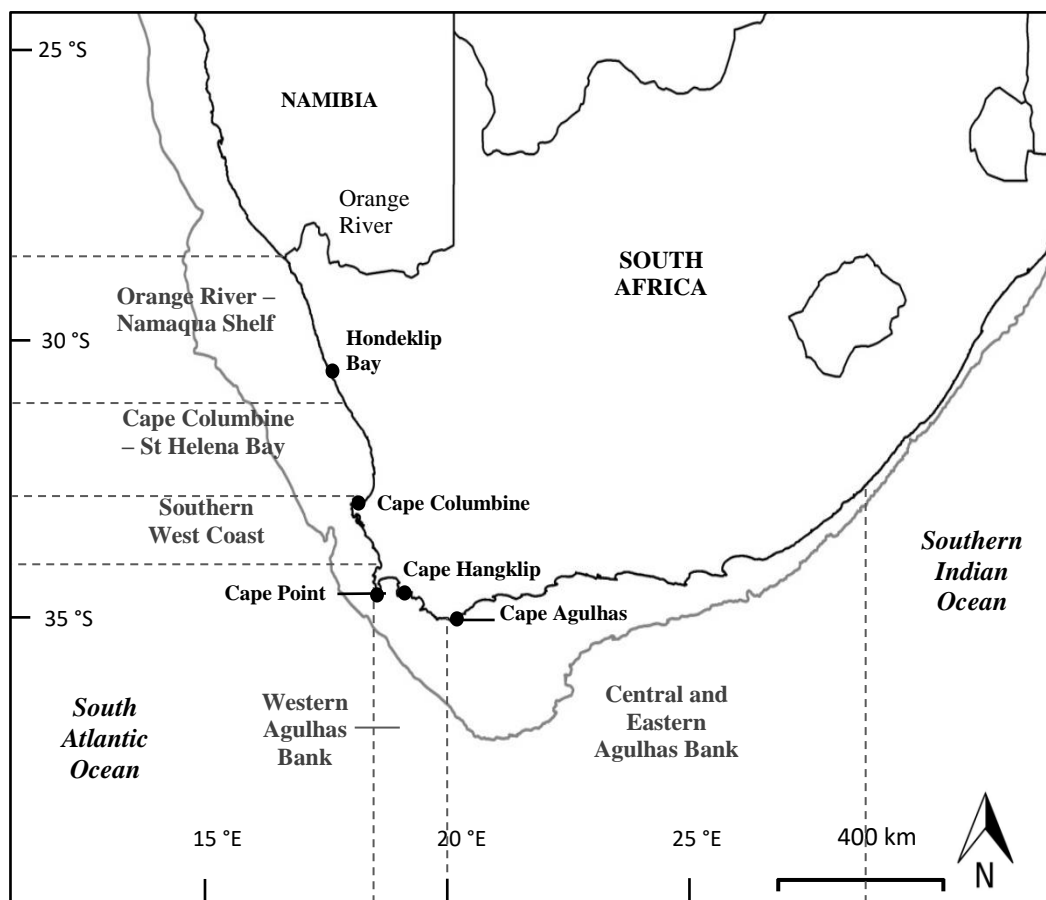
Figure 2.2: Map of the southern Benguela showing sites where upwelling indices were calculated (●) and regions where satellite chlorophyll data (- - -) were derived from (adjusted from Blamey *et al.*, 2015).

2.2.3. Indicator Trend Analysis

Linear regressions were utilised to identify trends in all indicators over selected periods. Although real world problems do not always correspond to the assumptions of a linear regression, an advantage of its use lies in the simplicity, interpretability and scientific acceptance of this approach and it is for these reasons that its use was decided upon. Ecological indicators were formulated by IndiSeas (Shin *et al.*, 2010a) so that a negative trend would be expected under increased fishing pressure. As a score based approach was adopted, indicators were attributed a score depending on whether the slope of the regression was significantly different from zero, as well as the direction of the trend. The following five scoring categories were utilised: highly significant positive trend ($p < 0.05$) = 1; ecologically significant positive trend ($0.05 \leq p < 0.10$) = 2; no trend ($p \geq 0.10$) = 3; ecologically significant negative trend ($0.05 \leq p < 0.10$) = 4; highly significant negative trend ($p < 0.05$) = 5. The term "ecologically" significant is attributed to indicators where although the trend is not statistically significant it could still be considered a significant trend in terms of the impact on the ecology of the ecosystem. The inclusion of this new category arose from discussions with ecosystem experts. This category acts to include trends that reflect important processes within the ecosystem and which are necessary to correctly interpret what is occurring within the ecosystem. A greater number of high scores would suggest that the ecosystem is more severely impacted, while low scores would suggest a less impacted system.

Several indicators (i.e. landings, marine trophic index, trophic level of landings, inverse fishing pressure and, in the case of Period 3, intrinsic vulnerability index, the data for which were only available for this period) can be considered measures of fishing pressure within the ecosystem. These indicators were therefore combined into one indicator of "overall fishing pressure" that could be used to gauge the influence of fishing on the other ecological

indicators. Initially all five fishing indicators were given an equal weighting, however this resulted in a change in fishing pressure trend from "no trend" to "significant increase" during Period 3, however such a weighting may have changed the score too drastically. A second weighting which attributed a weighting of 50% to the inverse fishing pressure indicator (the most representative indicator of direct fishing pressure on the ecosystem) and 50% to the remaining four fishing indicators (see Table 2.1) was therefore decided upon. This resulted in the fishing pressure indicator score increasing from one to two in Periods 1 and 2 and the score remaining at three for Period 3 (see Appendix 1– A1.1 for details of weightings). As has been done with inverse fishing pressure, this indicator has been formulated so that it trends in the same direction as other indicators (i.e. it should decrease, and hence receive a higher score, under increasing fishing pressure) (Shin *et al.*, 2010b). Therefore, the lower the score attributed to overall fishing pressure, the less the impact of fishing pressure on the ecosystem.

The impacts of fishing and the environment on ecological indicators can then be visualised through the use of decision trees. Following extensive literature reviews and discussions with ecosystem experts it was determined whether the trend in the overall fishing pressure indicator and trends in environmental indicators would impact the ecosystem in a way which might explain the observed trend of each ecological indicator. The involvement of an ecosystem expert was crucial at this stage, to ensure that the adjustments applied to indicator trends successfully reflected the expert's understanding of what was driving the ecosystem at each period. In order to simplify whether the impacts of fishing and the environment were impacting observed ecological indicator trends, three categories were created; yes (there appears to be a direct association between trend in fishing/environmental pressures and ecological indicator trends), partial (there appears to be a partial association between fishing/environmental pressure and ecological indicator trend) and no (there appears to be

no/inverse association between fishing/environmental pressure and ecological indicator trends). Original indicator trend scores attributed to ecological indicators were then adjusted by predetermined factors (Figure 2.3) in order to account for the impacts of fishing and the environment. These adjustment factors were chosen after careful consideration in order to adjust scores by a reasonable amount which would allow a difference in score to be observed, but not by so much that the change in score would be unrealistic.
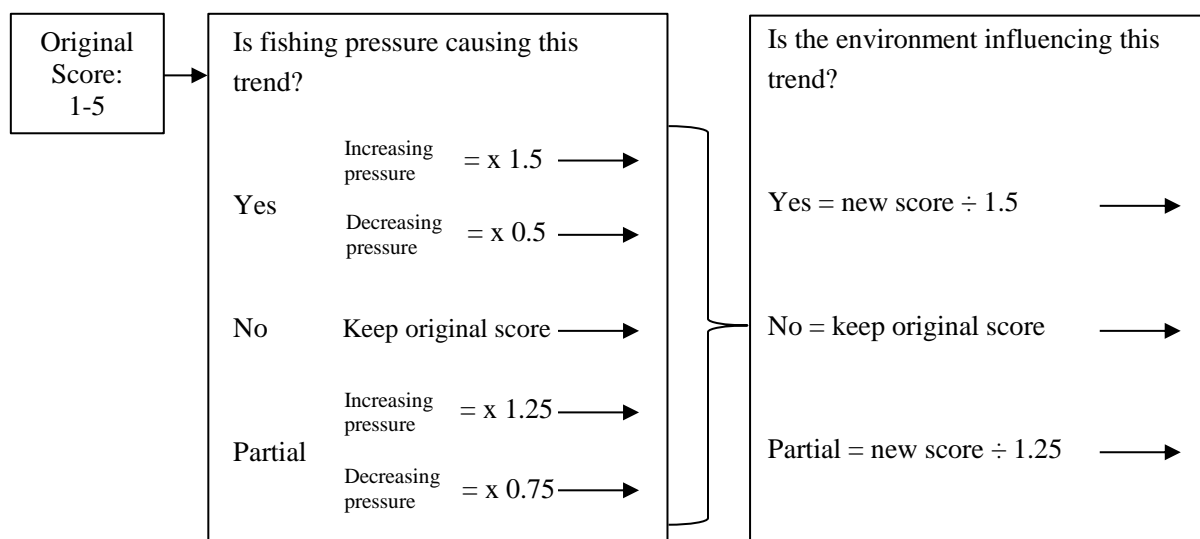
Figure 2.3: Adjustment of original scores attributed to ecological indicators. Original scores were multiplied by the appropriate factor to give a new score, depending on the impact and trend of fishing pressure. The new score was then adjusted again by multiplying by the appropriate factor depending on whether environmental variability could influence this trend to give a final score.

In order to adjust for impact of fishing pressure the original score was multiplied by the appropriate factor (see Figure 2.3) depending on trend in overall fishing pressure and whether the increase/decrease in fishing pressure could result in observed ecological indicator trends (e.g. if overall fishing pressure was observed to decrease and this impact could result in the observed ecological indicator trend the original ecological indicator score was multiplied by 1.5). If there was no change in fishing pressure over the period, the original ecological

indicator score remained the same, as fishing pressure should not have caused any trend. This new score was then carried over in order to identify the impacts of environmental drivers on the ecosystem. If environmental conditions influenced the ecosystem over the considered period in a way which could result in the observed trend in ecological indicator the new score was further adjusted. Again, if no change in environmental indicator was observed the trends score were not changed, keeping the "original score" or "new score" depending on the impact of fishing pressure. Environmental variability was used purely to identify whether environmental conditions could impact the ecological indicator, not whether the impact would be positive or negative for the ecosystem, due to the complex variations in the impacts of environmental conditions on different species within the ecosystem. Therefore, this score adjustment acted only to lessen the impacts of fishing on the indicator in question if the environment has documented reasons to be the cause of the observed trend in order to somewhat mitigate the impacts of environment variability. After this adjustment process each ecological indicator receives a "final score" (Figure 2.3).

Coll *et al*. (2016) noted that indicators which correlate with one another may be somewhat redundant. Therefore, the decision was made to identify correlations between indicators in the selected suites. Spearman's non-parametric rank order correlation coefficient was used to ascertain the statistical dependence between two indicators. This analysis was performed using R version 3.1.1. The "Proportion of Predators" and "Mean Lifespan" were the only indicators which showed significant correlation with each other over all periods in the Southern Benguela. This highlighted the potential redundancy of these indicators, however, at this point the decision was made not to exclude any indicators as all were selected for important reasons and after considerable discussion (Shannon *et al*., 2010; Shin and Shannon, 2010; Shin *et al*., 2010a). A weighting system was selected to reduce their contribution to the overall ecosystem score. A weighting of 0.18 was attributed to the final score (post

adjustment as seen in Figure 2.3) of all indicators which did not show significant correlations with other indicators, while "Proportion of Predators" and "Mean Lifespan" were given a weighting of 0.14. This gave correlated indicators a combined weighting 0.28, greater than that of a single non-correlated indicator yet smaller than two non-correlated indicators (see Appendix 1 – Section A1.2. for full details of weightings and sensitivity analysis). The weighted mean of the final indicator scores were then used to place the ecosystem into one of five categories for each period (Table 2.2).

Table 2.2: Overall ecosystem scores (after application of weightings) and corresponding ecosystem categories of ecosystem classification.

Overall Ecosystem Score	Categorisation
1-1.49	Improving
1.5-2.49	Possibly Improving
2.5-3.49	No Improvement or Deterioration
3.5-4.49	Possibly Deteriorating
4.5-5	Deteriorating

2.3. Results and Discussion

2.3.1. Period 1 (1978-1993)

The majority of linear regressions of ecological indicators for Period 1 showed no significant trend, with the exception of trophic level of the modelled community which showed a highly significant decline (Figure 2.4). This suggests that there was likely little change in ecosystem communities during this time, but highlights the need to use a suite of indicators as varying trends may be observed. The decrease in the trophic level of the modelled community reflects the decline in total modelled biomass of demersal fish over this time period (Shannon *et al.*, 2009), as well as the recovery of small pelagic biomass in the 1990s. It is also important to note that the trophic level of the modelled community includes a greater number of species than those which have been surveyed. The overall fishing pressure indicator showed a

declining trend, suggesting a lessening of fishing pressure. Therefore, the lack of change in almost all the survey and catch based indicators over this period may be unexpected. However, it is important to note that although fishing pressure declined it was still high and continued to exert pressure on species within the ecosystem, therefore care must be taken in interpreting the lack of indicator trends.

Little environmental variation was identified by environmental indicators during this period, with the only significant trend being a southward shift in the latitude of the South Atlantic High Pressure System (Table 2.3). Richter *et al.* (2008)suggest that this system plays a vital role in the Atlantic's mean climatology, and so its movement may therefore be expected to impact the other environmental indicators. The observed southward movement should theoretically result in changes in upwelling over this period, causing either increasing or decreasing as wind dynamics change when the system shifts, however no variability was observed (Table 2.3). During this period there was no significant trend in the upwelling at any of our sites, however Blamey *et al*. (2012) observed that intra-annual variability of upwelling increased over the first half of the 1990s. It is possible that due to the length of time covered in Period 1 such a trend may be masked by variability in other years. There were no significant trends in the longitude of the South Atlantic High Pressure System or in
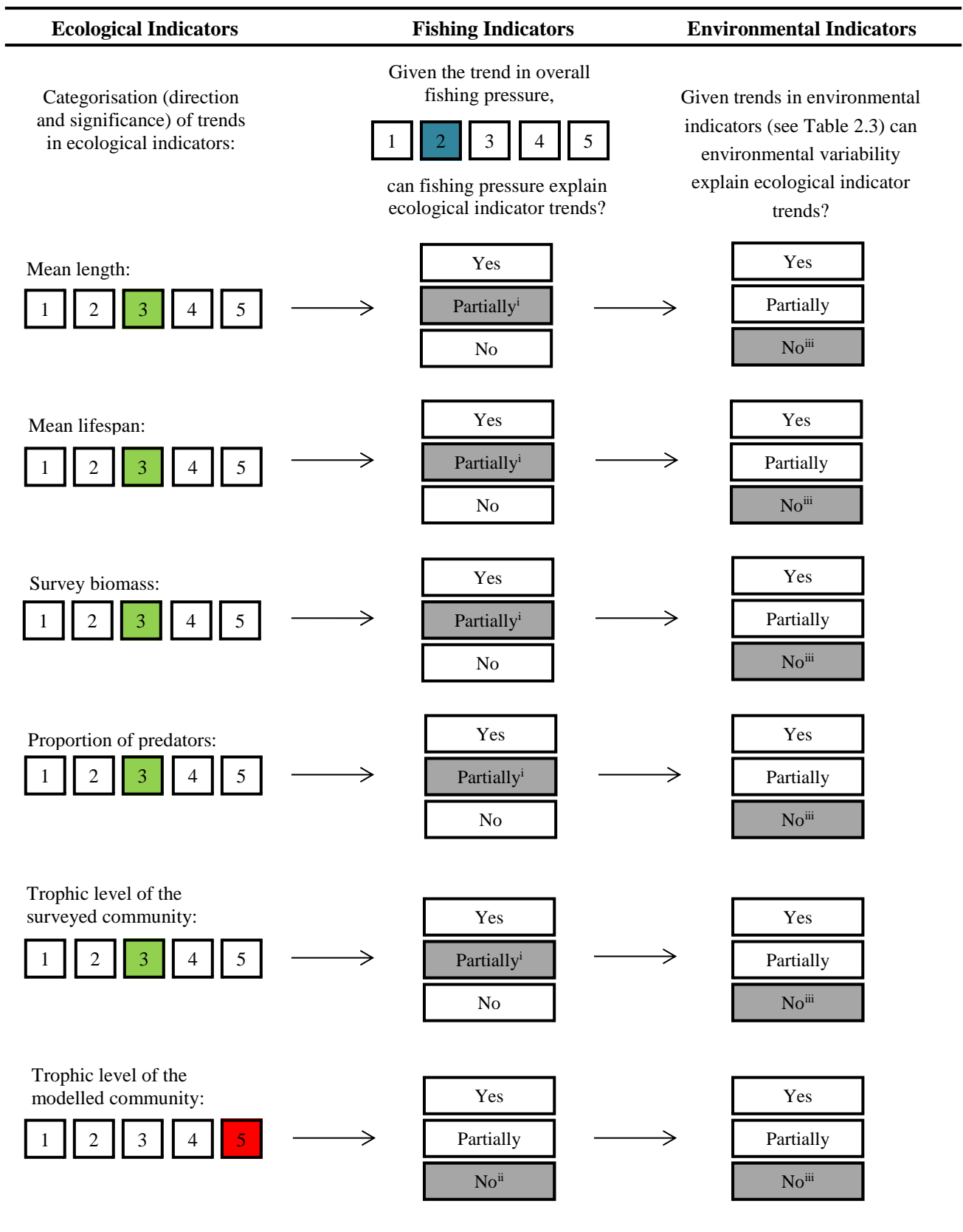
Figure 2.4: Decision tree of ecological indicator trends for Period 1 (1978-1993) and the sequential impacts of fishing pressure and the environment on these indicators. Scoring categories: 1= highly significant positive trend (p<0.05), 2= ecologically significant positive trend (0.05≤ p <0.10), 3= no trend (p = ≥0.10), 4= ecologically significant negative trend (0.05≤ p <0.10), 5= highly significant negative trend (p<0.05). See foot notes for details on how the observed fishing pressure and environmental trends impact each indicator.

SST over this period. We therefore assume that the environmental variability does able to explain observed trends in ecological indicators and so scores were not adjusted for this.

2.3.2. Period 2 (1994-2003)

The various trends in indicators observed over Period 2 (Figure 2.5) suggest complex ecosystem processes over this period. No trend in mean fish length was observed over this period, possibly suggesting a lack of change in the composition of fish within the ecosystem. However, this is contradicted by the highly significant decrease in mean lifespan, suggesting either a decrease in larger fish species or an increase in small fish species. This unexpected trend is possibly result of an unusual and short-lived increase in small pelagic fish species in the Southern Benguela which occurred in the early 2000s (Roy *et al*., 2001). The observed increase in the biomass of small pelagics was likely the cause of observed trends in other indicators over this period, such as the highly significant increase in survey biomass and the highly significant decrease in the proportion of predators. Such trends highlight the need for

expert advice when interpreting indicators as the increase in biomass here cannot be considered as a sign of recovery within the ecosystem as the indicator suggests. The trophic levels of both the surveyed and modelled communities showed highly significant decreases. These trends again highlight the need to be cautious when interpreting indicator trends, for while decreases in mean lifespan, proportion of predators and the trophic levels of both the modelled and surveyed communities may at first appear to suggest a deterioration in the ecosystem, in reality it is again likely that such trends also result from the short-lived but large increase in small pelagic fish in the early 2000s, which could mask trends in other ecosystem components.

A regime shift is known to have occurred in the Southern Benguela in the early 2000s, and is thought to have been induced by environmental changes aggravated by fishing (Roy *et al*., 2001; Howard *et al*., 2007; Blamey *et al*., 2012). Significant trends in environmental indicators may be therefore expected. Upwelling was observed to decrease at Hondeklip Bay, increase at Cape Columbine, Cape Point and Cape Hangklip and neither increase nor decrease at Cape Agulhas. It should be noted here when analysing upwelling during Period 2 the data from year 2003 was excluded as this was the year of the observed regime shift and the inclusion of this year masked true upwelling trends over the rest of the period. An offshore (westward) movement of the South Atlantic High Pressure System was observed during this period causing changes to wind patterns in several areas of the Southern Benguela. Thus, it is possible that this westward shift in the high pressure system resulted in the observed changes in upwelling at some locations (Table 2.3). The short-lived increase in small pelagic species, which likely caused the above-mentioned decreases in ecological indicators, is believed to have resulted from a change in environmental conditions (Howard *et al*., 2007). The positive shifts in pelagic stocks, especially recruits, detected during this period have been explained by anomalous oceanographic conditions, which occurred in the summer

of 1999/2000 providing greater food availability (Howard *et al*., 2007). These anomalous oceanic conditions were observed as a significant decrease in upwelling and an increase in SST which occurred in December 1999 (Roy *et al*., 2001). This may have considerably decreased the advective loss of larvae resulting in a greater number of larvae being transported to the west coast nursery grounds, leading to greater reproductive success. An increase in SST may also have resulted in more rapid larval growth. Following the decrease in upwelling, moderate upwelling was observed in January and February 2000 which may have sustained the high levels of both primary and secondary production thereby increasing food availability. The use of longer term trends in indicators can mask such short-term trends which although not detected in our analyses may still play an important role in ecosystem biology and need to be considered. This again highlights the need to fully consider all ecosystem changes when identifying ecosystem trends as well as the importance of expert knowledge of ecosystems.

Ecological Indicators	Fishing Indicators	Environmental Indicators

Categorisation (direction and significance) of trends in ecological indicators:

Given the trend in overall fishing pressure,

1	2	3	4	5

can fishing pressure explain ecological indicator trends?

Given trends in environmental indicators (see Table 2.3) can environmental variability explain ecological indicator trends?

Mean length:

1	2	3	4	5

Yes
Partially[i]
No

Yes[v]
Partially
No

Mean lifespan:

1	2	3	4	5

Yes
Partially
No[ii]

Yes[v]
Partially
No

Survey biomass:

1	2	3	4	5

Yes[iii]
Partially
No

Yes[vi]
Partially
No

Proportion of predators:

1	2	3	4	5

Yes
Partially
No[iv]

Yes
Partially[vii]
No

Trophic level of the surveyed community:

1	2	3	4	5

Yes
Partially
No[iv]

Yes[viii]
Partially
No

Trophic level of the modelled community:

1	2	3	4	5

Yes
Partially
No[iv]

Yes[viii]
Partially
No

Figure 2.5: Decision tree of ecological indicator trends for Period 2 (1994-2003) and the sequential impacts of fishing pressure and the environment on these indicators. Scoring categories: 1= highly significant positive trend (p<0.05), 2= ecologically significant positive trend (0.05≤ p <0.10), 3= no trend (p = ≥0.10), 4= ecologically significant negative trend (0.05≤ p <0.10), 5= highly significant negative trend (p<0.05). See foot notes for details on how the observed fishing pressure and environmental trends impact each indicator.

[i] Overall fishing pressure decreases, therefore an increase in mean fish length may be expected due to decreased pressure on the ecosystem. It is likely that although fishing pressure decreased over this time period the lack of change in mean fish length was a result of increases in small pelagic fish during this time period while predators have not yet started to recover (Roy *et al.*, 2001).

[ii] Overall fishing pressure decreases, therefore a significant decrease in mean lifespan would not be expected due to decreased pressure at all trophic levels. The highly significant decrease observed here is likely a result of an increase in small pelagic fish that was observed during this time period (Roy *et al.*, 2001).

[iii] Overall fishing pressure decreased; therefore, increased biomass within the ecosystem would be expected as a result of reduced mortality.

[iv] Overall fishing pressure decreased, therefore a decrease in the proportion of predators and trophic level of both the surveyed and modelled communities would not be expected. This may be a result of the unusual and short-lived increase in small pelagic (Roy *et al.*, 2001) while predatory fish populations had not yet shown a recovery.

[v] Observed offshore movement of the South Atlantic High Pressure System along with variability in upwelling (both increases and decreases are observed at different locations). This would influence nutrients and primary production as well as dispersal and recruitment impacting all levels of the ecosystem. This, along with variability in upwelling (see Table 2.3) may have influenced mean length and lifespan of fish as certain environmental conditions favoured certain species. A shift towards conditions which favoured small pelagic species, and their subsequent increase in abundance, could explain the decrease in mean lifespan (Gaylord and Gaines, 2000; Connolly *et al.*, 2001; Rochet and Trenkel, 2003).

[vi] Observed offshore movement of the South Atlantic High Pressure System along with variability in upwelling (both increases and decreases are observed at different locations). This would influence will influence primary productivity, food availability, and the transport of eggs and larvae towards or away from nursery grounds (Cole and McGlade, 1998), all of which could have resulted in the increased biomass within the ecosystem.

[vii] Observed offshore movement of the South Atlantic High Pressure System along with variability in upwelling (both increases and decreases are observed at different locations). It is unlikely this would have directly impacted predatory fish populations; however, there may have been some indirect influence through the impact of environmental variability on lower trophic level species (via impacts on phytoplankton and zooplankton production) which are the prey items of predatory fish.

[viii] Observed offshore movement of the South Atlantic High Pressure System along with variability in upwelling (both increases and decreases are observed at different locations). It is possible the environmental conditions created as a result of these trends may have favoured lower trophic level species, as observed in the increase in small pelagics during this time period (Roy *et al.*, 2001). This increase in small pelagic fish can explain the decrease in trophic level of both the surveyed and modelled community.

2.3.3. Period 3 (2004-2010)

The trends in indicators over Period 3 were the most variable, showing contrasting results (Figure 2.6). Mean fish length showed a highly significant decline, corresponding to a high biomass of small pelagic fish or a decline in larger fish species. However, in direct contrast to this, mean lifespan showed a highly significant increase, suggesting a possible increase in larger fish species with high longevity or a decrease in smaller species. The suggestion of an increase in larger fish species is supported by the significant increase in the proportion of predators observed over this period. However, despite the significant increase in the proportion of predators in the ecosystem, this ratio as well as predatory fish biomass index levels in demersal surveys remained lower in this period than in the previous two (see Blamey *et al.*, 2015 – Figure 8B), and therefore the increase must be carefully interpreted. Trophic level of the surveyed community showed a highly significant increase, again suggesting an increase in larger fish species over this time. It is possible that following the increase in small pelagic fish (which are an important food source for larger fish species) observed in the previous period along with no increase in fishing pressure during this period, predatory fish species were allowed to recover slightly. Trophic level of the modelled community was unfortunately unavailable over this period.

There were no significant trends in SST or chlorophyll concentration over this period, however there was a significant northward movement of the South Atlantic High Pressure System as well as increases in upwelling at Cape Columbine, Cape Hangklip and Cape Agulhas. An increase in offshore transport created when an increase in upwelling occurs may have resulted in a loss of eggs and larvae as they are transported into the open ocean where conditions are typically unfavourable for survival (Hutchings *et al.*, 1998; Parada *et al.*, 2008).

Ecological Indicators	Fishing Indicators	Environmental Indicators

Categorisation (direction and significance) of trends in ecological indicators:

Given the trend in overall fishing pressure,

| 1 | 2 | 3 | 4 | 5 |

can fishing pressure explain ecological indicator trends?

Given trends in environmental indicators (see Table 2.3) can environmental variability explain ecological indicator trends?

Mean length:

| 1 | 2 | 3 | 4 | 5 |

| Yes |
| Partially[i] |
| No |

| Yes |
| Partially[vi] |
| No |

Mean lifespan:

| 1 | 2 | 3 | 4 | 5 |

| Yes |
| Partially[ii] |
| No |

| Yes |
| Partially[vi] |
| No |

Survey biomass:

| 1 | 2 | 3 | 4 | 5 |

| Yes |
| Partially[iii] |
| No |

| Yes |
| Partially[vii] |
| No |

Proportion of predators:

| 1 | 2 | 3 | 4 | 5 |

| Yes |
| Partially[iv] |
| No |

| Yes |
| Partially[viii] |
| No |

Trophic level of the surveyed community:

| 1 | 2 | 3 | 4 | 5 |

| Yes |
| Partially[v] |
| No |

| Yes |
| Partially[iv] |
| No |

Figure 2.6: Decision tree of ecological indicator trends for Period 3 (2004-2010) and the sequential impacts of fishing pressure and the environment on these indicators. Scoring categories: 1= highly significant positive trend (p<0.05), 2= ecologically significant positive trend (0.05≤ p <0.10), 3= no trend (p = ≥0.10), 4= ecologically significant negative trend (0.05≤ p <0.10), 5= highly significant negative trend (p<0.05). See foot notes for details on how the observed fishing pressure and environmental trends impact each indicator.

[i] Overall fishing pressure neither increased/decreased. However, fishing pressure was likely still sustained at a high level, therefore partially explained the decrease in the mean fish length; yet such a significant decline may not have been expected.

[ii] Overall fishing pressure neither increased/decreased. However, under sustained fishing pressure this trend could only partially explain the increase in lifespan. It is possible that this significant increase in the mean lifespan of fish suggests a recovery of some predatory fish populations or a decrease in small pelagic species.

[iii] Overall fishing pressure neither increased/decreased. However, fishing pressure was likely still sustained at a high level and may have limited changes in biomass in the ecosystem.

[iv] Overall fishing pressure neither increased/decreased, and might have allowed predatory fish populations to recover. In the previous period there was a highly significant decrease in the proportion of predatory fish, therefore the significant increase in the proportion of predatory fish may suggest a recovery in their populations.

[v] Overall fishing pressure neither increased/decreased. This along with an increase in the proportion of predatory fish may account for the some of the increase in trophic level of the surveyed community. However, as fishing pressure was sustained we such a significant increase in trophic level may not be expected.

[vi] Northward movement of the South Atlantic High Pressure System and an increase in upwelling at some locations was observed. This may have resulted in changes in nutrient and both phytoplankton and zooplankton which would influence fish populations, potentially influencing mean length and lifespan. However over this time period although there are changes in environmental variables it is important to note that despite changes are happening it is not clear what impacts they will have on marine ecosystems (Blamey *et al.*, 2015).

[vii] Northward movement of the South Atlantic High Pressure System and an increase in upwelling at some locations. This may have influenced the transport of eggs and larvae, and stratification (Gaylord and Gaines, 2000; Connolly *et al.*, 2001). Offshore advection which is often negatively correlated with population size (Harley *et al.*, 2006). Therefore, this may have influenced the biomass within the ecosystem. However over this time period although there were changes in environmental variables it is important to note that despite these changes it is not clear what impacts they will have on marine ecosystems (Blamey *et al.*, 2015).

[viii] Northward movement of the South Atlantic High Pressure System and an increase in upwelling at some locations. This could possibly have an indirect influence on predatory fish populations through impacts on primary production and therefore lower trophic level prey species. However, over this time period although there were changes in environmental variables it is important to note that despite these changes it is not clear what impacts they will have on marine ecosystems (Blamey *et al.*, 2015).

[ix] Species respond individually and therefore would expect shifts in community dynamics and therefore the offshore northward movement of the South Atlantic High Pressure System and an increase upwelling may have resulted in an increase in trophic level of the surveyed community. However, over this time period although there were changes in environmental variables it is important to note that despite these changes it is not clear what impacts they will have on marine ecosystems (Blamey *et al.*, 2015).

Table 2.3: Trends in all environmental indicators over all time periods for the southern Benguela. P values show significance of trends and slope values indicate trend direction. Significant trends are highlighted in bold.

Indicator	Region	Period 1 (1978-1993)	Period 2 (1994-2003)	Period 3 (2004-2010)
South Atlantic High Pressure System	Latitude	**p = 0.03** **slope = -0.17** (Southward movement)	p = 0.76 slope = 0.04	**p = 0.02** **slope = 0.40** (Northward movement)
	Longitude	p=0.22 slope = -0.10	**p = 0.07** **slope = -0.19** (Offshore movement)	p = 0.85 slope = 0.02
Sea Surface Temperature			p = 0.18 slope = 0.11	p = 0.78 slope = 0.03	p = 0.2 slope = -0.22
Chlorophyll Concentration	Orange River-Namaqua		p = 0.90 slope = 0.00	p = 0.902 slope = -0.03
	Cape Columbine-St. Helena		p = 0.67 slope = 0.09	p = 0.524 slope = 0.13
	Southern Cape		p = 0.22 slope = 0.24	p = 0.281 slope = 0.22
	Western Agulhas Bank		p = 0.28 slope = 0.27	p = 0.306 slope = 0.19
	CAB & EAB		p = 0.10 slope = 0.10	p = 0.198 slope = 0.25
Upwelling	Hondeklip Bay	p = 0.62 slope = 0.04	**p = 0.04** **slope = -0.24**	**p = 0.00** **slope = 0.43**
	Cape Columbine	p = 0.7 slope = -0.02	**p = 0.05** **slope = 0.25**	p = 0.17 slope = 0.27
	Cape Point	p = 0.2 slope = -0.08	**p = 0.01** **slope = 0.30**	p = 0.72 slope = 0.07
	Cape Hangklip	p = 0.5 slope=-0.05	**p = 0.00** **slope= 0.33**	**p = 0.03** **slope = 0.37**
	Cape Agulhas	p = 0.4 slope = 0.05	p = 0.17 slope = 0.17	**p = 0.00** **slope = 0.44**

2.3.4. Ecosystem Trends

When indicator scores were adjusted for fishing and environmental driver impacts and weighted to account for the redundancy of some indicators, the final scores for the ecosystem could then be classified for each period (Table 2.4). Period 1 received an ecosystem score of 2.75 and the ecosystem was therefore classified as neither improving nor deteriorating over

Table 2.4: Scores of ecological indicators showing adjustments for the impacts of fishing pressure and the environment. Final ecosystem scores for classification are calculated from the weighted means of adjusted scores. Following score adjustments, the ecosystem was classified as neither improving nor deteriorating in Periods 1 and 2, and as possibly improving in Period 3.

Indicator	Period 1					Period 2					Period 3				
	Original Score	Fishing pressure trend - Does fishing pressure explain trend?	New score	Can trends in environmental indicators explain trend?	Final Score	Original Score	Trend - Does fishing pressure explain trend?	New score	Can trends in environmental indicators explain trend?	Final Score	Original Score	Trend - Does fishing pressure explain trend?	New score	Can trends in environmental indicators explain trend?	Final Score
Mean Length	3	Decreasing - Partially	2.25	No	2.25	3	Decreasing - Partially	2.25	Yes	1.5	5	No change - Partially	5	Partially	4
Mean Lifespan	3	Decreasing - Partially	2.25	No	2.25	5	Decreasing - No	5	Yes	3.33	1	No change - Partially	1	Partially	0.8
Biomass	3	Decreasing - Partially	2.25	No	2.25	1	Decreasing - Yes	0.5	Yes	0.33	3	No change - Yes	3	Partially	2.4
Proportion of Predators	3	Decreasing - Partially	2.25	No	2.25	5	Decreasing - No	5	Yes	4	2	No change - Partially	2	Partially	1.6
Trophic level of surveyed community	3	Decreasing - Partially	2.25	No	2.25	5	Decreasing - No	5	Yes	3.33	1	No change - Partially	1	Partially	0.8
Trophic level of modelled community	5	Decreasing - No	5	No	5	5	Decreasing - No	5	Yes	3.33					
Mean Score	3.33				2.7	4				2.64	2.4				1.92
Weighted Mean					2.75					2.56					1.99

this period, due to the lack of significant trends in the majority of indicators (see Figure 2.4). This is somewhat unexpected as under decreasing fishing pressure at least some ecological indicators would be expected to show an increasing trend, indicating some improvement within the ecosystems. It is possible that the increase in intra-annual variability of upwelling observed by Blamey *et al*. (2012) in the 1990s could be impacting ecological indicators trends in some way, however it is unclear whether this could account for the lack of trends observed over the entire period.

Period 2 received an ecosystem score of 2.56, and therefore during this period the ecosystem can again be considered neither improving nor deteriorating. Contrasting indicator trends were observed, and therefore care must be taken when interpreting indicator trends. Fishing pressure was observed to decrease, resulting in a reduction ("improvement") of some ecological indicator scores during score adjustments, while the environment was observed to strongly impact ecological indicators during this period, leading to a further decrease of ecological indicator scores (see Table 2.4). However, the reasoning behind this lack of improvement under decreasing fishing pressure differs from that in Period 1. For example, trends in indicators suggest that there was a decrease in the large pelagic biomass and a decrease in the proportion predatory fish, which also make up an important component of South Africa's fisheries. This is contrary to what might be expected under decreasing overall fishing pressure. However, it is likely the decreasing trend in the proportion of predators is a result of the increase in small pelagic fish during this period rather than a large decrease in predatory species (Shannon *et al*., 2010). Therefore, when such results are interpreted by fisheries managers and stakeholders it must be cautioned that while indicators may appear to show improvement in some areas, precautions should be taken because some components may remain vulnerable, highlighting the need for expert advice on ecosystem processes. This result contradicts other decision tree studies conducted on the Southern Benguela such as

Bundy *et al.* (2010) where a rule based approach was adopted and the ecosystem was determined to be "deteriorating" (see Bundy *et al.*, 2010 – Figure 1), however it was noted that this result was "more apparent than real" and therefore the classification of neither improving nor deteriorating may be more realistic.

Period 3 (2004–2010) received an ecosystem score of 1.99 and therefore over this period the state of the Southern Benguela ecosystem could be considered to be possibly improving. The fishing pressure over this period was stable, showing neither an increase nor decrease, and therefore did not impact the scores of ecological indicators. The impacts of the environment during this period are more poorly understood than the clear regime shift observed during Period 2 and therefore it is assumed that environmental indicators had only a partial impact on ecological indicator trends. It is possible that the ecosystem was stabilising during this period after the previous regime shift, allowing recovery of some fish stocks. However, in order to fully understand the impact of environmental variability during this period monitoring of future changes should be undertaken. In this study we look at long-term ecosystem scale changes, incorporating a large amount of uncertainty. This uncertainty cannot be resolved in terms of ecosystem assessments when making use of classical paradigms which are purely data driven. By creating a structured framework that makes use of data but also includes expert understanding of ecosystem processes we are able to improve our understandings of what is happening in the ecosystem, hence aiding the application of an EAF. The approach developed in this study could be extended further to incorporate the option of non-linear regression functions whereby improvement/deterioration in indicator trends are classified according to thresholds in indicator values. This would be an interesting next step but would rely on a strong basis for determining threshold values per indicator.

2.4. Conclusions

The sustainable management of fisheries is facilitated by a clear understanding of marine ecosystems and the abiotic, biotic and anthropogenic forces that act on them. The use of ecological, fishing and environmental indicators can greatly aid our comprehension of what is happening in marine ecosystems such as the Southern Benguela. The use of IndiSeas ecological indicators, with the vital addition of a suite of environmental indicators, allowed an expansion of the work with decision trees conducted by Bundy *et al.* (2010).

In addition to contributing to an overall assessment of ecosystem state, trends in indicators will also provide guidance for, and assist in the prioritisation of future short-term research and monitoring, for example if indicators are showing contradictory trends (e.g. as observed in mean length and mean lifespan observed in Period 3).

The structured framework we suggest here has provided meaningful results on ecosystem state for the Southern Benguela. However, although this process can be applied here we are aware that it may not be the best method of synthesis for other marine ecosystems. We therefore plan an analysis of indicators for comparable ecosystems in order to identify if this approach is generally applicable to marine ecosystems or whether it must be individually modified. It is hoped that the same process of using decision trees and an ecologically meaningful weighting and scoring system can be applied directly to other marine ecosystems, allowing a comparative analysis and a versatile method of evaluating ecosystem scale trends over time.

Chapter Three

The Use of Indicators for Decision Support in Northwestern Mediterranean Sea Fisheries

This chapter has been published as 'Lockerbie, E., Coll, M., Shannon, L. J. and Jarre, A. (2017). "The Use of Indicators for Decision Support in Northwestern Mediterranean Sea Fisheries." *Journal of Marine Systems,* **174**: 64-77'. This paper has been edited in its development into a thesis chapter to enhance the coherence of this thesis and to avoid repetition.

3.1. Introduction

3.1.1.   The Mediterranean Sea

The Mediterranean Sea is an enclosed basin, with narrow connections to the Atlantic Ocean via the Strait of Gibraltar, to the Red Sea via the Suez Canal and to the Black Sea via the Bosphorus Strait. This results in restricted exchange between the water masses, with considerable consequences for both circulation and productivity in the Mediterranean basin. Along with the confined nature of the basin this creates the potential for the impacts of climate change to be even more rapid (Calvo *et al.*, 2012). The Mediterranean Sea exhibits high levels of biodiversity, containing an estimated 7% of the world's marine species and with 67% of these species found in the western basin (Link *et al*., 2010; Calvo *et al*., 2012).

There is a long history of human impact in the Mediterranean region due to its habitation for thousands of years. The first observations of depletions of marine resources were reported in Roman times approximately 2500 years ago (Farrugio *et al.*, 1993) and coastal regions were

considered overexploited by the beginning of the 19th century (Margalef, 1985). Lotze *et al.* (2006) suggested that systems, such as those in the Mediterranean, which have been longest exposed to anthropogenic impacts and which have some of the highest coastal human populations, are among the most degraded. Declines in the trophic level of landings has been observed in ecosystems around the world, however, due to the long history of exploitation, Mediterranean fisheries may already be operating at lower trophic levels compared to other systems (Pauly *et al.*, 1998). In the Catalan Sea, the southern region of which is the focus of this project, extensive exploitation has resulted in significantly reduced biomass of predatory fish (Bas *et al.*, 1985), causing fishing fleets to replace large fish species as the top predators within the ecosystem. This has altered the top-down control of forage fish by predatory fish population, so that top predators are now only marginal within this ecosystem (Coll *et al.*, 2006b). It has been suggested that the degraded state of the Northwestern (NW) Mediterranean may have resulted in the region now being less responsive to the impacts of fishing, and potentially more vulnerable to fluctuations in low trophic level species (Shannon *et al*., 2009). Ecosystems which have been degraded by extended intense fishing activity, causing reduced levels of predatory fish, may also become more vulnerable to climatic variability than those which are in a pristine state (Odum, 1985).

Fishing is considered one of the most important drivers of biodiversity change in marine ecosystems (Jackson *et al.*, 2001; Worm *et al.*, 2006). It is believed that this acted as the first major anthropogenic disturbance to coastal areas of the Mediterranean, causing both structural and functional changes within the ecosystem (Lotze *et al*., 2006, 2011). A wide variety of fishing gear and practices are utilised in the Mediterranean, and the development of new fishing technologies and overcapitalisation, along with the increasing demand for marine resources, has further exacerbated the impacts of these activities on ecosystems (Coll *et al*.,

2014b). There is evidence that as a result of increased fishing pressure many stocks in the basin are now either fully exploited or over exploited (Colloca *et al.*, 2013; GFCM, 2013).

Several other human impacts not considered in this study also influence the Mediterranean Sea, including the destruction and degradation of habitats as coastal populations increase, pollution and the introduction of alien species (Lotze *et al.*, 2006; Link *et al.*, 2010; Coll *et al.*, 2012). Human and environmental impacts can effect species differently, and can result in trophic cascades as well as both predation and competitive releases (Ward and Myers, 2005).

Despite numerous studies conducted in the Mediterranean there is still a lack of information on the interactions of anthropogenic and environmental drivers on the ecosystem and the combined influence they will have in the future.

3.1.2. South Catalan Sea Ecosystem and an Ecosystem Approach to Fisheries

The South Catalan Sea in the NW Mediterranean was selected as the second site for the implementation of the decision tree framework described in Chapter 2 (Lockerbie *et al.*, 2016) due to its similarities to upwelling ecosystems, particularly the Southern Benguela (Coll *et al.*, 2006a). These include the dominance of the pelagic component in overall system dynamics, the dominance of small pelagic fish in terms of biomass and catch (notably sardine and anchovy), the role of small pelagics in exerting wasp-waist control within the ecosystem and the importance of hake and horse mackerel (Cury *et al.*, 2000; Shannon *et al.*, 2003). A comparative study of the NW Mediterranean and coastal upwelling systems conducted by Coll *et al.* (2006a) also highlighted that the NW Mediterranean appears to be most similar to the Southern Benguela in terms of biomasses, flows and consumption. It is possible this is a result of the inclusion of the Agulhas Bank in the South African model and the continental

shelf in the NW Mediterranean model which are important in terms of shelf habitat (Shannon and Jarre-Teichmann, 1999).

However, due to the degraded nature of the Mediterranean Sea it is possible that similarities that arise between the two systems, such as the importance of small pelagic fish, may result from the historical declines of large pelagic fish rather than natural similarities. Other anthropogenic impacts have also been linked to small pelagics in the Mediterranean, with the large increase observed in the 1970s being linked anthropogenic impacts, such as the input of nutrients into the water from sewage (Papaconstantinou and Farrugio, 2000).

Indicators have been identified as a useful tool in understanding ecosystem structure and functioning over time and the use of a suite of indicators is necessary in order to assess the impacts of fishing and environmental variability on marine ecosystems (Shannon *et al*., 2010; Shin *et al*., 2010a; Shin and Shannon, 2010). This study follows on from those conducted by the IndiSeas project (www.indiseas.com) and in Chapter 2 of this thesis (Lockerbie *et al*., 2016), making use of a suite of ecological, fishing and environmental indicators in order to aid decision support within South Catalan Sea fisheries and the implementation of an ecosystem approach to fisheries.

3.1.3. The Impacts of Climate Variability in the South Catalan Sea

Current climate change scenario analyses have highlighted a critical need to understand the impacts of climate on marine organisms. The impacts of environmental variability on the Mediterranean Sea are numerous and strongly influence both primary and secondary production (Lloret *et al.*, 2001; Santojanni *et al.*, 2006; Palomera *et al.*, 2007). Fluctuations in environmental conditions are known to occur in the Mediterranean Sea (Lloret *et al*., 2001) and were used to identify the impacts of the environment on the ecosystem. In this study five

environmental drivers were identified as particularly important in the South Catalan Sea, based on knowledge from ecosystem experts, namely the mean annual and winter sea surface temperatures (SST), the North Atlantic Oscillation (NAO), the Western Mediterranean Oscillation (WeMO) and Ebro River Runoff. Although the Atlantic Multidecadal Oscillation (AMO) is also known to influence the Mediterranean Sea as a whole. Link *et al.* (2010) concluded that the AMO does not directly influence the South Catalan Sea ecosystem, and therefore it was not included as a key environmental indicator in this study.

SST has increased on a global scale in recent decades (Trenberth *et al.*, 2007) and several studies show increasing trends in temperature at all depths in the western Mediterranean (Salat *et al.*, 2002; Rixen *et al.*, 2005; Vargas-Yáñez *et al.*, 2008; Coma *et al.*, 2009). Positive SST trends are known to reduce the vertical mixing of the water column in the basin, resulting in nutrient depleted surface waters, and therefore reduced primary production and phytoplankton biomass (Barale *et al.*, 2008; Tsiaras *et al.*, 2012; Volpe *et al.*, 2012). Such impacts can propagate through the food web and influence higher trophic level species, as well as fisheries (Verity and Smetacek, 1996; Bopp *et al.*, 2005). Temperature can also influence fish at various life stages through the survival and growth of larvae, food availability and spawning (Stenseth *et al.*, 2002; Maynou *et al.*, 2014). In the western Mediterranean another significant effect of increases in temperature is the northward expansion of both pelagic and benthic organisms typically found in the warmer southernmost regions. Increasing water temperatures are thought to have resulted in favourable conditions for round sardinella (*Sardinella aurita*)(Sabates *et al.*, 2006), although it is important to note that the expansion of this species has also benefited from decreases in sardine and anchovy over the last fifteen years (Sabates *et al*., 2006).

An important source of climatic variability in the Mediterranean Sea is the NAO which strongly influences weather and climatic variability in the Northern Hemisphere. The NAO is

a large-scale fluctuation in differences in atmospheric pressures along a north-south gradient between the sub-tropical Atlantic (centred on the Azores) and the sub-polar regions (centred on Iceland) (Gordo *et al.*, 2011). These fluctuations influence the speed and direction of westerly surface winds over the North Atlantic, altering the number and track of storms and their associated weather (Hurrell, 1995; Hurrell and Van Loon, 1997; Hurrell, 2005). The NAO affects marine ecosystems in numerous ways. For example, resultant changes to weather conditions influence water column dynamics in turn affecting vertical mixing and therefore nutrient enrichment of the water column. Understanding the dynamics of the water column is essential in understanding fluctuations in phytoplankton within the ecosystem, and it has been observed that some changes in phytoplankton communities in the Mediterranean are related to NAO fluctuations (e.g. Katara *et al.*, 2008; Gladan *et al.*, 2010). Through such changes in phytoplankton the NAO can also alter dominant zooplankton within the ecosystem. Dominant species have been observed to change under positive and negative NAO phases as a result of changes in hydrodynamic features (Molinero *et al.*, 2008). Again, the impacts of such changes can propagate through the food web, influencing marine organisms at all levels.

The WeMO has been defined as a low frequency variability pattern of atmospheric circulation (Martin-Vide and Lopez-Bustins, 2006). Positive WeMO phases are consistent with an anticyclone over the Azores, and the resultant winds blowing from the northwest over the NW Mediterranean (Martín *et al.*, 2012). Negative WeMO phases correspond with the central European anticyclone located north of the Italian peninsula and a low-pressure centre in the Iberian south-west. During this phase winds blow from the eastern sector over the NW Mediterranean (Martín *et al.*, 2012). Martín *et al.* (2012) noted that both positive and negative WeMO phases occurred between 1970 and 2009, with high values identified in mid-1970s and late-1990s and low values in early-1970s, mid-1980s, mid-1990s and towards the end of

the study period. It was also noted that WeMO and NAO exhibited opposite phases, with a negative WeMO observed when NAO was in a positive phase, and vice versa, although the duration of the cycle of these oscillations was different (Martín *et al*., 2012). Further, Martín *et al*. (2012) observed that at the annual scale a significant negative correlation was observed between the WeMO and SST, and a marginally significant positive correlation with Ebro River runoff. The landings per unit effort (LPUE) of both sardine and anchovy were also observed to significantly correlate with the WeMO, while a significant negative correlation existed between LPUE of these species and SST (Martín *et al*., 2012).

River runoff that has been terrestrially enriched is known to favourably impact biological processes such as growth, survival and recruitment, thereby influencing fisheries production (Grimes, 2001). In oligotrophic regions inputs of nutrients can be particularly important as the presence of these nutrients, and the associated primary production, can result in such regions being preferred spawning habitats for some commercially important species (e.g. anchovy: Salat, 1996; Coombs *et al.*, 1997; Sabatés *et al.*, 2001). The Ebro River flows through areas where agriculture is prevalent, resulting in waters which are notably nutrient rich, particularly in nitrogen (Cruzado *et al.*, 2002). Direct nutrient inputs from the Ebro River have been observed to account for between 10% and 25% of all nutrients in the water column of the continental shelf in this region, becoming directly available for use by surface phytoplankton (Salat *et al.*, 2002; Lloret *et al.*, 2004). Lloret *et al*. (2004) observed that the Ebro River runoff plays an important role in anchovy production, while wind mixing is more important in sardine production, although these drivers explained only a small amount of the variability (<25%) in landings within the ecosystem.

3.1.4. Decision Support

The use of indicators for decision support in fisheries management and their value in describing pressures influencing ecosystems and conveying multifaceted interactions to non-specialised audiences has long been recognised (e.g. Garcia *et al*., 2000; Rice, 2000; Rochet and Trenkel, 2003). As for the Southern Benguela system (Chapter 2), this study makes use of the direction and significance of trends in suites of ecological, fishing and environmental indicators in order to aid decision making, rather than the specific reference points utilised in other EAF studies (e.g. Link *et al*., 2002; Sainsbury and Sumaila, 2003). The use of an indicator-based EAF approach may be particularly helpful in the Mediterranean Sea where a significant proportion of landed biomass comes from data-deficient fisheries where stock levels and dynamics may not be known (Pilling *et al.*, 2009).

The use of decision trees allows the visualisation of the sequential analysis of the impacts of both fishing pressure and environmental variability on ecological indicators (Bundy *et al*., 2010). Heymans *et al*. (2014) noted that the food webs in systems with different depths and sizes displayed various levels of fishing. This suggested different exploitation strategies and differing impacts of fishing between ecosystems. Therefore, it is likely that decision trees will need to be 'fine-tuned' for each ecosystem. The objective of this study was to apply the framework created in Chapter 2 (Lockerbie *et al.*, 2016), with slight adjustments in terms of environmental variability indicators, to a similar ecosystem with the potential for application to broader marine systems. Lockerbie *et al*. (2016) aimed to build upon the decision trees developed in Bundy *et al*. (2010) with the addition of suitable indicators of environmental variability (see Chapter 2).

3.2. Materials and Methods

3.2.1. Study Area

The study area was located in the NW Mediterranean Sea (Figure 3.1), one of the most intensely impacted areas of the Mediterranean basin, and is exploited both by Spanish and French fleets (Coll *et al.*, 2012; Micheli *et al.*, 2013a). The majority of fishing activity takes place in coastal regions although important offshore fisheries, such as those for bluefin tuna, swordfish and dolphin fish, also exist.

Figure 3.1: Map of the northwestern Mediterranean Sea showing the study area, the South Catalan Sea. (Continental shelf depth does not exceed 200m).

Four species of small pelagics, namely the European anchovy (*Engraulis encrasicolus*), sardine (*Sardina pilchardus*), round sardinella (*Sardinella aurita*) and sprat (*Sprattus*

sprattus) represent roughly 50% of total landings in the Mediterranean Sea (Lleonart and Maynou, 2003). Of these sardine and anchovy are the most important small pelagic species in terms of biomass and commercial interest, representing the majority of the catch in the South Catalan Sea.

While sardine make up the largest proportion of the catch, anchovy achieve the highest commercial value and are consequently subjected to heavier fishing pressure (Palomera *et al.*, 2007). Sardine are known to exert a wasp-waist control in the NW Mediterranean (as defined by Rice, 1995; Cury *et al.*, 2000) influencing the abundance of both lower and higher trophic level species. Other small pelagics, such as anchovy and round sardinella, have been recognised as important in bottom-up flow control (Hunter and Price, 1992; Coll *et al.*, 2008b). A significant shift in small pelagic fish in the NW Mediterranean Sea, including the decline of biomass of sardine and anchovy and an increase in sprat, has recently been observed (GFCM, 2011; Van Beveren *et al.*, 2014). However, notwithstanding the decrease in biomass the abundance of both sardine and anchovy has remained relatively high, suggesting a decrease in the size of these species (GFCM, 2011, 2012). Van Beveren *et al.* (2014) observed that despite declines in small pelagic biomass sardine recruitment remained high, suggesting changes in biomass are unlikely to be a result of recruitment failure. However, the condition of small pelagic fish species in different areas of the Mediterranean Sea has been declining (Brosset *et al.*, 2017).

These results imply a combined impact of poor condition, slower growth and the decline in larger and older individuals, which can be related to both fishing pressure and environmental variability, were the cause of biomass declines.

3.2.2. Ecological and Fishing Pressure Indicators

Due to the varied and complex natures of ecosystems it has been agreed that no single indicator is capable of describing all aspects of ecosystem dynamics (Cury and Christensen, 2005) and therefore a suite of indicators must be utilised (Shin *et al.*, 2010b). As in Chapter 2 a suite of six ecological and five fishing indicators that can be applied to various marine ecosystems were employed (Shin and Shannon, 2010). These comprised of size-based, species/life-history based, trophodynamic, pressure based and simple biomass-related indicators (Table 2.1, Chapter 2). Biological data were derived mainly from bottom trawling, acoustic surveys, plankton sampling and research cruises conducted by the Institute of Marine Science in Barcelona (ICM-CSIC) while landings data were obtained from the Catalan Government and ICM (see www.indiseas.org).

3.2.3. Environmental Indicators

The underlying processes of how environmental indicators influence the biological processes in the South Catalan Sea have been considered in several studies (e.g. Lloret *et al*., 2001; Coll *et al*., 2005; Link *et al*., 2010). Environmental drivers are known to impact marine ecosystems in numerous ways including influencing the impacts of fishing on ecosystem components and the ability of ecological indicators to capture the effects of fishing alone.

Changes in the atmospheric circulation captured by the NAO index have been linked to changes in weather patterns in the northern hemisphere as well as eliciting varied responses of the surface-, intermediate- and deep-layers of the ocean (Visbeck *et al.*, 1998). SST across the Mediterranean has also been significantly correlated with NAO variability and drier conditions have been associated with high NAO index winters. Although the NAO is active throughout the year its variability and impacts on weather in the northern hemisphere are

most pronounced in winter. Therefore, the winter NAO index, calculated from December to March, was utilised in this study. NAO index values were taken from the public UCAR/NCAR website (www.cgd.ucar.edu/cas/catalog/climind/index.html) while SST was derived from the 4 km Pathfinder v5.2 AVHRR satellite. Both annual mean and winter SST values were collected from near Tarragona (40°N, 2°E) following Coll *et al*. (2008b). Runoff from the Ebro River (annual mean, m3 s−1) was recorded at the Tortosa station (Ebro River Hydrographic Confederation). It is important to note that, due to the building of reservoirs, river runoff may no longer be directly proportional to precipitation in some regions. This is true for the Ebro River which has been extensively dammed, with reservoir capacity now being close to 60% of total annual runoff (Batalla *et al*., 2004). The WeMO index was calculated as the difference in standardised values of sea level pressure between Cádiz-San Fernando, Spain, and Padua, Italy (see Martin-Vide and Lopez-Bustins, 2006) and data were taken from http://www.ub.edu/gc/English/wemo.htm.

3.2.4. Division of Data Series into Time Periods

Data were available for the South Catalan Sea from 1978-2010. However, due to known changes in environmental conditions the decision was made to divide this into two time periods. Several changes were noted in the ecosystem in the late 1980s including changes in atmospheric, hydrological and ecological systems (Conversi *et al*., 2010). Increasing temperature conditions throughout the 1990s also strongly impacted sardine biomass (Palomera *et al*., 2007). A sensitivity analysis was conducted to identify at what point can the known environmental shifts, as mentioned above, can be identified in the environmental indicator trends (see Appendix 2, - Tables A2.1a and A2.1b). Based on analyses of trends in environmental indicators, through which increases in both mean and winter SST, an increase

in the WeMO index and increases and decreases in the NAO were detected, it was decided to divide the data into Period 1 (1978-1990) and Period 2 (1991-2010). This time split was selected as it allowed the detection of documented changes within the ecosystem from the available data.

3.2.5. Analysis of Indicator Trends

As in Chapter 2, for the Southern Benguela, linear regressions were performed in order to analyse the direction and significance of trends in indicators. Following IndiSeas practice (see www.indiseas. org), data series for a given period were standardised before analysis as (indicator value - mean)/standard deviation.

To make this study comparable to that of the Southern Benguela (Chapter 2, Lockerbie *et al*., 2016) the same scoring system was adopted. The same five scoring categories were employed: highly significant positive trend ($p<0.05$) = 1; ecologically significant positive trend ($0.05 \leq p <0.10$) = 2; no trend ($p \geq 0.10$) = 3; ecologically significant negative trend ($0.05 \leq p < 0.10$) = 4; highly significant negative trend ($p<0.05$) = 5. The term "ecologically significant" was used to describe indicators which, although not displaying a statistically significant trend, could still be considered significant in terms of ecological impact on the ecosystem. In this study, strong correlations with a coefficient value larger/smaller than +/- 0.7 were included with a p value less than 0.05 (based on Fowler *et al.*, 1998).

Fishing indicators were combined into one indicator of "overall fishing pressure" using a weighted mean, following the rationale given in Chapter 2, Section 2.3.3 (Lockerbie *et al*. 2016). A weighting of 50% was attributed to inverse fishing pressure (the indicator that is most representative of direct fishing pressure on the ecosystem) and 50% allocated equally to the remaining fishing indicators (Table 3.1).

Table 3.1: Trends in fishing pressure indicators and "overall fishing pressure" scores

Indicator	Period 1	Period 2
Inverse Fishing Pressure	**p = 0.039, slope = 0.099** **Score: 1**	**p = 0.038, slope = 0.082** **Score: 1**
Landings	p = 0.456, slope = -0.054 Score: 3	**p = 0.002, slope = -0.106** **Score: 5**
Trophic Level of Landings	p = 0.301, slope = 0.207 Score: 3	p = 0.908, slope = -0.024 Score: 3
Marine Trophic Index	**p = 0.000, slope = -0.083** **Score: 1**	p = 0.580, slope = 0.003 Score: 3
Intrinsic Vulnerability Index	**p = 0.079, slope = 0.091** **Score: 2**	p = 0.116, slope = 0.0602 Score: 3
Mean Indicator Score	2	3
Overall Fishing Pressure Indicator Score (weighted)	**1.63**	**2.25**

Consistent with the theory behind the IndiSeas project a greater number of high scores suggested a deterioration of the ecosystem during a defined period. Extensive reviews of relevant literature, alongside knowledge gained from ecosystem experts, allowed a full understanding of the impacts of the selected fishing and environmental indicators on the ecosystem and its components. As in Chapter 2, in order to simplify these impacts for use in the decision trees, three categories were adopted here: yes (there appears to be a direct association between trend in fishing/environmental pressure and ecological indicator trends); partial (there appears to be a partial association between fishing/environmental pressures and ecological indicator trends); and no (there appears to be no association between fishing/environmental pressures and ecological indicator trends). To account for these effects original ecological indicator scores were then adjusted by predetermined factors (Figure 3.2). These adjustment factors were selected after careful consideration and sensitivity analyses (see Appendix 2). This allowed the adjustment of original scores by an amount sufficient to make the influences of fishing and the environment clear but not to the extent that the trend would be unrealistically altered (following reasoning in Chapter 2, Section 2.3.3; Lockerbie *et al*., 2016).

Scores were first adjusted to account for the impacts of fishing pressure on the ecosystem (Table 3.1) and whether observed increases or decreases in overall fishing pressure could potentially explain observed trends in each ecological indicator (Figure 3.2). If no trend in fishing pressure was observed no changes would be made to the original ecological indicator scores as, theoretically, fishing pressure should not have caused trends observed in the ecological indicators. The new score attributed to ecological indicators was then adjusted to account for the impacts of environmental drivers on ecosystem components. If the observed trends in environmental indicators could be the cause of trends in ecological indicators, the "new" indicator score was further adjusted (Figure 3.2). Again, if the environment could not be considered to be influencing ecological indicators the score would not be adjusted. It is important to note that at this point environmental indicators were used only to identify whether environmental variability would have influenced the ecosystem and not whether the influence would have positive or negative influences. Due to the complex natures of ecosystems it is likely that environmental conditions would influence the various species within the ecosystem differently, with the potential for both positive and negative impacts. Therefore, the score adjustment for environmental variability acted to lessen the impacts of fishing, assuming that one or more environmental indicator contributed to observed ecological indicator trends.
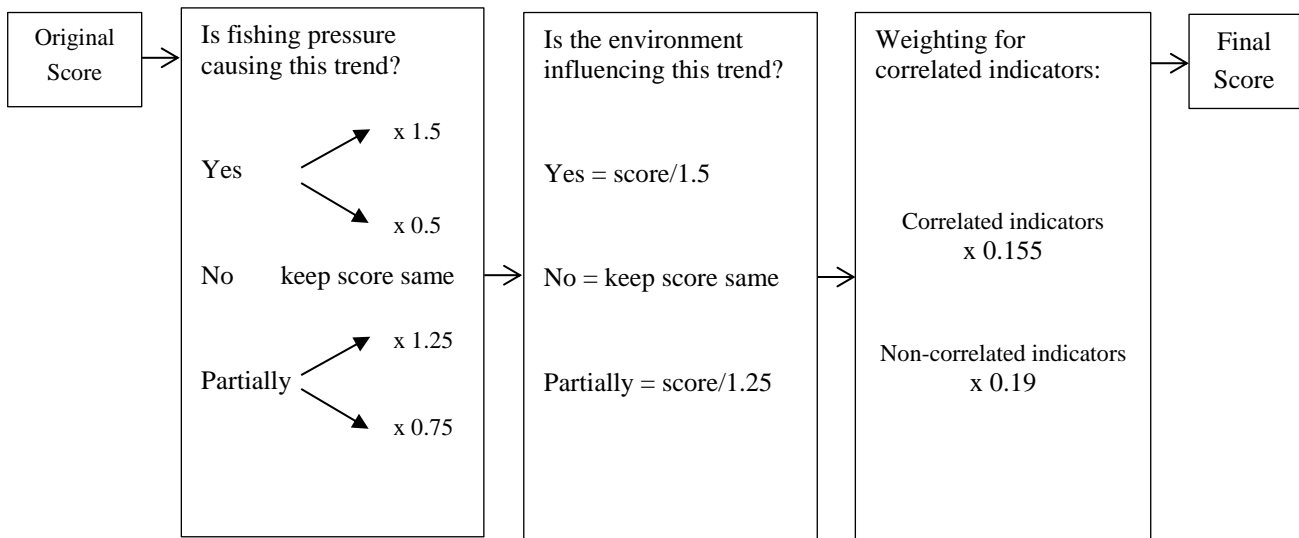
Figure 3.2: Adjustment of original scores attributed to ecological indicators, following Chapter 2, Lockerbie *et al.* (2016). Original scores were multiplied by the appropriate factor to give an intermediate score, depending on the impact (positive / negative as reflected by the arrows) and trend of fishing pressure. The intermediate score was then adjusted again by multiplying by the appropriate factor depending on whether environmental variability could influence this trend to result in the final score for the indicator in question. Finally, the score is adjusted to account for the weighting of correlated and non-correlated indicators (adjustment is ecosystem specific).

As in Chapter 2 a consideration that was taken into account when identifying indicator trends was the possible redundancies of certain indicators. Coll *et al*. (2016) noted that indicators that were correlated might be somewhat redundant. Therefore, Spearman's non-parametric rank order correlation coefficient, using R version 3.1.1., was again used to ascertain the statistical dependence of ecological indicators used in this study. As was observed in Chapter 2, for the Southern Benguela (Lockerbie *et al*., 2016), "proportion of predators" and "mean lifespan" significantly correlated with each other, but in this case study this was only observed in the first period (Appendix 2, Tables A2.2.a & b). The "proportion of predators" and "biomass" were also found to significantly correlate in this ecosystem, occurring over both periods. Such correlations highlight the potential redundancy of these indicators and the need to alter the method used depending on the ecosystem being studied. However, as the indicators were compiled after careful consideration (in the IndiSeas project, see Shannon *et*

al., 2010; Shin et al., 2010a; Shin and Shannon, 2010) the decision was made to retain them all in this study. Therefore to account for the possible redundancies, again following the thinking outlined in Chapter 2, Section 2.3.3. (Lockerbie et al., 2016), the contribution of these indicators to the overall ecosystem score was reduced. A weighting of 0.19 was assigned to non-correlated indicators (post adjustment – see Figure 3.2), while a weighting of 0.155 was assigned to correlated indicators. Once the weighted mean of all indicators was calculated the interpretation of the results allowed the classification of the condition (trend) of the ecosystem during each period according to five categories.

3.3. Results and Discussion

It is important that the historical context of this ecosystem, as described in the Introduction of this chapter, was taken into account when considering the trends of ecological indicators.

3.3.1. The Issue of Illegal, Unreported and Unregulated Catch

The underestimation of catches is a global issue (Zeller and Pauly, 2007) resulting from illegal, unreported and unregulated (IUU) catches (Bray, 2001; Evans, 2001; Pitcher et al., 2002; Agnew et al., 2009). Official reported data are therefore often incapable of capturing actual fishing effort trends, making it increasingly important to identify gaps, biases and misreporting in fisheries (Anticamara et al., 2011; Gorelli et al., 2016).

In the Mediterranean, official landings increased from the 1950s to 1980s, declined in the 1990s and have fluctuated since then. Despite the large number of stakeholders involved in the management of Mediterranean fisheries there are still considerable unreported landings in many Mediterranean countries (Pauly et al., 2014). The European Court of Auditors has highlighted the unreliability of collected data from Member States of the European Union

(resulting in inadequate "total allowable catches"), ineffective national inspection procedures and the lack of sufficient penalties to deter infringements (Court of Auditors, 2007).

The fishing activity in the South Catalan Sea includes bottom trawling, purse seine, bottom and surface longlines and an artisanal fleet. Extensive research of literature and interviews with both fishers and government employees have been conducted in the western Mediterranean in order to determine unreported landings within the Spanish fleet (see Coll *et al*., 2014a, 2014b). Literature reviews focussed on illegal catch, illegal fishing techniques, portions of misreported legal catch and catch not reported in official statistics (e.g. recreational fishing). Interviews with fishers were able to verify the existence of unreported catches and the major sources of non-reporting. This has allowed the quantification of unreported catch and an understanding of how this has changed over time. In conjunction with this, additional information on discarding has greatly added to current knowledge of fishing activities in Mediterranean ecosystems (Coll *et al*., 2014a). In terms of discards, historical data were taken into account where possible, with extrapolations being used when data were not available. By combining information gained from literature with that from interviews with fishers, the percentage of discarded catch for major commercial species has been estimated (Coll *et al*., 2014b).

The 'black market' and subsistence fishing were found to be important sources of discrepancies, including the so called 'take-home catch' and fish that went directly into family businesses or were sold directly to traders. These sources were estimated to add around 20% to the officially reported catch, although responses from fishers in the Gulf of Cadiz put this value closer to 50% (Coll *et al*., 2014b). Another factor that must be considered is increased engine power on vessels that has occurred in recent years. This may have biased estimates and, consequently, catch per unit effort. Engine size and power dictate the size of fishing gear used, decrease time taken to reach fishing grounds (therefore

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increasing time spent fishing) and increase capabilities to work in bad weather (Eigaard *et al.*, 2011; Gorelli *et al.*, 2016). A study similar to that conducted by Coll *et al.* (2014b) was conducted by Gorelli *et al.* (2016) and made use of data from interviews with fishers for deep sea shrimp (*Aristeus antennatus*) in the NW Mediterranean Sea. This study found that 98% of shrimp trawlers misreported engine size in official records, particularly after a limit to engine power of 500 horse power came into effect in 1988. It is likely that the increase in fishing effort was the cause of observed increases in catch in this fishery rather than an increase in the shrimp population (Gorelli *et al.*, 2016). Another source of unreported catches is illegal activity including illegal catch (e.g. undersized specimens or exceeding species quotas), illegal fishing techniques (e.g. Spanish driftnet fishery) and the catch of protected/high risk species (e.g. pelagic sharks) (Coll *et al.*, 2014b). One of the most important components of unreported catch was that of juvenile individuals of commercial species such as hake and anchovy. The unreported catch and bycatch of large pelagic species also make up an important component of total catch with such species being particularly vulnerable and data deficient (Coll *et al.*, 2014b).

The contribution of IUU catches to the total removal of species is not, however, included in the IndiSeas indicators for which data were derived from official records. Therefore, the decrease in overall fishing pressure observed in preliminary indicator trend analysis (Table 3.1) is likely to be misleading. Analysis of ecosystem trends making use of the official data results in the ecosystem being categorised as "possibly improving" (see Appendix 2 – Table A2.3) which, following literature reviews and considering the issues mentioned above, is unlikely to be truly representative of the ecosystem. Following the estimations made in Coll *et al.* (2014b) which suggest an exponential increase in the removal of species from the ecosystem the "overall fishing pressure" scores were adjusted from "ecologically significant decreasing" trends to "ecologically significant increasing" trends. When considering the

almost exponential increase in fishing pressure that has been estimated this is a conservative increase and changes fishing pressure to a more realistic score reflecting an ecologically significant increase.

3.3.2. Period 1 (1978-1990)

As expected, under increased fishing pressure no positive trends were observed in ecological indicators during Period 1. Highly significant decreasing trends were observed in indicators of mean length, mean lifespan and trophic levels of both surveyed and modelled communities, while neither survey biomass nor the proportion of predators showed significant trends (Figure 3.3).

The decrease in the trophic level of the modelled community (TLmc) over this period was likely a result of an increase in the biomass of low trophic level species, such as benthic invertebrates, which significantly increased over this period in the corresponding ecosystem model (Coll *et al.*, 2008a, b). The decrease in the biomass of anglerfish and adult hake observed in the model during this time may also have somewhat contributed to this trend (see Appendix 2 Tables A2.4 and A2.5 for species specific trends). This is consistent with Coll *et al.* (2008b) where such decreases were related to increased fishing mortality and with stock assessments which suggested a number of demersal stocks may have been fully exploited or over-exploited (e.g. Papaconstantinou and Farrugio, 2000). A decrease in demersal sharks was also observed. However, in the Mediterranean due to strong declines in populations of high trophic level predatory fish (e.g. Ferretti *et al.*, 2008; MacKenzie *et al.*, 2009) the biomass of high trophic level species is now so low it is unlikely this would have impacted overall indicator trend. For example, large pelagic fish in the Mediterranean contributed only 8% of the Mediterranean yearly catches between 1983 to 1992. The combined impacts of

high fishing pressure and low selectivity have led to most commercial stocks now being dominated by age one and age two individuals with a low occurrence of older, larger individuals (STEFC, 2009a). The decreasing trend observed in the trophic level of the surveyed community (TLsc) was also likely a result of significant decreases in the surveyed biomass of anglerfish and adult hake and an increase in survey biomass of juvenile hake (Appendix 2, Table A2.4).

Significant decreases in both mean length and lifespan during this period can be linked to increases in low trophic level species and decreases in demersal and large pelagic species. In this ecosystem, size at first catch is similar to size at recruitment for some demersal species, potentially further contributing to these trends (Papaconstantinou and Farrugio, 2000). The decrease in larger species might have resulted in decreased predation mortality on species such as juvenile hake, an increase of which was observed over this period. The decrease in species, such as adult hake, which are known to prey on low trophic level species, may also have contributed to the increase in benthic invertebrates observed in modelled biomass (e.g. Bozzano *et al.*, 2005). The decreases in the trophic level of both the modelled and surveyed communities suggest that the South Catalan Sea may be progressing towards being a demersal-dominated ecosystem (Coll *et al.*, 2005, 2008b).

The lack of trend in survey biomass over this period reflects decreases observed in the abundances of some species and increases of others (e.g. increases in juvenile hake biomass and decreases in adult hake and anglerfish biomass. See Appendix 2 – Table A2.4). The lack of significant trends in the proportion of predators can be related, as previously mentioned, to the already depleted state of the Mediterranean Sea (Bas *et al.*, 1985; Farrugio *et al.*, 1993; Ferretti *et al.*, 2008). The declines in trophic levels, length and lifespan are characteristic of ecosystems that have been exposed to exploitation for extensive periods (Bas *et al.*, 1985; Pauly *et al.*, 1998; Coll *et al.*, 2006b). The already low biomass of predators in the system

Ecological Indicators	Fishing Indicators	Environmental Indicators
Categorisation (direction and significance) of trends in ecological indicators:	Given the trend in overall fishing pressure, [1] [2] [3] [4] [5] can fishing pressure explain ecological indicator trends?	Given trends in environmental indicators (see Table 3.2) can environmental variability explain ecological indicator trends?

Mean length:
[1] [2] [3] [4] [**5**] →

Yes[i]
Partially
No

→

Yes[iv]
Partially
No

Mean lifespan:
[1] [2] [3] [4] [**5**] →

Yes[i]
Partially
No

→

Yes[iv]
Partially
No

Survey biomass:
[1] [2] [**3**] [4] [5] →

Yes
Partially[ii]
No

→

Yes
Partially[v]
No

Proportion of predators:
[1] [2] [**3**] [4] [5] →

Yes
Partially[ii]
No

→

Yes
Partially[vi]
No

Trophic level of the surveyed community:
[1] [2] [3] [4] [**5**] →

Yes[iii]
Partially
No

→

Yes[iv]
Partially
No

Trophic level of the modelled community:
[1] [2] [3] [4] [**5**] →

Yes[iii]
Partially
No

→

Yes[iv]
Partially
No

Figure 3.3: Decision tree of ecological indicator trends for Period 1 (1978-1990) and the sequential impacts of fishing pressure and the environment on these indicators. Scoring categories: 1= highly significant positive trend (p<0.05), 2= ecologically significant positive trend ($0.05 \leq p < 0.10$), 3= no trend (p = ≥ 0.10), 4= ecologically significant negative trend ($0.05 \leq p < 0.10$), 5= highly significant negative trend (p<0.05). See footnotes for details on how the observed fishing pressure and environmental trends impact each indicator.

[i] *With an increase in fishing pressure, decreases in mean length and mean lifespan would be expected in the ecosystem as a result of the targeting of large, high trophic level species and their slower recovery rates.*

[ii] *With increasing fishing pressure it may be expected that survey biomass and the proportion of predators should show a decrease. The lack of changes in survey biomass may result from increases in some species, such as low trophic level species that are released from predation pressure, and decreases in other species, such as demersal species. The lack of change in the proportion of predators can be related to the depleted state of predators within the ecosystem due to the long history of fishing in the Mediterranean (Bas et al., 1985; Farrugio et al., 1993).*

[iii] *With an increase in fishing pressure a decrease in the trophic levels of the survey and modelled communities would be expected. A decrease in the trophic level of surveyed species may have resulted from a decline in the biomass of high trophic level species such as hake and demersal fish such as anglerfish, along with the increase in juvenile hake during this period. A decrease in trophic level of the modelled community may be related to an increase in benthic invertebrates and other low trophic level species as a result of release from predation.*

[iv] *WeMO index decreases over this time period and it is known that the persistence of negative values (which occur towards the end of this time period) can result in decreased river runoff and increased SST, both of which are observed. Increased SST in the oligotrophic Mediterranean reduces vertical mixing and results in nutrient depleted surface waters as well as decreased primary production and phytoplankton biomass (Barale et al., 2008; Tsiaras et al., 2012; Volpe et al., 2012). Under conditions of reduced river runoff the availability of nutrients may restrict phytoplankton growth. River runoff is also known to impact the spawning and growth of anchovy. These factors influence the growth, reproduction and survival of larvae and planktivorous fish and their predators.*

[v] *The increase in the WeMO index and SST observed will influence species within the ecosystem in different ways. It is possible that the environmental conditions that prevailed in Period 1 positively influenced some species while negatively influencing others (e.g. sardine and round sardinella) which could account for the lack of change in ecosystem biomass.*

[vi] *The prevailing environmental conditions are unlikely to strongly influence predatory fish species directly. However, the influence of environmental variability on lower trophic level species may indirectly influence predatory species and, therefore, some change to the proportion of predatory fish may be expected. The lack of change in this indicator may result from the replacement of some prey fish, such as sardine, by other similar species, such as round sardinella.*

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due to heavy exploitation over several decades may have caused this indicator to become less responsive to fishing pressure (Shannon *et al*., 2009).

In terms of environmental variability there are several trends that could have contributed to the observed trends in ecological indicators (Table 3.2). Decreasing trends in WeMO index and river runoff and an increase in mean SST were observed over this period. The decrease in the WeMO index and the persistent negative index values towards the end of Period 1 have been associated with decreased river runoff and increased SST (Martín *et al*., 2012).

Declines in catch of anchovy and sardine, with a one-year lag (Martín *et al*., 2012), may be related to Negative WeMO values over Period 1. Increased SST in the already oligotrophic Mediterranean has been associated with a reduction in vertical mixing and therefore a reduction in nutrients available for primary production in surface waters (Barale *et al*., 2008; Tsiaras *et al*., 2012; Volpe *et al*., 2012). Temperature is known to influence spawning in several species, including that of anchovy and sardine. It is possible that increasing SST may negatively impact sardine which spawn when waters in the region are coolest (Palomera *et al*., 2007). As previously mentioned river runoff is an important source of nutrients in the South Catalan Sea and therefore the decrease in runoff from the Ebro River may further reduce the levels of nutrients being introduced into the ecosystem. River runoff has also been associated with the spawning and growth of anchovy (Salat, 1996; Coombs *et al*., 1997; Sabatés *et al*., 2001), thus a decrease in river runoff may negatively impact the abundance of this species.

Table 3.2: Trends in all environmental indicators over all time periods for the South Catalan Sea. p values show the significance of trends and slope values indicate trend direction. Significant trends are highlighted in bold. Time series of environmental indicators are plotted in the Appendix (Figures A1 and A2).

INDICATOR	PERIOD 1 (1978-1990)	PERIOD 2 (1991-2010)
NAO	p = 0.4149 slope = 0.05223	**p = 0.00483** **slope = -0.10688**
WEMOI	**p = 0.0009** **slope = -0.28189**	p = 0.2577 slope = -0.03333
RIVER RUNOFF	**p = 0.0367** **slope = -0.19750**	p = 0.694 slope =-0.01127
MEAN SST	**p = 0.0080** **slope = 0.15627**	**p = 0.0324** **slope = 0.1006**
WINTER SST	p = 0.2160 slope = 0.05395	**p = 0.0121** **slope = 0.14518**

3.3.3. Period 2 (1991-2010)

Over this period ecological indicators show a variety of trends (Figure 3.4). An important species trend observed over this time were the significant decreases in both the surveyed and modelled biomass of sardine (see Appendix 2 – Table A2.4 and A2.5), one of the most important species in terms of both biomass and commercial value. Given its role in wasp-waist control (e.g. Cury *et al*., 2000; Coll *et al*., 2008b), and commercial importance, sardine likely contributes to the observed decreasing trends in several ecological indicators, including survey biomass and TLmc. Sardine biomass peaked in the mid-1990s and has since shown a decline. Despite the low sardine biomass during Period 2 catches of the species remained high, peaking in the late 1990s/early 2000s, possibly a result of an increase in fishing pressure during this period.

When attempting to understand sardine biomass and catch trends it is important to consider the impacts of environmental variability, which is known to strongly influence small pelagic fish. In other systems, such as the Pacific Ocean and the Benguela upwelling ecosystems, sardine abundance has been positively related to warm periods due to changes in

phytoplankton composition (Chavez *et al*., 2003; Shannon *et al*., 2004; Van der Lingen *et al*., 2006).

Increases in both Mean Annual and Winter SST were observed over this period (Table 3.2). As European sardine favour cool waters for spawning it is possible that the negative impacts of warm periods that have occurred during the spawning season in the western Mediterranean (i.e. increased Winter SST) could mask the possible positive impacts of warm water throughout the rest of the year in terms of growth (Palomera *et al*., 2007). It is therefore likely that increased SST, particularly in winter, over this period may have played a role in the decline of sardine. It is possible that, as suggested by Van Beveren *et al.* (2014), fishing may consequently not be the primary cause of declines in small pelagics in the Mediterranean. As in Period 1, there is no trend in the proportion of predators even under increasing fishing pressure. The same reasons are likely behind this lack of trend, with the already severely depleted levels of large fish within the ecosystem making the indicator less responsive to fishing pressure impacts (Shannon *et al*., 2009).

Ecological Indicators	Fishing Indicators	Environmental Indicators

Categorisation (direction and significance) of trends in ecological indicators:

Given the trend in overall fishing pressure,

| 1 | 2 | 3 | 4 | 5 |

can fishing pressure explain ecological indicator trends?

Given trends in environmental indicators (see Table 3.2) can environmental variability explain ecological indicator trends?

Mean length:

| 1 | 2 | 3 | 4 | 5 |

| Yes[i] |
| Partially |
| No |

| Yes[vii] |
| Partially |
| No |

Mean lifespan:

| 1 | 2 | 3 | 4 | 5 |

| Yes |
| Partially[ii] |
| No |

| Yes[vii] |
| Partially |
| No |

Survey biomass:

| 1 | 2 | 3 | 4 | 5 |

| Yes[iii] |
| Partially |
| No |

| Yes[viii] |
| Partially |
| No |

Proportion of predators:

| 1 | 2 | 3 | 4 | 5 |

| Yes |
| Partially[iv] |
| No |

| Yes |
| Partially[ix] |
| No |

Trophic level of the surveyed community:

| 1 | 2 | 3 | 4 | 5 |

| Yes |
| Partially[v] |
| No |

| Yes[x] |
| Partially |
| No |

Trophic level of the modelled community:

| 1 | 2 | 3 | 4 | 5 |

| Yes[vi] |
| Partially |
| No |

| Yes[x] |
| Partially |
| No |

Figure 3.4: Decision tree of ecological indicator trends for Period 2 (1991-2010) and the sequential impacts of fishing pressure and the environment on these indicators. Scoring categories: 1= highly significant positive trend (p<0.05), 2= ecologically significant positive trend (0.05≤ p <0.10), 3= no trend (p = ≥0.10), 4= ecologically significant negative trend (0.05≤ p <0.10), 5= highly significant negative trend (p<0.05). See foot notes for details on how the observed fishing pressure and environmental trends impact each indicator.

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[i] *With increasing fishing pressure, a decrease in mean length would be expected. This can be linked to increases in low trophic level species and decreases in some demersal large pelagics. For demersal fish species size at first catch is often similar to size at recruitment, potentially contributing to this trend (Papaconstantinou and Farrugio, 2000).*

[ii] *With increasing fishing pressure, an increase in mean lifespan would not be expected. This is likely a result of the significant decline in sardine biomass, along with an increase in some demersal species, such as hake, that have longer lifespans.*

[iii] *With increasing fishing pressure, a decrease in survey biomass would be expected. The observed decrease can be considered a result of the significant decline in the biomass of sardine, an extremely important species in the ecosystem in terms of biomass.*

[iv] *With increasing fishing pressure, a decrease in the proportion of predators may have been expected as these are often highly targeted species. The lack of trend can be related to the already severely depleted state of predators within the ecosystem.*

[v] *With an increase in fishing pressure, a decrease in the TLsc would be expected. However, the opposite is observed in Period 2. The significant increase in TLsc could be a result of a decrease in small pelagic species such as sardine and an increase in higher trophic level species such as hake.*

[vi] *With increasing fishing pressure, a decrease in the TL_{mc} would be expected. This is likely to be due to increased low trophic level species, such as benthic invertebrates, benthic cephalopods and some demersal fish as a result of release from predation pressure as well as an increase in round sardinella. Again, the significant decline in sardine over this period probably plays an important role in this decline.*

[vii] *Significant increases in both Mean Annual and Winter SST values were observed over Period 2. Increases in SST impact water column stratification, affecting small pelagic fish, e.g. sardine. Increased SST reduces vertical mixing and therefore results in nutrient depleted surface waters, decreased primary production and decreased phytoplankton biomass (Barale et al., 2008; Tsiaras et al., 2012; Volpe et al., 2012).*

[viii] *The increase in SST is known to adversely impact sardine which prefer cold waters for spawning (Palomera et al., 2007). This increase in temperature plays an important role in the observed decline of sardine over this period. This could account for the significant decline in survey biomass due to the importance of sardine in terms of biomass within the ecosystem.*

[ix] *Variability in environmental indicators may not directly influence predatory fish species, although indirect influence may occur through impacts on prey species. The lack of change in this indicator could result from the replacement of some prey fish, such as sardine, by other similar species such as round sardinella.*

[x] *The increase in Mean Annual and Winter SST are known to impact sardine spawning within the South Catalan Sea. It is likely the decrease in sardine biomass since the mid-1990s results from such changing environmental conditions (Palomera et al., 2007).*

As previously mentioned it is possible that such changes may make the South Catalan Sea more susceptible to changes in low trophic level organisms and climate variability than a less impacted ecosystem (Odum, 1985).

The decline in sardine and increasing temperatures may also be reasons behind increases of round sardinella in the NW Mediterranean. This small pelagic species is typically found in the warm waters of the eastern and south western basins of the Mediterranean (Ben-Tuvia, 1960), although in recent years it has appeared in cold northern regions (e.g. Francour *et al.*, 1994; Tsikliras and Antonopoulou, 2006). It is probable that it has taken advantage of warmer waters to fill the niche partly vacated by the decline in sardine.

The decrease in TLmc may be a result of increases in several low trophic level species, such as benthic invertebrates, juvenile hake and some small pelagic species such as round sardinella. Decreases in larger species, such as benthopelagic cephalopods and mackerel, may contribute to this decreasing trend. Due to the low levels of large species within the ecosystem the decreasing trend of this indicator signifies an increase in low trophic level species rather than a decrease in larger species. In contrast to this, the TLsc showed a significant increase. The contrasting trends of these indicators are most likely a result of the larger number of modelled (and estimated) species compared to those actually surveyed within the ecosystem. The increase in TLsc may be a result of a significant increase in adult hake during this period, which is also reflected in the modelled biomass (Appendix 2 – Table A2.5).

In terms of other environmental changes impacting indicators in Period 2, a decrease in the NAO was observed. An increase in precipitation over the Mediterranean may, therefore, have been expected (Visbeck *et al.*, 2001), as low NAO values have been linked to intense rainfall and flood. However, the construction of several reservoirs along the Ebro River may have

prevented the expected increase of river runoff into the Mediterranean Sea under such conditions, possibly resulting in the lack of a significant trend in river runoff over this period (Mikhailova, 2003).

3.3.4. Ecosystem Trends

Following the combined evaluation of all indicators (Table 3.3), an ecosystem score of 4.36 was obtained for Period 1 and the ecosystem was therefore classified as showing possible deterioration (Table 2.2, Chapter 2). This score would be expected under the increasing fishing pressure observed over this period (Coll *et al.*, 2014b) and reflects the decline in biomass of several larger species (e.g. hake and anglerfish) as well as the increase in low trophic level species (e.g. benthic invertebrates, juvenile hake and round sardinella). The observed decrease in WeMO index and the associated decrease in river runoff can be linked to a decrease in primary productivity in the NW Mediterranean, which could influence all levels of the South Catalan Sea food web. The increase in SST, which can also be associated with negative WeMO values and which is observed in the Mean Annual SST indicator, can be associated with potential increases in stratification of the water column, a deepening of the thermocline and a reduction in the food supply available for small pelagic species (Roemmich and McGowan, 1995; Martín *et al.*, 2012).

The combined evaluation of indicators for Period 2 resulted in an ecosystem score of 3.20 suggesting the ecosystem was neither improving nor deteriorating over this period. This was reflected in the contrasting trends observed in some ecological indicators. The major cause of trends observed in Period 2 is the decline in sardine biomass, which decreased after peaking in the mid-1990s. However, this decline in biomass coincided with an increase in sardine abundance, suggesting an increase in smaller individuals. A change in sardine has the

potential for ecosystem wide impacts due to the role of this species in wasp-waist control. It is important to note, however, that there are varying suggestions for the primary cause of this trend. It is possible that the trends in biomass and abundance of sardine are due to environmental variability rather than as a result of increasing fishing pressure, while other suggestions include the impacts of the removal of large individuals from the population and even potentially disease (Van Beveren *et al*., 2014). As previously mentioned, the increasing trends in Winter SST have negatively impacted the spawning and recruitment of sardine in the South Catalan Sea (Palomera *et al*., 2007) while simultaneously allowing round sardinella to populate the region (Sabates *et al*., 2006). In contrast to sardine, both anchovy and round sardinella are known to spawn in spring-summer with spawning of anchovy associated with temperatures of 16 °C and 24 °C and that of round sardinella associated with the highest annual temperature of surface waters (Sabatés *et al*., 2009). It may, therefore, be expected that increases in SST may increase the spawning periods available for these two species. However, it cannot be assumed that increasing temperature will continue to positively impact these species. Maynou *et al.* (2014) observed that under extreme warm temperature, such as those observed in 2003 (when temperatures reached 27.6 °C) both these species were negatively impacted. Anchovy eggs and larvae were negatively impacted above 26 °C, while spawning of round sardinella took place up to 27 °C. It is important, therefore, to consider that while in current studies increases in temperature may be advantageous to some species, this may not be the case in future as species have upper limits of temperature suitability.

If fishing pressure was not adjusted to account for the IUU removals documented in Coll *et al*. (2014a, 2014b) the ecosystem received an overall score of 2.5 for Period 1 and 2.06 for Period 2 (Appendix 2 – Table A2.3). This suggested no improvement/deterioration and a possible improvement of the ecosystem respectively. Thus, as expected, a much less pessimistic view of the ecosystem is presented when only official landings data were utilised.

Table 3.3: Scores of ecological indicators showing adjustments for the impacts of fishing pressure and the environment in the South Catalan Sea. Final ecosystem scores for classification are calculated from the weighted means of adjusted scores. Following score adjustments, the ecosystem was classified as possibly deteriorating in Period 1, and neither improving nor deteriorating in Period 2.

Indicator	Period 1					Period 2				
	Original Score	Fishing pressure trend - Does fishing pressure explain trend?	New score	Can trends in environmental indicators explain trend?	Final Score	Original Score	Trend - Does fishing pressure explain trend?	New score	Can trends in environmental indicators explain trend?	Final Score
Mean Length	5	Increasing - Yes	7.5	Yes	5	4	Increasing - Yes	6	Yes	4
Mean Lifespan	5	Increasing - Yes	7.5	Yes	5	1	Increasing - Partially	1.25	Yes	0.83
Biomass	3	Increasing - Partially	6.25	Partially	3	5	Increasing - Yes	7.5	Yes	5
Proportion of Predators	3	Increasing - Partially	6.25	Partially	3	3	Increasing - Partially	6.25	Partially	3
Trophic level of surveyed community	5	Increasing - Yes	7.5	Yes	5	1	Increasing - Partially	1.25	Yes	0.83
Trophic level of modelled community	5	Increasing - Yes	7.5	Yes	5	5	Increasing - Yes	7.5	Yes	5
Mean Score	4.3				4.3	3.2				3.11
Weighted Mean					4.36					3.20

However, the sources of IUU fishing identified in Coll *et al*. (2014a), and acknowledged by the European Court of Auditors (Court of Auditors, 2007), make it unlikely that official landings data are sufficient to determine true ecosystem trends. For example, IUU catches were estimated to average 40% of total reported removals from 1950 to 2010 (Coll *et al*., 2014b; Court of Auditors, 2007). This cannot be ignored and highlights the need for IUU catches to be considered, and suitable indicator adjustments to be made, in order to fully understand the impacts of fishing pressure on marine ecosystems.

3.4. Overall Discussion: The Comparative Context

The sustainable management of marine fisheries is hugely important for the future of marine ecosystems. This holds true particularly for the Mediterranean Sea where the demand for marine resources is so high. Interpreting how the combined influence of different drivers impacts the components of marine ecosystems is a necessary contribution to the knowledge base for fisheries management. Decision trees and expert systems, such as the one developed in this study, complement the results of stock and species assessments and ecosystem models and synthesize the signals of a large set of indicators in a transparent and consistent, hence scientifically defensible, manner.

The use of extensive literature reviews and communication with ecosystem experts allows decision trees to become a useful tool in understanding the state of marine ecosystems. The application of a weighting and scoring system and the use of decision trees applied to the Southern Benguela in Chapter 2 (Lockerbie *et al*., 2016) appears to work well for the South Catalan Sea. This was somewhat expected due to the similarities between the two ecosystems that were identified in the results of the IndiSeas I project, where a generic approach was developed making use of a more limited set of drivers of ecosystem state, namely those only

pertaining to fishing pressure (Bundy *et al*., 2010). Taking account of warnings expressed, e.g. by Heymans *et al*. (2014) this study represents a valuable test in how far an application developed for a specific case study (i.e. the Southern Benguela in Chapter 2, (Lockerbie *et al*., 2016) can be generalised, particularly in view of the addition of a major driver of dynamics in upwelling systems (i.e. climate variability). In this regard, this study has shown that for two broadly similar systems a selection of indicators generally showing the impact of fishing (Shin and Shannon, 2010), combined with the best specific indicators representing environmental drivers for each ecosystem, can be subjected to a general philosophy of synthetic evaluation, as represented through the successful use of the same decision tree framework (Figure 3.4 and Table 3.3).

The need to adjust the fishing pressure indicator from an "ecologically significant decrease" to an "ecologically significant increase" to account for IUU catches stresses an important issue within fisheries management and within the method described above. Lack of sufficient data and incomplete datasets present a significant problem when attempting to develop successful decision support system. IUU catches bias available data as well as the ability of scientists to successfully interpret the impacts of fishing on marine species and ecosystems (Zeller and Pauly, 2007; Coll *et al*., 2014b). When IUU fishing is not accounted for the outcome of the outlined method suggested that the ecosystem showed no decline in Period 1 and that it may have been improving across Period 2. This demonstrates that when fishing pressure is underestimated so are its impacts within the ecosystem. Without appropriate adjustments to account for IUU fishing it would not be possible to provide fisheries managers and stakeholders with sound advice. This could have serious consequences for sustainable fisheries management within the ecosystem with the potential to result in species declines and collapses due to overfishing and lack of necessary limits on vulnerable species. This again

emphasises the need to conduct extensive literature reviews and work closely with ecosystem experts for this method of decision support to be successful.

Despite their similarities some important differences between the South Catalan Sea and the Southern Benguela should be kept in mind. For example, primary production is lower in the NW Mediterranean than in the Southern Benguela system, with higher transfer efficiencies in the NW Mediterranean. This suggests it is a food limited system (Coll *et al*., 2006a), which is more consistent with the oligotrophic nature of the Mediterranean (Cushing, 1975) than with eastern boundary upwelling systems. Another important difference is the set of environmental drivers influencing ecosystem dynamics. For example, the consideration of the impacts of the NAO and WeMO in the Mediterranean would not be appropriate for the Southern Benguela. This emphasises the need to fine-tune the set of indicators entering the decision trees and expert systems for each ecosystem with regional and local data and expert advice.

Several processes known to impact Mediterranean ecosystems such as pollution, coastal zone development and the increase in invasive species are not considered in this study. As the development of coastal regions in the Mediterranean increases, pollution from urban developments, industries, agriculture, aquaculture and numerous other sources has become increasingly harmful to marine ecosystems (Coll *et al*., 2012; Micheli *et al*., 2013). Invasive species have been, and continue to be, introduced into the Mediterranean through various sources including ballast water, the Suez Canal and mariculture. Although these processes also occur in the Benguela their importance is thought to be much lower than in the Mediterranean.

Despite the above-mentioned differences between the Southern Benguela and the South Catalan Sea, this study shows that the same basic structured framework can be successfully applied to different ecosystems to produce ecologically meaningful results. The following

chapter will include the application of this framework to an ecosystem where conditions are not as similar as the two considered thus far, given the overall objective to produce a versatile tool for evaluating marine ecosystems around the world, with relevance to ecosystem-based management.

3.5. Conclusions

The application of the decision tree framework developed in Chapter 2 (Lockerbie *et al*., 2016) for the Southern Benguela appears to be directly applicable to the South Catalan Sea, with the inclusion of appropriate environmental drivers of the Mediterranean ecosystem. The South Catalan Sea was observed to be possibly deteriorating during Period 1 and neither improving nor deteriorating in Period 2. This is consistent with ecological theory for the ecosystem over these periods and can be explained by species trends and with expert advice. Trends in indicators in both the Southern Benguela and South Catalan Sea show the impacts of fishing within these ecosystems and highlight the need to employ ecosystem-based management. And finally, the creation of expert systems needs to be a dynamic process (Jarre *et al*., 2006), adjusting as our knowledge on ecosystems and their processes, and data availability, improves.

Chapter Four

Applying a decision tree framework in support of an ecosystem approach to fisheries: IndiSeas indicators in the North Sea

The contents of this chapter have been submitted for publication 'Lockerbie, E., Lynam, C. P., Shannon, L. J. and Jarre, A. (submitted). "The Use of a Decision Tree Framework to Support the Ecosystem Approach to Fisheries Management in the North Sea." *ICES Journal of Marine Science'*. Reviewer comments have been received and the paper is being prepared for resubmission. This paper has been edited in its development into a thesis chapter to enhance the coherence of this thesis and to avoid repetition.

4.1. Introduction

4.1.1. The Impact of Fisheries on the North Sea Ecosystem

Although the North Sea covers less than 1% of the global ocean, it is one of the most important economic regions in the world. Among the most intensively exploited shelf seas, it is impacted by many anthropogenic activities including fishing, pollution, oil and gas production, wind farms and shipping (Eastwood *et al.*, 2007; Emeis *et al.*, 2015). While the consequences of such exploitation are not fully understood, the North Sea is now in a highly perturbed state (see ICES, 2016). Selective removal of species from the North Sea has altered community structure and predator-prey interactions within food webs, likely influencing ecosystem functioning (Jennings and Kaiser, 1998; ICES, 2016). However, fishing effort in

the North Sea has been closely monitored over recent decades and has significantly reduced since the 2002 Common Fisheries Policy reforms (Jennings et al., 2002; Gray and Hatchard, 2003; ICES, 2016). While fishing mortality of commercial stocks has declined, several species are still subjected to relatively high fishing pressure (ICES, 2016).

Fishing is not the only driver of change within marine ecosystems. The impacts of environmental variability on marine ecosystems and their components, often acting in synergy with fishing pressure, must also be examined (Heath, 2005a; Cury et al., 2008; Hoegh-Gulderberg and Bruno, 2010; Link et al., 2010).

4.1.2. Climate Variability in the North Sea

Environmental variability takes many forms in the North Sea, influencing distributions, demography and phenology of marine species and communities (e.g. Perry et al., 2005) and recruitment (Cushing, 1995; Köster et al., 2003; Planque et al., 2010), potentially adding greater stress to ecosystems already under pressure (Stenseth et al., 2002; Walther et al., 2002). The identification of environmental trends allows some mitigation for the impacts of environmental variability on ecological indicators, better aiding decision support in relation to fishing pressure.

Temperature plays an important role within marine ecosystems (Hoegh-Gulderberg and Bruno, 2010), and ocean temperatures are increasing on a global scale (IPCC, 2013). In the North Sea the increase in temperature over the past ten years has occurred alongside changes in currents (EEA, 2012; EEA, 2015; ICES, 2016). Under conditions of increased sea surface temperature (SST), some species have been observed to move northward into colder waters (EEA, 2015), potentially impacting production of the entire ecosystem with effects cascading through the food web.

The dominant atmospheric signal in the North Atlantic, and over the entire North Sea basin, is the North Atlantic Oscillation (NAO) (Hurrell, 2005; Pingree, 2005). Within the North Sea the NAO appears to be a good proxy for winter SST and wind strength (Ottersen *et al.*, 2001). Changes in SST associated with the NAO impact phytoplankton production (Reid et al., 1998), and therefore the NAO likely impacts the North Sea ecosystem both directly and indirectly (e.g. Ottersen *et al.*, 2001). The Atlantic Multidecadal Oscillation (AMO) also influences climate variability in the North Sea region (e.g. Belkin, 2009; Knudsen *et al.*, 2011). Positive phases of the AMO are typified by poleward shifts in distributions of marine organisms, while negative phases are typically accompanied by equatorial shifts (Nye *et al.*, 2014).

4.1.3. An Ecosystem Approach to Fisheries

Sole dependence on single-species fish stock assessments is now considered insufficient for successful management of fisheries. A global move towards the implementation of an ecosystem approach to fisheries (EAF) has therefore been made in recent decades (FAO, 2003; Cochrane and De Young, 2008).

Following the development of the Marine Strategy Framework Directive (MSFD) (EC, 2008) in Europe, the EAF has become enshrined in law. Fisheries should now be managed in line with environmental objectives and evaluated alongside indicator assessments. Similarities exist between the MSFD approach and the approach adopted in this paper, with indicators utilised by IndiSeas complimenting those adopted in the MSFD (see Appendix 3 – Table A3.1). Similar indicators have also been included in ecosystem assessments and cross-ecosystem comparisons in marine systems around the world (e.g. Samhouri *et al.*, 2013; Tam

et al., 2017). Such approaches aim at applying an ecosystem based approach to managing human activities, enabling the sustainable use of marine goods and services.

Here we diverge from the MSFD approach typically implemented in the North Sea, and employ a framework aimed at implementing a novel, comparative approach across marine ecosystems, including systems less data-rich than the North Sea, which draws heavily on research survey-based indicators. Our framework, developed for IndiSeas and based on that developed in Bundy *et al*., (2010), has been successfully applied to two marine ecosystems with characteristics of warm-temperate shelf and upwelling ecosystems (Lockerbie *et al.*, 2016; Lockerbie *et al.*, 2017). The application of the same framework to the North Sea, a cold-temperate shelf ecosystem, should not only allow for a comparison with results obtained from the MSFD indicator framework, but also aid the development of a comparative approach applicable across the numerous ecosystems included in the IndiSeas project (Blanchard *et al.*, 2010; Coll *et al.*, 2010; Shin *et al.*, 2010). Therefore, this framework should help to bridge the gap between global-scale (IndiSeas) and regional-scale (EU-MSFD) indicator syntheses for ecosystem status and trend.

4.1.4. Decision Support

Indicators have been recognised as useful tools in aiding the implementation of an EAF and it has been argued that trends in indicators may be sufficient to support decision making in fisheries management (Jennings and Dulvy, 2005; Trenkel *et al.*, 2007; Möllmann *et al.*, 2013). Here the direction and significance of indicator trends are assessed, rather than using indicator reference values (e.g. in Link *et al.*, 2002; Sainsbury and Sumaila, 2003b; Samhouri *et al.*, 2013; Tam *et al.*, 2017). As we are aware of the vast differences between ecosystems,

we anticipate the need to "fine-tune" the method described in Lockerbie *et al.* (2016) to apply the framework to multiple ecosystems.

Decision trees can be used as a form of multicriteria decision analysis, a method widely used in fisheries and resource management (e.g. McDaniels, 1995; Mardle and Pascoe, 1999; Jarre *et al.*, 2008; Bundy *et al.*, 2010; Dunn *et al.*, 2011; Hazen *et al.*, 2016). They allow the consolidation of trends in numerous indicators to analyse overall ecosystem trends across a period. The way in which decision trees were formulated here permits the sequential analysis of the impacts of both fishing pressure and environmental variability on ecological indicators. The inclusion of information from literature reviews and ecosystem experts enables trends to be more fully interpreted.

4.2. Methods

4.2.1. Ecological, Fishing and Environmental Indicators

To be useful in the implementation of an EAF, ecological and fishing pressure indicators must be capable of tracking state or trends of ecosystem components that may be negatively affected by fishing. Indicators were selected from those utilised in the IndiSeas project and which have been employed to determine the impacts of fishing on ecosystems around the world (e.g. Shannon et al., 2010; Shin and Shannon, 2010; Bundy et al., 2012; Shin et al., 2012; Coll et al., 2016). The indicators were divided into ecological indicators, which identify changes within ecosystem components, and fishing indicators, which identify different aspects of fishing pressure in the ecosystem (see Chapter 2 – Table 2.2 for full indicator description). IndiSeas indicators are primarily derived from fisheries-independent

surveys, allowing them to better reflect community-level changes than catch-based indicators, that may be confounded by fleet dynamics and under-reporting (Shannon et al., 2014).

Environmental indicators were selected that are capable of tracking environmental variability within the ecosystem. It was necessary to select environmental indicators that directly influenced the ecosystem in question. Following extensive literature reviews on the environmental drivers in the ecosystems, and together with consultation with ecosystem experts, suitable environmental indicators were selected. In this case the indicators selected were the NAO, AMO and SST, with addition of small and large copepod indexes to account for the response of low trophic level organisms to environmental pressures.

The winter NAO index, December to March, was utilised in this study and derived from the public UCAR/NCAR website (www.cgd.ucar.edu/cas/catalog/climind/index.html). Data for the AMO were obtained from the National Oceanic and Atmospheric Administration Centre (NOAA) at www.esrl.noaa.gov/psd/data/correlation/amon.us.data (Kooij *et al*., 2016). SST data were derived from the Met Office Hadley Centre observations data set (HadISST; www.metoffice.gov.uk), through the Met Office Marine Data Bank (MDB) and from the International Comprehensive Ocean-Atmosphere Data Set (ICOADS) (Kooij *et al.*, 2016).

Phytoplankton and zooplankton are strongly influenced by environmental variability. As plankton are not included in the IndiSeas suite of indicators, the abundance of small and large (> 2 mm in length) copepods were included as an additional measure of environmental conditions in this study. The copepod indicators were based on Continuous Plankton Recorder (CPR) data supplied by Sir Alister Hardy Foundation for Ocean Science (Richardson *et al.*, 2006). Of approximately 100 copepod taxa collected within the North Sea, 42 are considered typically small taxa and 62 large taxa. Small copepods are identified and

counted during the 'traverse' stage of analysis where 1/50 of the CPR sample silk is analysed. Large copepods are not subsampled; instead each individual is identified and counted.

4.2.2. Division of Time Periods

IndiSeas data for the North Sea were available from 1983-2010, spanning a range of differing environmental conditions. To identify stable time periods, environmental indicator data were analysed through breakpoint analysis to detect potential structural change points (see Zeileis et al., 2001; Bai and Perron, 2003). Analyses of environmental indicator data were conducted using the "strucchange" package (Zeileis et al., 2001) built for the statistical software program R (R Core Team, 2016). This package provides functions to test for non-stationarity of time-series, and once this is confirmed the optimal position of breaks within the data set can be found, thus detecting environmental shifts that may have occurred within the ecosystem. The optimal position of break is established by reducing the sum of squares, while the optimal number of breaks can be established by reducing information criteria (Zeileis *et al.*, 2001; Bai and Perron, 2003; Verbesselt *et al.*, 2010). In this way, significant shifts ($p < 0.05$) in the environmental data can be detected.

Significant shifts in environmental data were detected in the mid-1990s and early 2000s (see Appendix 3 – Section A.3.6. The shift observed around 2002 also coincides with known changes in management measure implemented in this year. Based on these points, three periods were defined: 1983-1992, 1993-2003 and 2004-2010. While the division of the data series into time periods will undoubtedly influence the detection of indicators trends, such a division ensures all trends in ecological indicators are detected. Following sensitivity analysis (see Appendix 3 – Section A3.5), it was observed that many ecological indicator trends were masked when considering the entire data series as a single period.

However, to detect significant trends in ecological and fishing indicators it was reasoned that a time lag would be required before changes in production would be evident in fish large enough to be sampled by survey gear. A two-year lag to the periods of ecosystem indicators, compared to the environmental indicators, was decided upon (see Appendix 3– Section A3.3). We are aware, however, that various other time lags have been suggested (e.g. Daan et al., 2005) and that the time lags also vary depending on the species considered (e.g. Walters and Kitchell, 2001). However, based on the age at recruitment for many important and abundant species found within the ecosystem (Daan et al., 1990), we believe this lag allowed sufficient time for ecosystem components to react. Alongside this, this is sufficient time for species to respond to environmental and low trophic level changes.

4.2.3. Indicator Trend Analysis

Trends in indicators were identified using simple linear regressions. IndiSeas indicators are formulated in such a way that a suite of indicators with overwhelmingly negative trends ought to reflect an ecosystem that is being strongly impacted by fishing, while positive trends should indicate a recovery (Shin *et al.,* 2010a). Previous indicator studies have made use of a rule-based approach (e.g. Bundy *et al.*, 2010); however, here we follow the score based approach developed in Lockerbie *et al.* (2016), which attributes a score to each indicator. Scores were appointed based on trend direction and whether the slope of the regression was significantly different from zero. Indicators were assigned one five possible scores: highly significant positive trend ($p < 0.05$) = 1; ecologically significant positive trend ($0.05 \leq p < 0.10$) = 2; no trend ($p \geq 0.10$) = 3; ecologically significant negative trend ($0.05 \leq p < 0.10$) = 4; highly significant negative trend ($p < 0.05$) = 5 (Lockerbie *et al.*, 2016). The term

"ecologically significant" has been adopted to describe marginally important trends, which could reflect ecological changes within the ecosystem and thus should not be dismissed

The indicators of the impacts of fishing pressure (see Table 2.1) were combined to create an indicator of overall fishing pressure (Table 4.1). In the case of the North Sea, a long term decrease in landings can be attributed to management restrictions on quotas (Total Allowable Catch) that can be landed, rather than a decrease of fish within the ecosystem. This highlights the possibility of misinterpreting trends in fishing indicators due to the way in which they are formulated, emphasising the need for in-depth knowledge of the ecosystem to successfully interpret indicator trends (Shannon *et al.*, 2014a). As trends in landings likely reflected the impacts of management measures better than the impacts of fishing pressure, the previous weighting system (see Lockerbie *et al.*, 2016) was adjusted. As the decrease in landings represented a positive change it was necessary to alter the score for the inverse fishing pressure indicator from 5 (negative trend suggesting negative impact on the ecosystem) to a score of 1 (positive trend suggesting positive impact on the ecosystem). A 50% weighting was attributed to this indicator, with the other four fishing pressure indicators receiving a combined weighting of 50%. By adjusting scores in this way the overall fishing pressure score more accurately reflects what is known to have occurred within the ecosystem (ICES, 2007).

Coll *et al.* (2016) noted that correlated indicators may be somewhat redundant. In Lockerbie *et al.* (2016) and Lockerbie *et al.* (2017) mean lifespan and the proportion of predators significantly correlated over all periods and thus were less heavily weighted to reduce their contribution to the overall ecosystem score. However, in the North Sea correlations were observed between a great number and variety of indicators across the three periods (see Appendix 3 – Tables A3.2a, b & c). Therefore, the decision was made not to differently

weight correlated indicators in this study but rather to weight each ecological indicator equally.

Table 4.1.: Trends in all fishing pressure indicators and overall fishing pressure scores over all periods for the North Sea. p values show the significance of trends and slope values indicate trend direction. Significant trends are highlighted in bold.

INDICATOR	PERIOD 1 (1983-1992)	PERIOD 2 (1993-2003)	PERIOD 3 (2004-2010)
LANDINGS	p = 0.285 Slope = 0.033 Score = 3	**p = 0.042** **Slope = -0.167** **Score = 5** **Adjusted score = 1**	**p = 0.036** **Slope = -0.244** **Score = 5** **Adjusted score = 1**
INVERSE FISHING PRESSURE	**p = 0.018** **Slope = 0.165** **Score = 1**	**p = 0.003** **Slope = 0.384** **Score = 1**	p = 0.751 Slope = 0.062 Score = 3
MTI		**p = 0.003** **Slope = -0.294** **Score = 5**	p = 0.078 Slope = 0.176 Score = 2
TL LANDINGS	**p = 0.003** **Slope = -0.187** **Score = 5**	p = 0.600 Slope = 0.065 Score = 3	p = 0.219 Slope = -0.144 Score = 3
IVI		p = 0.358 Slope = -0.108 Score = 3	**p = 0.007** **Slope = 0.498** **Score = 1**
OVERALL SCORE	3	2	1.62

Following the method developed in Lockerbie *et al.* (2016), ecological indicator scores were sequentially adjusted by predetermined factors to account for the impacts of fishing pressure and environmental variability (Figure 4.1). At this point, due to the complex nature of marine ecosystems, the analysis of indicators was aimed at determining whether environmental variability could have influenced the ecosystem to an extent that would impact ecological indicator trends, not whether the impact would be positive or negative. For this reason, the adjustment of scores has been formulated to mitigate for the impact of environmental variability. Following the adjustment of indicators, the ecosystem received an overall ecosystem score for each period (Table 4.2).
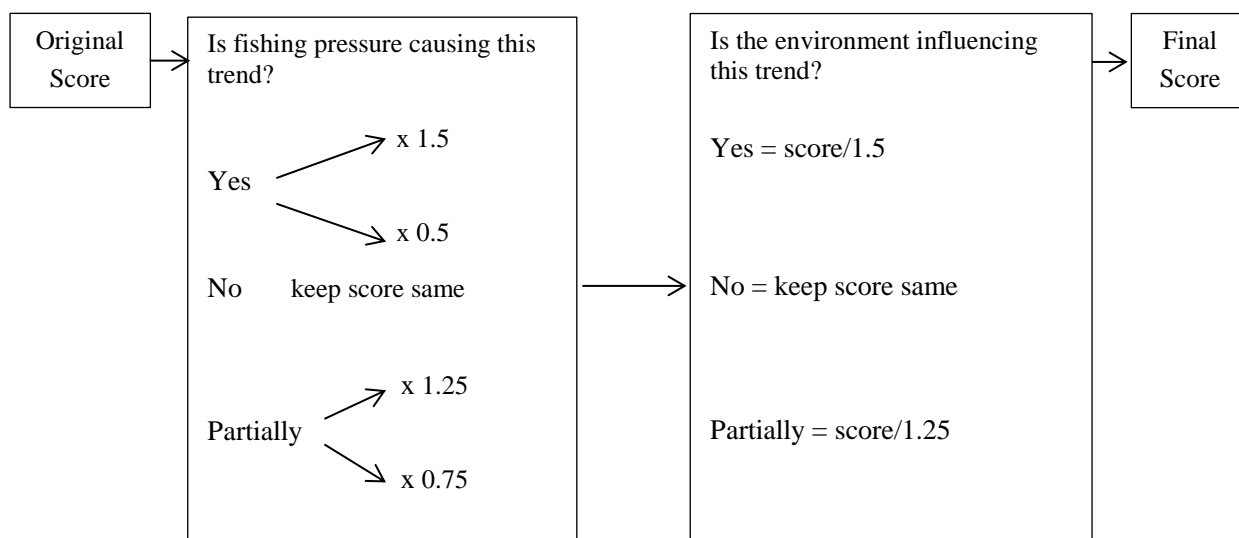
Figure 4.1: Adjustment of original scores attributed to ecological indicators, following Lockerbie *et al.* (2016). Original scores were multiplied by the appropriate factor to give an intermediate score, depending on the impact (positive / negative as reflected by the arrows) and trend of fishing pressure. The intermediate score was then adjusted again by multiplying by the appropriate factor depending on whether environmental variability could influence this trend to result in the final score for the indicator in question. Finally, the score is adjusted to account for the weighting of correlated and non-correlated indicators (adjustment is ecosystem specific) (adapted from Lockerbie *et al.* (2016)).

Table 4.2: Overall ecosystem scores (after sequential application of weightings to account for the influences of fishing pressure and environmental variability) and corresponding ecosystem categories for classification (from Lockerbie *et al.*, 2016).

Overall Ecosystem Score	Categorisation
1-1.49	Improving
1.5-2.49	Possibly Improving
2.5-3.49	No Improvement or Deterioration
3.5-4.49	Possible Deterioration
4.5-5	Deteriorating

4.3. Results

4.3.1. Period 1 (1983–1994)

Analysis identified a lack of trends in most ecological indicators over this period (Figure 4.2), likely resulting from declines of some species while others increased. Only the trophic level of the surveyed community (TLsc) indicator showed a significant trend, increasing over this period. Although fishing pressure did not show a significant trend, it is known to have remained high during this time. However, this does not explain the concurrent increase in the TLsc indicator, which suggested an increase in larger species or size classes. In terms of environmental variability, the only significant environmental indicator trend over this period was an increase in SST (Table 4.3). As no significant trends were observed in the other environmental indicators, it was decided that it was unlikely that environmental variability would cause the observed trends. Ecological indicator scores were, therefore, not adjusted to account for environmental variability (see Table 4.4).

Table 4.3.: Trends in all environmental indicators over all periods for the North Sea. p values show the significance of trends and slope values indicate trend direction. Significant trends are highlighted in bold.

INDICATOR	PERIOD 1 (1983-1992)	PERIOD 2 (1993-2003)	PERIOD 3 (2002-2010)
LARGE COPEPODS	p = 0.517 Slope = 0.052	**p = 0.017 Slope = 0.054**	p = 0.506 Slope = 0.098
SMALL COPEPODS	p = 0.642 Slope = 0.052	**p < 0.001 Slope = -0.086**	**p = 0.095 Slope = -0.094**
NAO (WINTER)	p = 0.456 Slope = 0.072	p = 0.102 Slope = -0.038	p = 0.330 Slope = -0.131
AMO	p = 0.708 Slope = 0.028	**p = 0.000 Slope = 0.091**	p = 0.741 Slope = 0.026
SST	**p = 0.046 Slope = 0.195**	**p = 0.000 Slope = 0.086**	**p = 0.098 Slope = -0.113**

Ecological Indicators	Fishing Indicators	Environmental Indicators
Categorisation (direction and significance) of trends in ecological indicators:	Given the trend in overall fishing pressure, [1] [2] [**3**] [4] [5] can fishing pressure explain ecological indicator trends?	Given trends in environmental indicators (see Table 4.3) can environmental variability explain ecological indicator trends?
Mean length: [1] [2] [**3**] [4] [5] →	Yes / Partially / No[i] →	Yes / Partially / No[iii]
Mean lifespan: [1] [2] [**3**] [4] [5] →	Yes / Partially / No[i] →	Yes / Partially / No[iii]
Survey biomass: [1] [2] [**3**] [4] [5] →	Yes / Partially / No[i] →	Yes / Partially / No[iii]
Proportion of Predators: [1] [2] [**3**] [4] [5] →	Yes / Partially / No[i] →	Yes / Partially / No[iii]
Trophic level of surveyed community: [**1**] [2] [3] [4] [5] →	Yes / Partially / No[ii] →	Yes / Partially / No[iii]

Figure 4.2: Decision tree of ecological indicator trends for Period 1 (1978-1993) and the sequential impacts of fishing pressure and the environment on these indicators. Scoring categories: 1= highly significant positive trend (p<0.05), 2= ecologically significant positive trend (0.05≤ p <0.10), 3= no trend (p = ≥0.10), 4= ecologically significant negative trend (0.05≤ p <0.10), 5= highly significant negative trend (p<0.05). See foot notes for details on how the observed fishing pressure and environmental trends impact each indicator.

Overall fishing pressure does not show a significant trend over this period and therefore cannot explain trends in mean length, mean lifespan, survey biomass or the proportion of predators.

ii A significant increase in the TLsc would not be expected under a situation of no significant trend in overall fishing pressure and, therefore, the overall fishing pressure indicator cannot explain the observed trend.

iii There is a lack of the trend in environmental indicators over this period, with the exception of a significant increase in SST. As the trends in other environmental indicators which are typically associated with increases in SST (e.g. trends in NAO and AMO) we suggest that it is unlikely that the environment can explain trends in ecological indicators over this period.

4.3.2. Period 2 (1995-2003)

Overall fishing pressure was observed to decrease, accompanied by a highly significant increase in survey biomass and an ecologically significant decrease in the TLsc (Figure 4.3). However, it is important to note that there was no significant trend in the TLmc, an indicator which contains a greater number of species than those based on survey data. Changes in community structure may have occurred due to density compensation amongst species, despite a consistent overall modelled biomass in the ecosystem as a whole. Therefore, increases in a greater number of indicators may have been expected as the ecosystem should have shown some recovery over this time.

Several significant trends in environmental indicators were observed over this period (Table 4.3). The small copepod index decreased significantly, accompanied by an increase in the large copepod index, AMO index and SST. The combined impacts of a significant decrease in the small copepod index and increase in SST over this period are expected to have had important ecological impacts on indicator trends. However, due to the perturbed state of the ecosystem, it is unlikely that the full effect of such environmental variability would be picked up by ecological indicator trends. Therefore, the decision was made to adjust the ecological

indicator score in a way which suggests the environment can only partially explain the observed trends.

4.3.3. Period 3 (2004-2010)

Under decreased fishing pressure, during this period and the previous one, some recovery of larger fish species within the ecosystem should be apparent. This was reflected in increases in mean lifespan and TLsc indicators (Figure 4.4). However, some unexpected trends are also observed; there was a highly significant decrease in mean length and an ecologically significant decrease in the proportion of predators, possibly due to high recruitment and an increase in non-predatory species. However, it is important to note that although there was a decline in the proportion of predators, the mean proportion of predators for Period 3 was still significantly higher than the mean in Period 1 (see Appendix 3 - Figure A3.4d), possibly suggesting some recovery.

Over this Period significant decreases in SST and the small copepod index were observed. However, despite cooling SST in the third period, the warming SST observed in the preceding Periods 1 and 2 may have hampered species' recoveries. A decrease in the small copepod index over this period potentially influenced the ecosystem as copepods are an important prey for many fish within the ecosystem.

Figure 4.3: Decision tree of ecological indicator trends for Period 2 (1994-2003) and the sequential impacts of fishing pressure and the environment on these indicators. Scoring categories: 1= highly significant positive trend (p<0.05), 2= ecologically significant positive trend ($0.05 \leq p < 0.10$), 3= no trend (p = ≥ 0.10), 4= ecologically significant negative trend ($0.05 \leq p < 0.10$), 5= highly significant negative trend (p<0.05). See foot notes for details on how the observed fishing pressure and environmental trends impact each indicator.

[i] There was an ecologically significant decrease in fishing pressure over this period. Therefore, an increase in mean length and mean lifespan might be expected over this period as the ecosystem begins to recover.

[ii] An increase in survey biomass would be expected under a situation of decreasing fishing pressure, potentially suggesting some recovery of species within the ecosystem. However, as there was an ecologically significant decrease in the TLsc over this period it is likely that this increase signified an increase of low trophic level species, as can be seen in the significant increase in large copepods.

[iii] There was an ecologically significant decrease in fishing pressure over this period. Therefore, an increase in survey biomass may be expected. The lack of trend in this indicator may be due larger, predatory fish species taking longer to recover when fishing pressure is reduced, due to their life histories.

[iv] There was an ecologically significant decrease in fishing pressure over this period. Therefore, a decrease in the TLsc would not be expected. As this trend was accompanied by a significant increase in survey biomass it is likely that an increase in low trophic level species caused this trend in TLsc.

[v] There was an ecologically significant decrease in fishing pressure over this period. Therefore, an increase in the TLmc may be expected as the ecosystem begins to recover. The contrast between the trends in TLmc and TLsc was likely a result of the fact that a greater number of species are included in the modelled community than are collected in surveys.

[vi] There were significant trends in several environmental indicators over this period, including significant decrease the small copepod index and significant increases in large copepods, the AMO index and SST. However, it is likely that management methods implemented in the ecosystem over this period were affecting trends in ecological indicators more strongly than environmental impacts, for most species. Therefore, it has been suggested that the environmental indicators can partially explain the trends observed in ecological indicators.
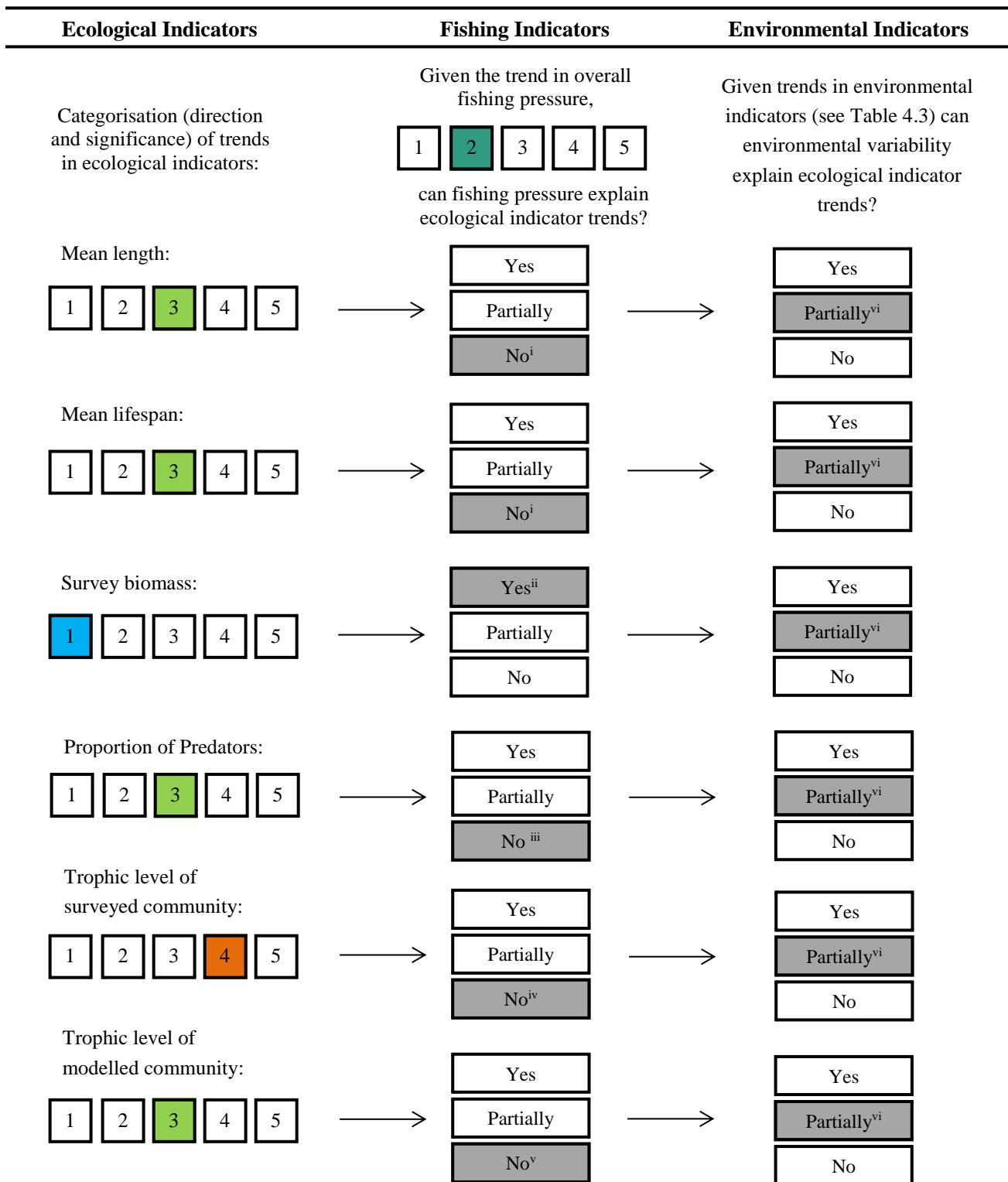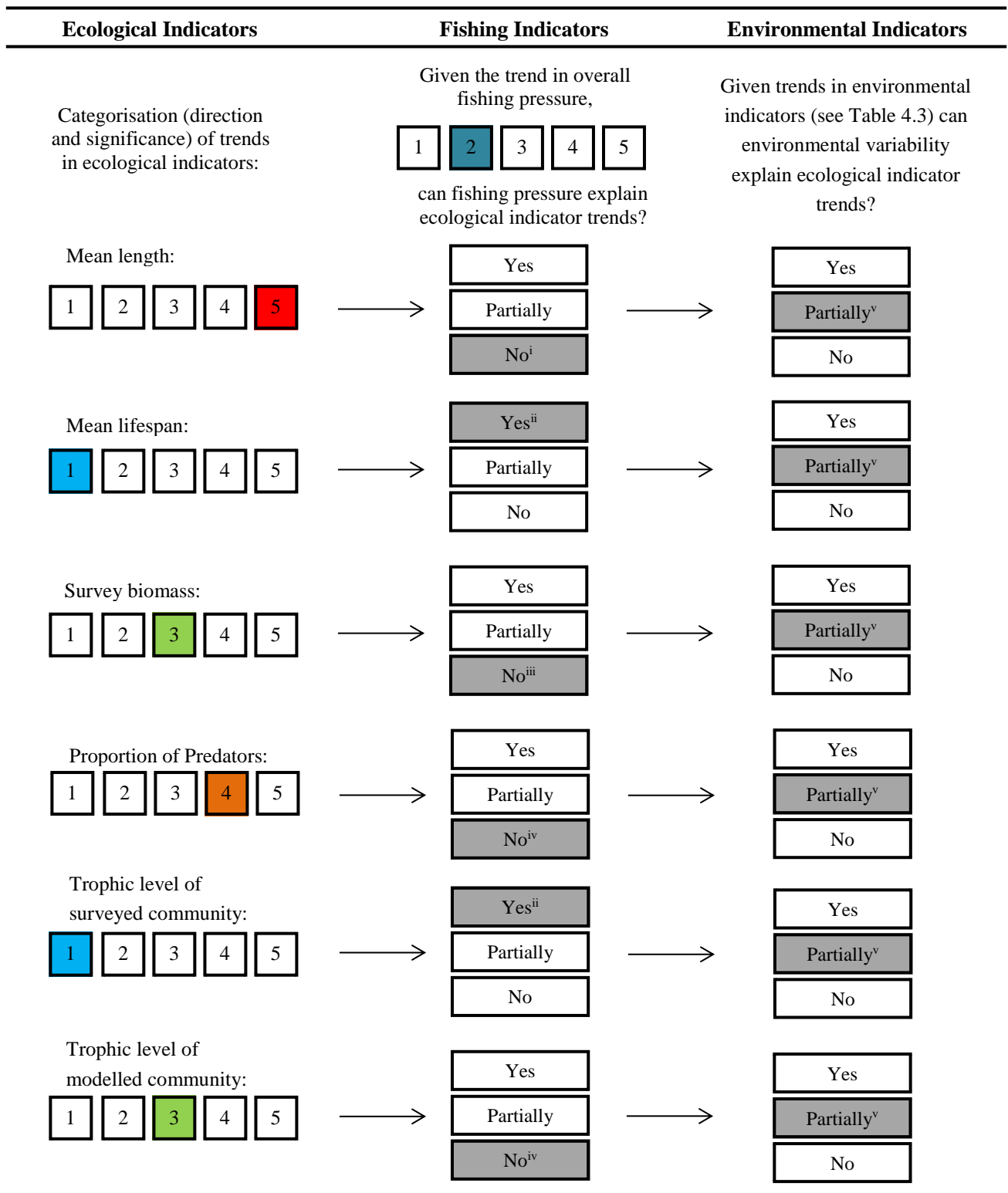
Figure 4.4: Decision tree of ecological indicator trends for Period 3 (2004-2010) and the sequential impacts of fishing pressure and the environment on these indicators. Scoring categories: 1= highly significant positive trend (p<0.05), 2= ecologically significant positive trend (0.05≤ p <0.10), 3= no trend (p = ≥0.10), 4= ecologically significant negative trend (0.05≤ p <0.10), 5= highly significant negative trend (p<0.05). See foot notes for details on how the observed fishing pressure and environmental trends impact each indicator.

[i] There was an ecologically significant decrease in overall fishing pressure over this period. Under decreasing fishing pressure, a significant decrease in the mean length would not be expected.

[ii] There was an ecologically significant decrease in overall fishing pressure over this period. Under decreasing fishing pressure an increase in mean lifespan and TLsc would be expected as the ecosystem begins to recover.

[iii] There was an ecologically significant decrease in overall fishing pressure over this period. Under decreasing fishing pressure, an increase in survey biomass may be expected as the ecosystem begins to show some recovery.

[iv] Under decreasing fishing pressure a decrease in the proportion of predators would not be expected. Considering the decreasing trend in overall fishing pressure observed even predatory species should have showed some recovery by this time. However, although the proportion of predators was decreasing over this period the mean for the period was higher than that observed in the first period (see Appendix – Figure 3d), potentially suggesting some recovery of predatory fish.

[iv] There was an ecologically significant decrease in overall fishing pressure over this period. Therefore, the increase in TLmc may have reflected some recovery within the ecosystem. The contrast between this indicator trend and the increasing trend observed in the TLsc likely resulted from a greater number of species being included in the modelled community than are collected in surveys.

[v] There were significant decreasing in the small copepod index and in SST over this time period. However, similarly to the previous period, it is likely that management methods implemented in the ecosystem were affecting trends in ecological indicators more strongly than environmental impacts, for most species. Therefore, it has been suggested that environmental indicators can partially explain the observed ecological indicators trends.

Considering the potential impacts of fishing pressure and environmental variability over this period, scores were adjusted accordingly. Again, however, due to the perturbed state of the ecosystem and the fact that the full impacts of environmental variability would likely not be picked up by ecological indicator trends, it has been suggested that environmental variability could only partially explain ecological indicators trends (see Table 4.4).

Table 4.4: Summary of ecological indicator scores showing adjustments for the impacts of fishing pressure and the environment (detailed in Figure 4.2, 4.3 and 4.4). Final ecosystem scores for classification are calculated from the weighted means of adjusted scores. Ecosystem is classified as neither improving nor deteriorating in Period 1, and as possibly improving in Periods 2 and 3.

Indicator	Period 1					Period 2					Period 3				
	Original Score	Fishing pressure trend - Does fishing pressure explain trend?	New score	Can trends in environmental indicators explain trend?	Final Score	Original Score	Trend - Can fishing pressure explain trend?	New score	Can trends in environmental indicators explain trend?	Final Score	Original Score	Trend - Can fishing pressure explain trend?	New score	Can trends in environmental indicators explain trend?	Final Score
Mean Length	3	No trend	3	No	3	3	No – Decreasing	3	Partially	2.4	5	No – Decreasing	5	Partially	4
Mean Lifespan	3	No trend	3	No	3	3	No – Decreasing	3	Partially	2.4	1	Yes – Decreasing	0.5	Partially	0.4
Survey Biomass	3	No trend	3	No	3	1	Yes - decreasing	0.5	Partially	0.4	3	No – Decreasing	3	Partially	2.4
Proportion of Predators	3	No trend	3	No	3	3	No – Decreasing	3	Partially	2.4	4	No – Decreasing	4	Partially	3.2
TLsc	1	No trend	1	No	1	4	No - decreasing	4	Partially	3.2	1	Yes – Decreasing	0.5	Partially	0.4
TLmc						3	No – Decreasing	3	Partially	2.4	3	No – Decreasing	3	Partially	2.4
Mean Score	2.6				2.6	2.8				2.2	2.8				2.1

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4.4.Synthesis and Outlook

Over the first period (1983-1994) the North Sea ecosystem was classified as neither improving nor deteriorating, receiving an overall score of 2.6. This categorisation results from the lack of trends in most ecological indicators, potentially arising from species substitutions. For example, Heath (2005) observed substitutions in piscivores landings, where, when herring collapsed in the late 1970s, sprat landings increased, although substitutions were not observed in the demersal component. Heath (2005) therefore suggested that the pelagic component of the ecosystem is controlled by production processes, while the demersal component is controlled by predation. A suggested 10-year time lag between fishing pressure and change in demersal species composition and size structure could also explain the lack of measured trends.

Historically high levels of fishing have reduced the number of age groups in some fish stocks, resulting in fishing success becoming more dependent on recruitment (ICES, 2003). During this period, fishing activity removed an average of 30-40% of total biomass of exploited commercial species each year (Gislason, 1994). For species such as cod (*Gadus morhua*) this may imply that more than half the exploitable population at the beginning of the year was caught during the year (Gislason, 1994). Reductions in spawning stocks through fishing pressure are known to have resulted in declines in recruitment (Dickey-Collas *et al.*, 2010). A strong negative response of indicators may therefore have been expected over this period. However, the relatively short length of the periods used in this study may have masked trends in ecological and environmental indicators expected in the longer term.

Despite the lack of trends in the majority of indicators, when compared to the final period, some longer-term trends can be observed. The proportion of predators and inverse fishing pressure were significantly lower in the first period compared to Period 3 (see Appendix 3 -

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Figure A3.4d), suggesting a greater impact of fishing pressure at the beginning of the time series. Again, this would be expected given higher fishing pressure over this period and the stringent fisheries management measures implemented since 2002. Analysis of the entire data set also showed a possible improvement of the ecosystem over time (see Appendix 3 – Section A3.5), and an overall decline in the impacts of fishing pressure from 1983-2010.

The second (1995-2003) and third periods (2004-2010) were both categorised as possibly improving, a result that was anticipated following implementation of stricter management within these periods. A significant change in management occurred in Period 2, with the Common Fisheries Policy undergoing a considerable reform in 2002. This aimed at reducing fishing pressure on over-exploited stocks and introducing recovery plans to protect healthy stocks. A significant increase in survey biomass and decrease in TLsc over this period may indicate a recovery of small fish. This is supported by Minto and Worm (2012) who suggested smaller species recover first when fishing pressure is reduced. Further, following heavy exploitation, increases in small fish biomass generally surpass declines in biomass of large species (Greenstreet *et al.*, 1999; Blanchard *et al.*, 2005; Daan *et al.*, 2005).

Considering the peak in fishing pressure in the North Sea in the mid-1980s, and subsequent decline, the lack of trends in most other indicators over Period 2 was somewhat surprising (Daan *et al*., 2005). The lack of trends may have resulted from the previously high levels of exploitation on the ecosystem, or from high levels of fishing pressure over this period despite the decreasing trend observed in overall fishing pressure. Declines in some species, despite stringent management measures, may also have contributed to the lack of indicator trends observed over this period (ICES, 2003). For example, further declines in herring were observed in the mid-1990s. This was followed by the implementation of effective management methods for herring fisheries in 1996 (ICES, 2003), which resulted in a substantial decline in fishing mortality between 1996 and 2001.

While it is well known that fisheries impact the ability of marine ecosystems to cope with climatic variability, in this period the highly managed nature of the North Sea fisheries likely masked some environmental impacts (Lynam *et al.*, 2017). Kenny *et al.* (2009) suggested that the pelagic component of the ecosystem may be more greatly controlled by the environment than fishing pressure over this period. However, it was also noted that connection between the pelagic and demersal components of the ecosystem was still strong during Period 2, suggesting the influence of fishing pressure remained high (Kenny *et al.*, 2009).

Unfortunately, IndiSeas indicator values for the North Sea are currently only available up to 2010. The calculation of indicator values for more recent years may have allowed a greater number of positive trends to be observed, better reflecting the recovery of this ecosystem. Over the third period the increases in TLsc and mean lifespan observed may indicate increases in important species such as herring and plaice. The decline in the proportion of predators over this period was unexpected under decreasing fishing pressure, particularly following the restriction of total allowable catch of many North Sea fish species since 2000 (ICES, 2008). Within the North Sea the large fish indicator is often used when assessing ecosystems (ICES, 2007; Greenstreet *et al.*, 2011). This indicator was observed to be relatively stable over this period (Greenstreet *et al.*, 2011). However, the proportion of predators, mean length and mean lifespan are substitutes for this included in the IndiSeas project. As in the previous period it was possible that a decrease in mean length was related to an increase in small fish, also reflected in the decrease in the proportion of predators. While some recovery of larger fish species over recent years has been documented (ICES, 2016; ICES, 2017), many remain at low levels within the ecosystem and thus their increase may not reflect in ecosystem-level indicator trends. This species is also known to be negatively impacted by warming sea temperatures (O'Brien *et al.*, 2000).

Determining the extent to which the North Sea has altered is a complicated process as data have only been collected under situations of high fishing pressure. However, our study suggested possible improvement of the North Sea ecosystem over the periods examined from 1983 to 2010. Tried and tested indicators, such as those developed in the IndiSeas project, are important for the development of useful decision trees. The use of such indicators, accompanied with information from ecosystem experts and literature, allows decisions trees to aid stakeholders and fisheries managers in the decision-making process. Decision trees can then be computerised to develop expert systems, containing large amounts of information describing reasoning behind observed trends. Such systems are created in a way that makes the information easily accessible and understood by fisheries managers and stakeholders. However, the development of expert systems will need to be a dynamic process that evolves as knowledge of the ecosystem and ecosystem processes improves (Jarre *et al.*, 2006b; Jarre *et al.*, 2008).

Application of the framework utilised in this study has now been successfully applied in three marine ecosystems of varying type, exploitation history, climatic variation and management strategy; a shelf system (North Sea, this study), a shelf system with upwelling characteristics (South Catalan Sea; see Lockerbie *et al.* 2017) and a true upwelling system (Southern Benguela; see Lockerbie *et al*. 2016). Successful application relies on local scale, data-based, ecological and environmental indicators, and has proven the strength and efficacy of the framework in assessing state and trends of marine ecosystems within a global comparative context. The application of the IndiSeas suite of indicators, which are readily available for many less data-rich ecosystems, together with inputs and interpretative insights from local experts, will aid the implementation of this approach to a wide array of marine systems. The strength of the framework can therefore be further tested on ecosystems with various

histories, undergoing varying levels of exploitation and exposed to diverse effects of climate variability.

Chapter Five

Synthesis and Conclusions

5.1. Thesis Overview

5.1.1. Methodological Approach

The Ecosystem Approach to Fisheries (EAF) has faced many difficulties in its implementation in management. Despite global interest in this approach over recent decades, its formalisation in 2001 (FAO, 2003), and widespread approval at the World Summit for Sustainable Development in Johannesburg in 2002, there has been limited success in its application to real world fisheries. This is, in part, due to the complexity of processes in ecosystems, and imperfect knowledge of such complexities (e.g. Drakou *et al.*, 2017).

The focus of this thesis was to build upon the work conducted by the IndiSeas project, creating a decision tree framework to synthesise indicator trends in a repeatable, transparent manner (Coll *et al.*, 2009; Bundy *et al.*, 2010; Shannon *et al.*, 2014a). The framework developed here allows for the sequential analysis of indicator trends to determine the impacts of fishing and environmental pressure on each ecological indicator, concluding with a score describing the overall ecosystem trend. A key feature of the approach developed was the use of knowledge from ecosystem experts and extensive literature reviews, in addition to maintaining flexibility for further modification due to theoretical and practical considerations (e.g. Jarre *et al.*, 2008; McGregor, 2015). The framework was developed and applied to each ecosystem with the inclusion of advice from regional experts at every step. These experts provided local knowledge, such as potential biases in the official data, ensuring that the framework could be appropriately adjusted and that the results of indicator trend analysis could be better interpreted. This helped to ensure that the framework was robust and

meaningful in the context of global comparisons (Shin *et al.*, 2012), which is particularly important for the future application of this framework to other marine ecosystems. The ultimate goal was to allow more comprehensive interpretation of indicator trends to provide sound decision support to fisheries stakeholders and managers.

The decision tree framework developed in this study was based on that developed in Bundy *et al.* (2010). Making use of a smaller suite of indicators, Bundy *et al.* (2010) decided on the use of decision trees with the adoption of a rule-based method to classify marine ecosystems as improving, stationary or deteriorating based on indicator trends. Their approach to classifying ecosystems was very precautionary, with ecosystems being classified as deteriorating if a minimum of one or two indicators showed a negative trend, depending on the decision rule being adopted. An ecosystem could only be considered improving if no indicators showed negative trends, and at least three indicators showed a positive trend.

This thesis developed a score-based approach which more readily facilitates the synthesis of large sets of indicators than a rule-based approach. This new decision tree framework was designed making use of data series from the Southern Benguela ecosystem. Although IndiSeas ecological indicators are formulated in a way which allows them to respond to increases and decreases in fishing pressure, it has been suggested that indicators would also be influenced by other drivers, such as pollution, the economy and especially the environment (e.g. Walther *et al.*, 2002; Islam and Tanaka, 2004), and that non-linearity may well come into play (Bundy *et al.*, 2010). A key step in the development of an appropriate assessment of indicators for the application of an EAF was, therefore, the inclusion of the suites of environmental indicators that provide the necessary context in which to interpret observed ecological indicator trends.

As an upwelling ecosystem, the Southern Benguela is strongly influenced by environmental drivers, particularly as upwelling is wind driven. The correct interpretation of ecological indicator trends is therefore very dependent on understanding the importance of the environment (Shannon *et al.*, 2010). The framework developed in this thesis includes a weighting system by which the scores of ecological indicators could be adjusted based on the impacts of fishing pressure and environmental variability. In addition to this, weightings were applied to decrease the contribution of correlated indicators to the overall ecosystem score. Sensitivity analyses conducted for the case study ecosystems (e.g. Appendix 1 – Section A1.2) made it possible to ascertain the robustness of this approach. The methods described in this thesis have been applied to several ecosystems to assess and strengthen the framework, by looking at systems with varying levels of exploitation, fishing histories and management strategies to identify the relative state of each ecosystem (e.g. Blanchard *et al.*, 2010; Bundy *et al.*, 2010).

The aim of this study was to determine whether a generic decision tree could be developed, that could successfully categorise each ecosystem. It was, however, acknowledged from the start of the study that due to the huge variations between marine ecosystems in terms of both current and historical fishing pressure, the impacts of climate variability and species compositions and interactions, decision trees may need to be 'fine-tuned' before they can be successfully applied. It is unlikely that a 'one size fits all' approach of applying decision trees will be appropriate to accurately assess multiple marine ecosystems. This could already be seen in the need to select environmental drivers applicable to individual ecosystems to better assess the influence of environmental variability. It was therefore anticipated that the weighting system mentioned above may also need to be adjusted to be applicable to each ecosystem. The selection of ecosystems of varying size, species composition, fishing pressure and environmental drivers was aimed to help determine to what extent a generic approach is

appropriate. As indicated by the Mediterranean and North Sea case studies considered in this thesis, the framework is sufficiently flexible to allow accommodation for the characteristics of individual data series through thoughtful, well-reasoned, change of score direction or adjustment of weights.

This approach is less precautionary than that described in Bundy *et al.* (2010) and considers all trends within a larger set of indicators before classifying the ecosystem. Notably, the development of this framework, as part of the second phase of the IndiSeas project, also includes the addition of a suite of environmental drivers. While environmental indicators have recently been included in studies conducted as part of the IndiSeas project (e.g. Fu *et al.*, 2015), this step had not yet been included in decision tree frameworks to formally assist with the interpretation of observed trends in ecological indicators.

5.1.2. Case Study 1: The Southern Benguela

The Southern Benguela is under wasp-waist control, with small pelagic fish having a dominant role in trophic flows between high and low trophic level species (Cury *et al.*, 2000). The processes within the ecosystem become more complex when considering that climatic and fishing pressure drivers act simultaneously, influencing ecosystem components and therefore trophic flows. In the Southern Benguela the high level of primary production resulting from the upwelling nature of the ecosystem may somewhat dampen the propagation of the impacts of fishing through the food web (Travers-Trolet *et al.*, 2014). It was necessary to divide the data series for the Southern Benguela ecosystem into distinct time periods, based on known regime shifts that have occurred in the Southern Benguela (Blamey *et al.*, 2012). These shifts are known to have resulted in changes in environemental variability within the ecosystem. The periods selected were; Period 1 (1978-1993), Period 2 (1994-2003)

and Period 3 (2004-2010). Trends observed in the Southern Benguela ecosystem successfully reflected species trends that are known to have occurred within the ecosystem, as did the interpretation of trends following the above-mentioned score adjustments. For example, the significant decrease in the trophic level of the modelled community reflected a decline in demersal fish and an increase in small pelagic fish biomass over the first period, as detailed in Shannon *et al.* (2009b). This period was classified as neither *improving nor deteriorating* (Table 2.4). A decrease in mean lifespan in the second period could be linked to a significant but short lived increase in small pelagic fish in the early 2000s (Roy *et al.*, 2001), which was also reflected in the significant increase in survey biomass and decrease in the proportion of predatory fish in the community. These trends were the first to highlight the need to take care when interpreting indicator trends. For example, the decrease in the proportion of predatory fish indicator reflected an increase in small pelagic fish, rather than a decrease in large fish. The well-documented regime shift that occurred in the Southern Benguela in the early 2000s, caused by environmental change but intensified by the influence of fishing pressure (e.g. Roy *et al.*, 2001; Coetzee *et al.*, 2008; Blamey *et al.*, 2012), was likely the overarching reason behind observed ecological trends in the second period, more so than fishing pressure. Over this period the ecosystem was similarly classified as *neither improving* nor *deteriorating*, resulting from the increase in small pelagic species that was observed. Trends in the third period were the most variable, with both significant increases and decreases observed. The impacts of the environment over this period are not fully understood, and it is possible that a further ecosystem shift could be occurring due to increased variability (Blamey *et al.*, 2012; Blamey *et al.*, 2015) and a shift in the South Atlantic High Pressure System (Jarre *et al.*, 2015). Despite this uncertainty, the ecosystem was classified as being *possibly improving* over this period. The classification of ecosystems therefore needs to be a dynamic, continuous process, that will improve as knowledge and data sets for the ecosystem advance.

Methods of time-series analysis, such as employed in the present work, or in (Blamey *et al.*, 2012) have been effective in defining shifts between periods of relative stability, i.e. the timing of shifts.

Within the Southern Benguela there has been no increase in fishing pressure over recent decades, with decreasing overall fishing pressure observed in the first two periods. Therefore, a greater number of positive trends may have been expected. However, the impact of environmental variability may have prevented expected trends from being observed.

5.1.3. Case Study 2: The South Catalan Sea

The South Catalan Sea was selected as the second ecosystem owing to its similarities to the Southern Benguela (Coll *et al.*, 2006a). The region is impacted by localised, wind driven upwelling and is dominated by small pelagic fish. These similarities observed between the Southern Benguela and South Catalan Sea are thought to arise from the inclusion of the Agulhas Bank and the continental shelf (respectively) in the ecosystems. The Mediterranean ecosystem is one of the world's most impacted seas and therefore it is not surprising that the South Catalan Sea has been categorised as "a highly impacted" region (Coll *et al.*, 2009). In this case the interpretation of fishing indicator trends was hampered by the fact that official data does not account for IUU catches. However, the interpretation of trends was greatly aided by the work conducted in Coll *et al.* (2014a) and Coll *et al.* (2014b) that accounted for the impacts of IUU fishing to better understand the impacts of fishing pressure on the ecosystem. This case study highlighted the critical importance of considering the relevant literature and gaining advice from ecosystem experts to accurately interpret the results detected in indicator analyses.

As with the Southern Benguela, changes in environmental conditions were used to distinguish periods within the ecosystem. Changes in atmospheric, hydrological and ecological systems were noted in the late 1980s (Conversi *et al.*, 2010), and so, following sensitivity analysis (Appendix 2 – Section A2.1.) the data series was divided into two periods, 1978-1990 and 1991-2010. When calculating overall fishing pressure, it was necessary to somewhat alter the method utilised for the Southern Benguela. The official data collected for the ecosystem, which were utilised when calculating fishing pressure indicators, did not include information on the presence of illegal, unreported and unregulated (IUU) catch within the ecosystem. If the decision tree framework was applied without adjusting the fishing pressure indicators for the impacts of IUU catch the ecosystem would have been classified as neither *improving nor deteriorating* in the first period and *possibly improving* in the second. This did not align with what is known to have occurred in the ecosystem (Coll *et al.*, 2014a; Coll *et al.*, 2014b). It was found, therefore, that the use of official data sets was insufficient to determine true ecosystem trends. The European Court of Auditors (Court of Auditors, 2007) has acknowledged that it is important to include the influence of IUU catch within the ecosystem and recent literature has suggested an exponential increase in the removal of fish from the ecosystem (Coll *et al.*, 2014b). Therefore, the decision was made to alter the overall fishing pressure indicator score from an 'ecologically significant decrease' to an 'ecologically significant increase'. This was a conservative adjustment considering the suggested exponential increase in removals and allowed indicators trends to more accurately reflect changes in fishing pressure for correct interpretation in the decision tree framework (Coll *et al.*, 2014b). Similar to the Southern Benguela, the proportion of predators and biomass correlated significantly with each other over both periods. The same weighting system was therefore applied to these indicators in the South Catalan Sea case study.

Over the first period (1978-1990) the scores of most South Catalan Sea indicators reflected what would be expected under increasing fishing pressure. Significant decreases occurred in all indicators, except survey biomass and the proportion of predators, which did not show any significant trends. Over the second period (1991-2010) a variety of indicator trends were observed. One important species trend that may have influenced indicator trends was the significant decrease in the biomass of sardine. This likely contributed to the decreasing trends observed in two indicators, namely survey biomass and the trophic level of the modelled community. The lack of trends in the proportion of predators across both periods reflects the depleted nature of the ecosystem in terms of large fish, making this indicator less responsive to the influence of fishing pressure. The ecosystem was classified as possibly deteriorating over the first period, and neither *deteriorating nor improving* over the second period.

Within the South Catalan Sea, increased fishing pressure over both periods was clearly reflected in the decreasing trends in the majority of indicators, and the resultant deteriorating trend in Period 1. The lack of deteriorating classification in Period 2 was somewhat surprising, however the majority of trends can be related to changes in small pelagic fish.

5.1.4. Case Study 3: The North Sea

The final ecosystem selected for this study was the North Sea. This is one of the best studied and most economically important marine ecosystems in the world. This ecosystem was selected due to it having some similarities with the first two systems, such as the temperate climate and the presence of the continental shelf as a defining feature. However, the differences in species and the lack of upwelling dynamics provided enough of a contrast for the ecosystem to act as an additional case study. The North Sea presented a different challenge, due to both the long history of overfishing as well as the impact of recent stringent

management methods that have been implemented. It is generally considered that the North

Sea is controlled by bottom-up forces, with climatic drivers influencing lower trophic level

species, including plankton, planktivorous fish and demersal fish in their pelagic phases

(Beaugrand *et al.*, 2009; Olsen *et al.*, 2010). Bottom-up processes have controlled changes in

abundances of planktonic groups within the system since the 1960s, while exploitation

through fisheries has controlled changes in the biomass of those species which are

commercially exploited (Lynam *et al.*, 2017). Planktivorous fish can therefore play key roles

linking these low and high trophic level species. In the North Sea, the historical and

considerable influence of fishing on higher trophic levels, has limited the propagation of

bottom-up influences through the food web (Lynam *et al.*, 2017). Thus, the impact of climate

on ecosystem components appears to be limited to low trophic levels. Despite these

intricacies, the recognition that the cause of several indicator trends was the impact of

management measures, rather than fishing pressure or environmental drivers, allows indicator

trends to be correctly interpreted.

This ecosystem presented a challenge during analysis as in some cases ecological indicator

trends no longer reflected the impacts of fishing and the environment but rather showed that

trends resulted from the implementation of successful management measures. It was

necessary, therefore, to alter the calculation of the overall fishing pressure indicators to more

accurately represent what was happening within the ecosystem. Due to the stringent

management methods put in place within the ecosystem, landings, rather than the impacts of

fishing pressure, more accurately reflected management measures. The method of

determining time periods based on environmental shifts was improved in the analysis of this

ecosystem. As opposed to making use of literature reviews and sensitivity analysis to

ascertain years in which environmental shifts occurred, here breakpoint analysis was used to

detect potential structural change points (Zeileis *et al.*, 2001; Bai and Perron, 2003), in

extension of the methodology used by Blamey *et al*. (2012) for breakpoints analysis in the Benguela ecosystem. Based on the detected breakpoints the data series were divided into three periods, namely Period 1 (1983-1992), Period 2 (1993-2003) and Period 3 (2004-2010). The method of indicator analysis needed to be adjusted slightly for application to the North Sea. When correlations were conducted on ecological indicators many significant correlations were detected, particularly across the third period. As no indicators could therefore be considered completely redundant (Coll *et al.*, 2016), all indicators were equally weighted. Following this decision, it was necessary to adjust the scoring system and remove the weightings applied to account for potential redundancies that had been applied in the previous two case studies.

Over the first period the ecosystem was classified as neither *improving nor deteriorating*. There was a general lack of trends across this period which was somewhat surprising as fishing pressure was highest during this time. The lack of trends likely reflected the long history of heavy exploitation within the ecosystem. However, when compared to the final period, some indicators, such as the proportion of predators, were significantly lower during this time. Over the second period the ecosystem was classified as *possibly improving*. The observed ecological indicator scores suggested that the ecosystem became dominated by smaller individuals and small species over this period. This may be a sign of some recovery of the ecosystem as an increase in small individuals likely reflected an increase in their recruitment of several commercial species. Although the proportion of predatory fish was observed to decline over this period it remained higher than the first period, supporting the suggestion that some predatory species may have begun to recover. For the final period the ecosystem was also classified as *possibly improving*. This was expected as stringent management methods had been in place for several years (EC, 2008) and resulting significant

increases in both mean lifespan and trophic level of the surveyed community were likely the primary cause of the ultimate classification.

The impacts of fishing on the North Sea are very clear, with improvements starting to be observed in recent years. The impacts of the environment are less clear due to the exploited nature of this ecosystem.

5.1.5. Evaluation and Evolution of the Framework

With the application of the framework to each case study ecosystem adjustments and improvements were made. The application of the framework to new ecosystems included discussions of the framework with new scientific experts, each of whom added their unique knowledge and understanding to the framework. Varying issues arose with the use of the framework in specific ecosystems. The ecosystems experts were able to offer unique insight into how to improve the framework to deal with these matters. These improvements strengthened the framework, and should aid its successful application in further ecosystems.

One of the most notable advancements in the framework was the inclusion of breakpoint analysis in the North Sea ecosystem. The use of statistical methods to divide the ecosystem into separate periods, based on environmental shifts, increases the likelihood that the detected periods are a true reflection of environmental variability. While this approach would not necessarily be needed in the Southern Benguela system, where extensive work has been done on the presence of regime shifts and significant statistical analysis has conducted (see Blamey *et al.*, 2012; Blamey *et al.*, 2015), it would likely have benefitted analysis of the South Catalan Sea. The division of the data set in the South Catalan Sea was based on knowledge gained from the literature, and following sensitivity analyses and guidance from an ecosystem expert. The use of statistical analyses to determine shifts would increase certainty

in the exact points at which the divide the data set. Therefore, the application of formal statistical methods of detecting change in variable, short time series, such as the method of breakpoint analysis employed in the North Sea (Chapter 4), is recommended for use in future applications of the framework to other ecosystems.

However, although breakpoint analysis aids the detection of shifts in environmental indicators, the division of data series into distinct periods has resulted in some of the time-series potentially being too short for clear trends to be determined. It must, therefore, be decided for each ecosystem whether it is appropriate to divide the time series at all, or, whether the detection of trends would benefit from analysis of the entire data set. This concern has somewhat been accounted for in the comparison of indicator values in the first and final periods in the North Sea (Appendix 3, Section A3.4) and which have now been conducted for the Southern Benguela (Appendix 1, Section A1.3) and South Catalan Sea (Appendix 2, A2.6). The identification of significant differences between values in the first and final period allows insight into the overall change in the ecosystem over time. Consideration of longer term trends, alongside those of individual periods when appropriate, may allow a better understanding of what is occurring within ecosystems.

Another imperative development in the framework was the inclusion of time lags in the North Sea ecosystem, which were not included in the analysis of the Southern Benguela and South Catalan Sea ecosystems. The inclusion of these lags allowed time for the impacts of environmental variability to propagate through the food web and the effects to be more clearly identified in ecological indicators trends. However, while the application of time-lags to data series has been much discussed (e.g. Daan et al., 2005; Walters and Kitchell, 2001) it is known to vary based on the species encompassed within individual ecosystems, and therefore time lags will be expected to vary between ecosystems. Consequently, while the inclusion of time lags in future applications of the framework may be beneficial, this will

very much depend on the ecosystem being considered and its components. It is likely, therefore, that this aspect of the framework would benefit greatly from the inclusion of knowledge from ecosystem experts.

It is envisaged that the developments that have been made through the development and applications of this framework will add to the current EAF knowledge base. This framework could provide a method of providing stakeholders and managers with a structured synthesis large amounts of information, verified by ecosystem experts, which could be used to ease the decision-making process. It is anticipated that this framework would not be used to replace current management methods, but rather to add to the knowledge they provide to increase our understanding of the drivers within marine ecosystems. The inclusion of environmental variability and knowledge from ecosystem experts will play a key role in the utilisation of this framework to advance an EAF.

5.1.6. Lessons Learnt from these Case Studies

The framework developed here was easily adjusted to account for difference that arose between ecosystems, such as the influence of IUU catches in the Mediterranean and management implications in the North Sea. Several features developed in the framework allowed for this (See Chapter 1 – Table 1.1.). The inclusion of two additional categories, *possibly deteriorating* and *possibly improving*, to the categories developed in Bundy *et al.* (2010) allowed the categorisation of ecosystems to be more sensitive to indicator trends. Alongside this, all indicators trends were considered before the ecosystem was categorised, ensuring no indicator trends were ignored. The structure of the decision trees developed in the current study also means that the order of indicator indicators in the analysis did not impact the classification of the ecosystem. This could become useful in ecosystems where

data for all indicators are not available. However, as previously mentioned, the formal inclusion of environmental indicators in the framework was a novel feature that enhanced the decision tree approach and advanced the process of understanding ecosystem state in the context of indicator trends. The methodology described in the preceding chapters of this thesis produced meaningful results in all three case studies, and these results complemented previous indicator studies on these ecosystems. It is therefore proposed that the described method could be a significant step in the understanding of ecosystem-level state and trends, and that this is a baseline from which to evolve and adjust fisheries management strategies in an EAF.

Re-iterating, the ultimate goal of the present research was to develop an indicator framework to synthesise IndiSeas I and II indicators. This framework can be implemented in different types of systems, in line with the IndiSeas approach, and provides information on the state and trends of marine ecosystems. This will aid decision making in an ecosystem approach to fisheries management. The resultant framework was found to be robust to three implementations, from upwelling to temperate systems, remaining consistently applicable, useful and relevant across ecosystems. Although some adjustments were necessary within each case study to ensure ecologically meaningful results. This process was relatively simple and the overarching structure of the framework was found to be sound. In all three ecosystems, across all periods, some unexpected trends were observed in the suite of indicators underlying the evaluation. The altered nature of marine ecosystems through long-term exploitation, as well as alterations induced by climate change, means that ecological indicators do not always respond to changes in fishing pressure or environmental variability in ways expected from our understandings of past dynamics. The combination of fishing pressure and environmental variability have jointly shaped marine ecosystems, which could explain some of the unexpected trends observed.

5.2. Potential Improvements for Future Applications

While this framework successfully categorised the three ecosystems included in the case studies, there are several aspects which could be improved upon for future applications. For example, it currently stands, the framework is dependent on several assumptions that have been made when analysing indicator trends. The overall ecosystem score for each period relies heavily on the decisions and assumptions that have been made throughout the framework. These assumptions include the impacts of fishing pressure and the environmental on ecological indicators. For example, in the North Sea, it has been suggested that the influence of management on the ecosystem affected indicators more strongly than the environment in the second and third periods (Lynam, 2017). Therefore, despite significant trends detected in environmental indicators, it was recommended when adjusting scores environmental variability was considered to only partially explain trends observed in ecological indicators. However, this assumption could alter the classification of the ecosystem over these periods. It is possible that the stringent management enforced in recent years could entirely overshadow the impacts of the environment on ecological indicators within the ecosystem. If this were the case the ecosystem would be classified as 2.75 and 2.66 in Periods 2 and 3, respectively, and would considered neither improving nor deteriorating over any of the periods. This highlights the importance of thoroughly understanding ecosystem dynamics throughout the application of the framework, to ensure the correct interpretation of indicators trends. However, the inclusion of expert knowledge at each step when applying the framework should somewhat reduce the possibility of wrongly classifying ecosystems.

Another aspect of the framework that could benefit from re-evaluation is the use of linear regressions, the concern over which has been acknowledged throughout the thesis. It has been acknowledged that strength of the framework may be enhanced by the use other statistical

methods. Several studies have suggested that relationships are non-linear (e.g. Maunder and Punt, 2004; Barbier *et al.*, 2008), and therefore linear regressions may not give a true indication of trends. However, it is important to note that the division of data series into time periods, as is done in the above case-studies, effectively accounts for non-linearity.

Nevertheless, future applications of the framework may benefit form the inclusion of non-linear regression functions, where indicator trends are classified based on reference points in indicator values may enhance the detection of indicator trends. However, such an approach would rely strongly on the development of sound thresholds for each indicator. Reference points provide a foundation for comparing an indicator's value to an internal/external standard (Samhouri *et al.*, 2017). Thresholds in ecosystem states provide one of the most informative methods used to establish such reference points. Ecosystem thresholds can be identified as large response or an abrupt change in ecosystem state or function in response to an small change in an anthropogenic or environmental pressure (Groffman *et al.*, 2006; Samhouri *et al.*, 2017). These thresholds can help guide management actions and avoid undesirable shifts in ecosystem state (Foley *et al.*, 2015). It is unsurprising, therefore, that several recent publications have outlined potential methods to determine indicator threshold, moving away from traditionally developed baselines (e.g. Large *et al.*, 2013; Foley *et al.*, 2015; Samhouri *et al.*, 2017; Tam *et al.*, 2017). It is anticipated that the identification of thresholds of a common suite of indicators, such as those used in the IndiSeas project, will allow the assessment of ecosystem status across region, as well as assisting in the discovery of cross-ecosystem trends (Tam *et al.*, 2017).

There are several routes that could be adopted if an attempt was made to include thresholds into the framework developed in this thesis. Samhouri *et al.* (2017) have proposed a framework based on a multimodel inference to define thresholds for environmental and human pressures. This approach developed a common, step-wise, accessible and data-driven

approach by which to categorise thresholds in state-pressure relationships. However, the use of multiple lines of evidence in this approach led to some inconsistencies. Tam *et al.* (2017) have also proposed a method of developing operational reference points, quantifying thresholds for a suite of ecological indicators. This involved a complex statistical analysis, however offered clear ecosystem-level outputs of threshold ranges across pressure gradients. Therefore, the developed thresholds offer a powerful tool to delineate and communicate tipping points within ecosystems (Foley *et al.*, 2015; Tam *et al.*, 2017).

A further potential improvement to the current framework would be the inclusion of the management objectives assigned to each indicator, see Table 2.1, when structuring the framework. Incorporating these objectives into the structure of the framework may aid the interpretation of trends, and facilitate the communication of the observed trends to fisheries managers and stakeholders. It is likely that this aspect of the framework would need to be discussed when reviewing the framework with fisheries managers and other stakeholders.

One of the fundamental differences between previously proposed decision tree frameworks and that presented here is the separation of ecological and fishing indicators, and the combination of indicators fishing indicators into one indicator of "overall fishing pressure". However, the adopted method of combining these indicators may not allow a full understanding the influence of fishing pressure on the ecosystem. By combining these indicators into one indicator we may, therefore, be missing important trends influencing the ecosystem. Future implementations of the framework to other marine ecosystems could, therefore, potentially greatly benefit from an adjustment the overall fishing pressure indicator. One possible method of improving our interpretation of the influence of fishing pressure would be to recognise the different aspects of fishing pressure that each of these indicators represents. For example; the inverse fishing pressure indicators (1/(landings/biomass)) would be kept as the direct indicator of fishing pressure on the

ecosystem. MTI and TL landings would be included as indicators of fishing patterns within ecosystems. Landings would again be separate, as trends in this indicator, as seen in the North Sea ecosystem, can result from management measures employed, not solely the levels of fishing pressure on the ecosystem. By separating these indicators into three distinct groups, rather than combining them into one indicator, it may be possible to better interpret the influence of fishing pressure on ecosystems.

In future applications, the intrinsic vulnerability index should also be excluded from the "fishing" indicators. This indicator is intended to represent ecosystem functioning, rather than the influence of fishing pressure on ecosystems, therefore it is possible it may confound ecosystem classification.

Finally, an additional issue that arose when analysing overall fishing pressure was the fact that opposing trends in fishing indicators would effectively mask each other when scores are averaged to determine overall fishing pressure. As discussed above, the selected fishing indicators represent varying aspects of fishing pressure, and therefore cannot necessarily be expected to trend in the same direction. It is possible that the method of splitting the fishing indicators into separate groups, as described above, may somewhat mitigate for this issue. The use of correlations could also help in determining fishing pressure in an ecosystem. For example, if fishing indicators were highly correlated one indicator could be selected to represent the impact of fishing pressure over a period. However, this is something that should be further considered in future applications of the framework.

5.3. Ecosystem Complexities

To understand how indicators respond to the impacts of fishing pressure and climate variability it is important to consider the complex nature of marine ecosystems (Rochet and

Trenkel, 2003; Jennings, 2005; Link, 2005). Each ecosystem considered in this thesis contained different biological components and was influenced by different drivers. The impacts of bottom-up and top-down controls do not necessarily act separately, and therefore cascading trophic interactions can be mediated by opposing forces (Lynam *et al.*, 2017). This makes it difficult to predict the impacts of climate and fishing pressure on ecosystem components. However, it is anticipated that the inclusion of expert knowledge throughout the application of the framework will allow a better understanding on the complex impacts of fishing and environmental drivers on the ecosystems studied.

In all ecosystems it was extremely important to take care when interpreting indicator trends. An example of this can be found in the Southern Benguela and South Catalan Sea ecosystems which are dominated by small pelagic fish. In such cases, positive trends do not always signify a recovery within the ecosystem (see Shannon *et al*., 2010). When small pelagic fish become depleted, through either fishing pressure or due to environmental conditions, it is expected that indicators of mean length, mean lifespan, TL of landings and proportion of predators will increase. However, these increases do not necessarily reflect an improving situation or an increase in large species within the ecosystem. An example of such contrasting trends was noted in the North Sea, where an increase in biomass was accompanied by a decrease in mean length. Such conflicting trends can be caused by the indirect effects of fishing, including the release of prey species from predation pressure as large fish are targeted by fisheries, and the enhanced survival of recruits (Blanchard *et al.*, 2005; Daan *et al.*, 2005). Within each ecosystem different correlations between indicators were also observed, which may reflect the different histories of fishing and different management methods in each ecosystem (Blanchard *et al.*, 2010). However, adjustments of the developed weighting system allowed such differences in correlations to be accounted for, while maintaining the validity of the results.

5.4. The Global Context

The application of the framework developed in this thesis appears to have worked successfully for the ecosystems considered in the study, however, it is important consider the framework within the context of other approaches that have been employed in assessing marine ecosystems around the world. The move towards integrated approaches based on collaborative scientific research is necessary to better inform decision-making (Sutherland *et al.*, 2012). The framework described in this thesis incorporates several features of the methods that are currently being applied by fisheries scientists and managers around the world, described in Chapter 1, Section 1.7. This includes the use of indicators, the application of methodologies to multiple ecosystems and the aim of synthesising large amounts of information into easily communicable outputs.

The similarities of the framework developed here to work such as that envisaged by PICES-FUTURE could allow it to be used to synthesise indicators generated as output from forecasting runs of ecosystem models. Outputs from models such as EwE, Osmose or Atlantis could be used in this way, and allow the evaluation of possible ecosystem effects of management strategies in the medium and long-term.

There will always be a trade-off between methods of assessing ecosystem states and trends. Those methods which are highly specific to individual ecosystems will produce more detailed results, better capturing what is occurring within single components or specific communities and assemblages in the ecosystem in question. Yet the development of a more generalized approach allows for cross-ecosystem comparisons, an advantage of which includes allowing a better understanding of the broader ecosystem perspective, and at the same level of organisation. The framework described in this thesis falls somewhere in between these two approaches. While the generalised framework remains the same when applied to different

ecosystems, some fine tuning is necessary to make the approach ecosystem-specific. The similarities between the decision tree approach developed here and current international methods of ecosystem assessment highlight the potential success of its formal application in strategic management.

Similarly to the integrated ecosystem assessments (IEAs) utilised by the International Council for the Exploration of the Sea (ICES), an attempt at synthesising large amounts of scientific data has been made in this thesis. The decision tree framework developed in the present study demonstrates that it is possible to synthesise a comprehensive set of indicators at the level of ecosystem functioning into a meaningful assessment. However, the inclusion of indicators pertaining to the human dimensions in fisheries, as well as governance, are only beginning to be developed by IndiSeas (Bundy *et al.*, 2012; Bundy *et al.*, 2017). Inclusion of such indicators into the current decision tree framework is therefore beyond the scope of the present research.

5.5. Limitations and Recommendations

This thesis shows the successful application of the developed framework to three case studies, spanning temperate and upwelling ecosystems and shelf seas. Given the relatively short data series, it is possible that indicator trends may have been present but were not detected in the case studies of this thesis. In the current study the division of the data sets into smaller periods based on environmental shifts may therefore affect the ability of apparent trends to be detected as statistically significant. As the division of data sets based on environmental shifts does not allow for periods to be lengthened, it is possible that alternative statistical tests should be employed to better detect trends. The number of years included in analysed time series can greatly impact the ability of statistical tests to detect trends

(Nicholson and Jennings, 2004). However, in this framework the possibility for trends that were not statistically significant ($p<0.05$) to be interpreted as ecologically significant ($0.05 \leq p < 0.10$) was included. This allows potentially important trends to be included in the analysis and interpretation of ecosystem trends, ensuring no potentially important signals are missed.

Due to the nature of this study the use of reference values for ecosystem indicators was not considered appropriate. While reference levels may be suitable for individual ecosystems or those of similar location or type, they should constitute the sole benchmark for all ecosystems (Heymans *et al.*, 2014). Owing to the desire to apply the developed framework to various marine ecosystems in future, and because extensive model simulations and analyses are still underway to assist in the detection of reference levels for IndiSeas indicators (Shin, personal communication, 2017), the decision was made not to consider reference levels at this point.

It is also important to recognise that some indicators may take longer to respond to changes that others. For example, while biomass may increase relatively quickly, it may take the processes involved in changes of mean length longer to respond. This suggests that more work may be needed in order to determine how sensitive each indicator is which may be particularly important when short time series are utilised (Degnbol and Jarre, 2004; Blanchard *et al.*, 2010), and to consider different time lags in indicator responses.

There are also several processes that impact marine ecosystems that are not considered in this study such as pollution, coastal zone development and invasive species. Their inclusion in future frameworks is dependent on the development of appropriate data series for analysis, and/or the improvement of our understanding of the effects of these additional drivers.

5.6. Future Research

This thesis attempted to create a framework that would allow the gap in current methodologies to be bridged, lying between a 'one size fits all' (e.g. Bundy *et al.*, 2010) approach and approaches which focus on in-depth detail on each specific ecosystem and its components. The work conducted in this thesis, which works on smaller processes, could potentially add to the work conducted in IEAs. Levin *et al.* (2009) outlined five distinct steps involved in the IEA process; scoping, development of indicators, risk analysis, assessment of ecosystem status, management strategy evaluation and monitoring of ecosystem indicators and management effectiveness. The method of ecosystem assessment developed here could contribute to the assessment of ecosystem status, as well as the monitoring of ecosystem indicators and management effectiveness, potentially aiding the implementation of IEAs in marine ecosystems.

The successful implementation of the approach established within this thesis complements single-species studies by focussing on changes within the ecosystem as a whole. This should further the progress towards an ecosystem approach to fisheries in many regions.

The more ecosystems included in a comparative approach, the more significant the comparative analysis will be (Shin and Shannon, 2010). The next step would therefore be to apply the developed framework to additional ecosystems. The ecosystems considered here all have some similarities with one another, such as component composition and the importance of the continental shelf. It may therefore be sensible to apply the framework to other ecosystems that, while being relatively different to those systems analysed thus far, also has some similarities. This should increase our understanding of how differences feed through IndiSeas indicators and their combined evaluation in decision trees. In order to select ecosystems that would be easily comparable to those included in this thesis, in terms of data

available for indicators and the presence of models, ecosystems should preferably be selected from the well-studied set considered in Shannon *et al.* (2014a).

There are several ecosystems that have already been considered as potential case studies for further research. The Inner Ionian Sea in the eastern Mediterranean would provide an interesting contrast to the previously considered ecosystems. Although this area is relatively close to the South Catalan Sea geographically, as part of the Mediterranean Sea, the ecosystem dynamics and lack of localised upwelling should make for an interesting comparison as a new case study. This area of the Mediterranean is extremely oligotrophic, with levels of chlorophyll, nutrients and particulate organic carbon among the lowest in the entire Mediterranean (Pitta *et al.*, 1998). Several species of fish and mammals are now present only at low levels in the Ionian Sea, primarily thought to be a result of intense fishing effort which persisted until the end of the 1990s (Piroddi *et al.*, 2010), on top of the oligotrophic nature of the system.

The Western Scotian shelf, including the Bay of Fundy, would also make an interesting addition to future decision tree case studies. This ecosystem is located in the Northwest Atlantic and includes a wide continental shelf which is influenced by the northerly Gulf Stream and cold-water currents from Labrador and the Gulf of St. Lawrence. One of the issues faced in this ecosystem in the emergence of unsustainable new fisheries as traditional fisheries have decreased due to overfishing (Shelton *et al.*, 2006; Shackell and Frank, 2007).

Despite acknowledgement of the need to manage human activities in order to maintain the wellbeing of oceans and their ability to provide resources in future, the human dimension of an EAF has received relatively little attention. However, inclusion of the human dimension is a key element of an EAF (De Young *et al.*, 2008), and therefore the possibility of including this aspect into the current framework needs to be examined. Bundy *et al.* (2017) have

utilised several indicators which represent social, economic and governance characteristics. These included two socioeconomic indicators (the Human Development Index (HDI) and research and development (% of GDP)) and two general governance indicators (bad fisheries subsidies (% GDP) and political stability and absence of violence/terrorism). As these indicators were analysed in the context of the IndiSeas project it may be sensible to utilise these indicators when advancing the current framework to include the human dimension. Ideally these indicators would be included in a further step of the current framework, allowing the human dimension to be considered within the context of the influences of fishing pressure and environmental variability that have already been included.

Possibly the most important subsequent step in applying this framework would be the discussion of the use of the framework with fisheries managers and stakeholders. The framework provides a tool with which to summarise large amounts of scientific information, which is the first step in successful communication of the knowledge gained from ecosystem assessments described in this thesis. However, McGregor *et al.* (2015) highlighted the need to discuss such a tool with stakeholders for effective communication of the results. Tools, such as the decision tree developed here, are only useful when stakeholders are responsive to the information being presented (Johnson and Chess, 2006; Turnhout *et al.*, 2007). Therefore, although it appears to successfully categorise ecosystems, the true usefulness of the framework cannot be determined until it has been discussed with the appropriate audiences. It is also likely that including knowledge from stakeholders will allow further improvements to be made to the framework.

An iterative process of consultation with managers, listening to their questions on the use of the framework and potentially modify the tool based on their input, would be necessary to apply this framework within a fisheries management context. McGregor *et al.* (2015) acknowledged 'that scientists generally focus too much on the development of tools, and too

little on the opportunities for joint learning of multiple stakeholder groups'. The process of taking the framework to stakeholders would begin with meetings of focus groups to ensure that, as it currently stands, the framework addresses the requirements of each particular group. The use of focus groups has already proved useful in facilitating meaningful communications between stakeholders (Paterson *et al.*, 2010). The stakeholder groups involved in this process should contain experts on each system, EAF experts, academics, conservationist and management agencies. The current framework would be presented to the groups and, following this, discussions would be held on the best way to structure the framework, what aspects/indicators are the most valuable (i.e. potential new weightings) and how much detail must be included within the framework. This would also be a good point at which to discuss the structure of the decision tree in relation to the management objectives linked to each indicator.

5.7. Final Conclusions

Under the current levels of fishing pressure being exerted on marine ecosystems around the globe, the development of successful management methods to preserve ecosystem services and food security for future generations is becoming increasingly important. Alongside this, it has also been recognised that under the current levels of climate change it is vital to consider the impacts of environmental variability on marine ecosystems to ensure management measures can be adjusted for the resultant effects on ecosystem components.

The described decision tree framework allows the assimilation of large amounts of scientific data and knowledge into a format that should be easily conveyed to fisheries managers and stakeholders. The approach described in this thesis provides a unique method of analysing

state and trends of marine ecosystems of different types under the current changing environmental conditions.

The success of this approach in the ecosystems case studies included in this thesis serves to prove that a relatively simple framework, designed to incorporate large amounts of scientific knowledge, can effectively synthesise the trends of marine ecosystems under multiple drivers, and can be effectively applied to multiple marine ecosystems.

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Appendices

Appendix 1: The Southern Benguela Ecosystem

A1.1. Combining Indicators of the Impacts of Fishing

During preliminary analysis of the data, only "inverse fishing pressure" was used to identify the impacts of fishing on ecological indicators. However, several indicators in this study may be considered to represent the impacts of fishing on the ecosystem. These include "landings", "marine trophic index", "trophic level of landings" and "fishing pressure", with the addition of the "intrinsic vulnerability index" in Period 3 (2004-2010). The trends in these indicators can be used to explain trends in ecological indicators due to fishing pressure. Therefore the decision was made that these indicators should be combined, along with inverse fishing pressure, into a single indicator of the impacts of fishing. At this point, the decision was made to exclude the "marine trophic index" as it cuts trophic level at 3.25, which could exclude sardine from the study (Shannon *et al.*, 2014a). Two methods of combing these indicators were evaluated. The first method was to average the scores attributed to these fishing indicators equally, and the second to give "inverse fishing pressure" a 50% weighting (as this is the only direct indicator of fishing pressure) and the other fishing indicators a combined weighting of 50%. The weighted scores for each time period can be seen in Table A1. Although the weighting of fishing indicators resulted in an alteration of fishing pressure scores, there is little change in the actual observed trends (e.g. in Period 1 the new mean scores for both weightings remained as 2, which was still a significantly decreasing trend). The exception to this was Period 3 where, when indicators are given an equal weighting, the new score for fishing pressure was 3.5 which would be rounded up to 4. This resulted in a

change of indicator trend from no significant trend to a significantly decreasing trend, and was therefore considered to be too extreme a weighting. Therefore, the decision was made to utilise the 50/50 weighting.

Table A1.1: Combining all indicators of fishing pressure

Indicator	Original Fishing Pressure Score (only "inverse fishing pressure")	Mean with Equal Weighting	Mean with 50% Fishing pressure/50% other indicator weighing
Period 1 (1978-1993)	1	2.3	2
Period 2 (1994-2003)	1	2.3	2
Period 3 (2004-2010)	3	3.5	3.33

A1.2. Indicator Complementarity, Redundancy and Weighting

In order to assess the complementarity and redundancy of indicators correlations between ecological indicators were examined (following the methods used by Coll *et al.*, 2016). Correlations were evaluated using Spearman's non-parametric rank order correlation coefficient. This is a method of assessing the statistical dependence between two variables ranging between -1 and 1. A matrix was created for each time period to summarise positive and negative correlations in indicators along with the significance of these correlations, in order to assess overall redundancy (Table A2). This allowed us to ascertain whether it was necessary to retain the full suite of indicators, and whether and how correlated indicators should be weighted.

Table A1.2 a-c: Spearman's non-parametric rank order correlation coefficients (values below the diagonal) and associated p-values (values above the diagonal) of ecological indicators for **(a)** Period 1, **(b)** Period 2 and **(c)** Period 3. Significant correlations are highlighted in bold.

(a)	Period 1: 1978-1993					
Indicators	Mean Length	Mean Lifespan	Biomass	% Predators	TLsc	TLmc
Mean Length		0.058	0.175	0.103	0.103	0.685
Mean Lifespan	0.828		0.242	**0.017**	0.103	0.685
Biomass	-0.657	-0.600		0.336	0.297	0.919
Proportion of Predators	0.771	**0.942**	-0.486		0.175	1.000
Trophic Level of Surveyed Community	0.771	0.771	-0.543	0.657		0.136
Trophic Level of Modelled Community	0.257	0.257	-0.086	0.028	0.713	

(b)	Period 2: 1994-2003					
Indicators	Mean Length	Mean Lifespan	Biomass	% Predators	TLsc	TLmc
Mean Length		**0.017**	**0.033**	**0.033**	**0.017**	0.136
Mean Lifespan	**0.943**		**0.017**	**0.017**	**0.003**	0.242
Biomass	**-0.886**	**-0.943**		**0.003**	**0.017**	0.136
Proportion of Predators	**1.000**	**0.943**	-0.886		**0.017**	0.356
Trophic Level of Surveyed Community	**0.942**	**1.000**	-0.943	**0.943**		0.242
Trophic Level of Modelled Community	0.426	0.600	-0.714	0.486	0.6000	

(c)	Period 3: 2004-2010				
Indicators	Mean Length	Mean Lifespan	Biomass	% Predators	TLsc
Mean Length		0.139	0.556	0.139	0.048
Mean Lifespan	-0.643		0.066	**0.000**	0.066
Biomass	0.286	-0.750		0.066	0.556
% Predators	-0.643	**1.000**	-0.750		0.066
TLsc	-0.786	0.750	-0.286	0.750	

Indicator selection is a topic which has been under discussion since the initiation of the IndiSeas project, and all indicators have been selected for important reasons (Shin *et al*., 2005; Shin *et al*., 2010). The decision was therefore made that no ecological indicator selected by IndiSeas should be excluded from this study. Instead, ecological indicators which significantly correlated with others were weighted in order to downplay the impact of their score within ecosystem classification. It was found that "Mean Lifespan" and the "Proportion of Predators" were the only indicators that showed significantly correlated over all time periods. Following this line of thinking these indicators would have a lesser weighting and were assumed to contribute less to overall ecosystem scores than those indicators which did not show significant correlations with other indicators.

A sensitivity analysis was conducted in order to determine suitable weightings. Weighted means were calculated by multiplying final indicator scores by the appropriate weighting. Initially non-correlated indicators were given a weighting of 18%, while those correlated indicators were given a weighting of 14%. This gave correlated indicators a combined weighting of 28%, which is greater than the weighting of a single non-correlated indicators. In a second weighting attempt, 20% was attributed to non-correlated ecological indicators, while correlated indicators received a weighting of 10%. The combined weighting of the two

correlated indicators was therefore equal to the weighting of a single non-correlated indicator. Finally, a weighting of 22% was attributed to non-correlated indicators and a weighting of 6% to correlated indicators. This was an extreme weighting, with the correlated indicator's combined weighting being considerably less than that of a single non-correlated indicator. These three weightings were applied to the final ecological scores (after scores had been adjusted for the impact for the environment on indicators, see Figure 2.3. Chapter 2) for each time period (Table A3 a-c). Weightings were calculated by multiplying each final indicator score by the appropriate factors. For example, for weighting 1 where non-correlated indicators were attributed a weighting of 18%, the final indicator score was multiplied by 0.18 while correlated indicators were multiplied by 0.14 to give them a weighting of 14%. Once multiplied by these factors, a weighted mean was calculated by adding up these new weighted scores.

Weighting 1 was selected as the most appropriate to be applied in this study as although it lessens the contribution of correlated indicators to the final ecosystem score it still allows them a combined weighting greater than a single non-correlated indicator, therefore not completely masking their impact.

Table A1.3 a-c: Indicator weighting sensitivity analysis for **(a)** Period 1, **(b)** Period 2 and **(c)** Period 3. Note: As Period 3 has only five indicators (Trophic level of the modelled community is not available) it was necessary to divide the sum of the weighted scores by the total of the weightings. For example for weighting 1 two indicators are given a weighting of 0.14 and three are given a weighting of 0.18, therefore the total score must be divided by 0.82. Similarly the total score in weighting 2 must be divided by 0.8 and for weighting 3 by 0.78

(a)	**Period 1: 1978-1993**			
Indicator	Final Score	Weighting 1 (0.14/0.18)	Weighting 2 (0.1/0.2)	Weighting 3 (0.06/0.22)
Mean Length	2.25	0.405	0.45	0.495
Mean Lifespan	2.25	0.315	0.225	20.135
Biomass	2.25	0.405	0.45	0.495
Proportion of Predators	2.25	0.315	0.225	0.135
Trophic Level of Surveyed Community	2.25	0.405	0.45	0.495
Trophic Level of Modelled Community	5	0.9	1	1.1
Mean	**2.7**	**2.745**	**2.8**	**2.855**

(b)	**Period 2: 1994-2003**			
Indicator	Final Score	Weighting 1 (0.14/0.18)	Weighting 2 (0.1/0.2)	Weighting 3 (0.06/0.22)
Mean Length	1.5	0.27	0.3	0.33
Mean Lifespan	3.33	0.47	0.33	0.20
Biomass	0.33	0.06	0.07	0.072
Proportion of Predators	4	0.56	0.4	0.24
Trophic Level of Surveyed Community	3.33	0.6	0.66	0.73
Trophic Level of Modelled Community	3.33	0.6	0.66	0.73
Mean	2.64	2.56	2.42	2.3

(c)	**Period 3: 2004-2010**			
Indicator	Final Score	Weighting 1 (0.14/0.18)	Weighting 2 (0.1/0.2)	Weighting 3 (0.06/0.22)
Mean Length	4	0.72	0.8	0.88
Mean Lifespan	0.8	0.112	0.08	0.048
Biomass	2.4	0.432	0.48	0.528
Proportion of Predators	1.6	0.224	0.16	0.096
Trophic Level of Surveyed Community	0.8	0.144	0.16	0.176
Mean	**1.92**	**1.99**	**2.1**	**2.21**

A1.3 Comparison of Indicators Across Time Periods

To identify changes across the entire time series, Welch's two sample t-tests were utilised to identify the difference between mean indicators values in Period 1 and 2.
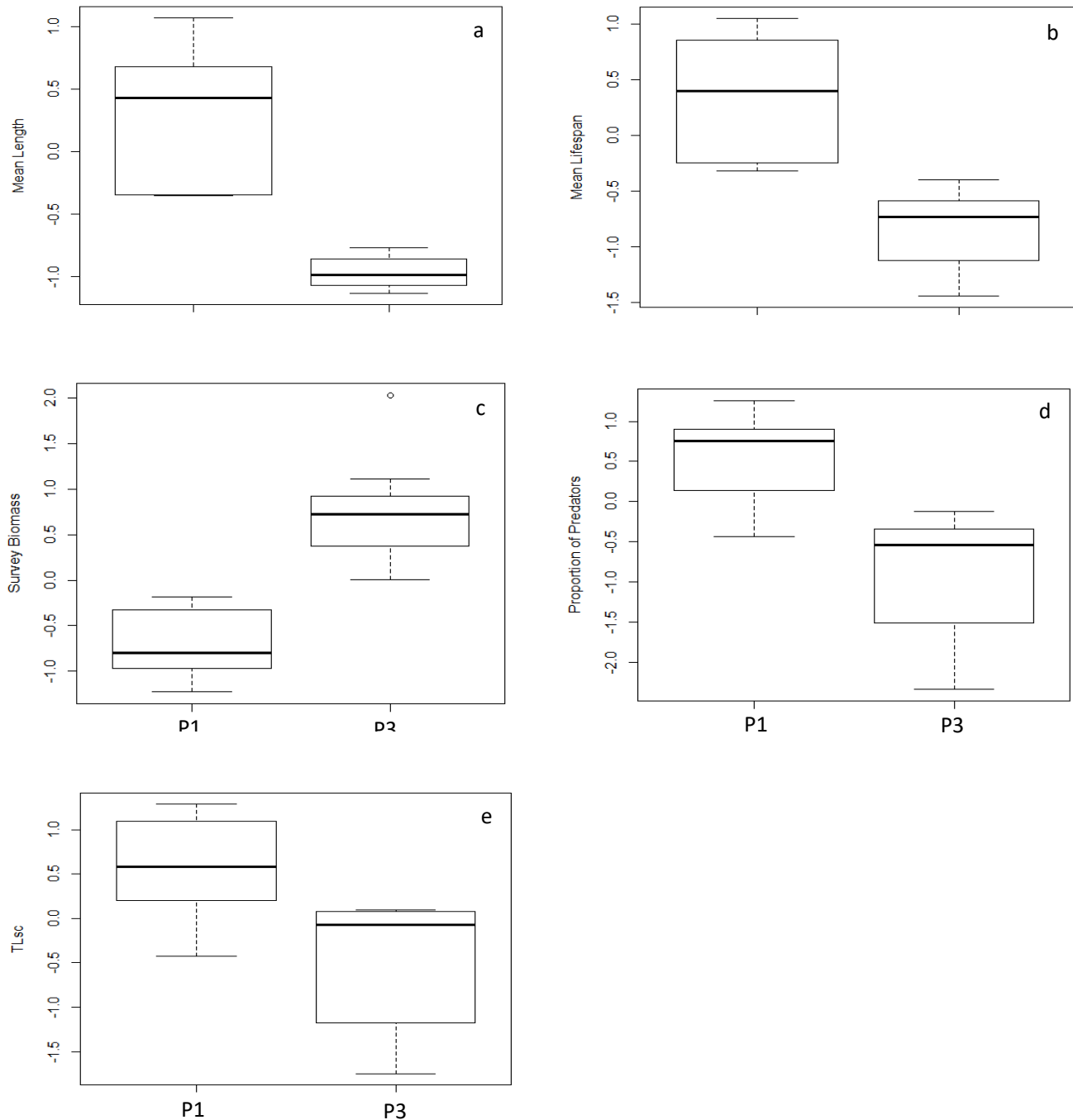
Figures A1.1 a-e: Comparison of means in of indicator means between Period 1 and Period 3 for (a) Mean length, (b) Mean lifespan, (c) Survey biomass, (d) Proportion of predators and (e) Trophic level of the surveyed community with corresponding p values.

Appendix 2: The South Catalan Sea Ecosystem

A2.1. Division of Data Series into Time Periods

Based on knowledge of environmental variability in the South Catalan Sea we acknowledged the need to divide the time series into distinct time periods. Analysis was conducted splitting the data in several different ways (Table A1a & b) with the aim of finding split which allowed the known environmental variability, as identified from the literature, to be picked up. Following analysis of all time splits it was found that dividing the data into Period 1 (1978-1990) and Period 2 (1991-2010) allowed the best detection of environmental variability occurring within the system

Table A2.1a & b: Analysis of trends in environmental indicators. Significant trends are highlighted in bold.

a.

Indicator	1978-1988	1978-1989	1978-1990	1978-1991	1978-1992
NAO	p = 0.412 slope = -0.055	p = 0.868 slope = -0.010	p =0.415 Slope = 0.05	p =0.368 slope = 0.049	p =0.136 slope = 0.07
WeMOi	**p = 0.001 slope = -.341**	**p = 0.000 slope = -0.345**	**p =0.000 slope = -0.281**	**p =0.002 slope = -0.230**	**p =0.0248 slope = -0.165**
River Runoff	p = 0.317 slope = -0.115	p = 0.098 slope = 0.177	**p =0.036 slope = -0.197**	**p =0.040 slope = -0.167**	p =0.058 slope = -0.136
Mean SST	p = 0.149 slope = 0.096	**p = 0.036 slope = 0.133**	**p =0.008 slope = 0.156**	p =0.124 slope = 0.092	p =0.220 slope= 0.065
Winter SST	p = 0.144 slope = 0.088	p =0.138 slope = 0.074	p =0.216 slope = 0.054	p =0.946 slope = -0.003	p =0.556 slope= -0.026

b.

Indicator	1989-2010	1990-2010	1991-2010	1992-2010	1993-2010
NAO	**p = 0.000 slope = -0.119**	**p = 0.000 slope = -0.122**	**p =0.004 slope = -0.106**	**p =0.008 slope = -0.109**	**p =0.040 slope = -0.086**
WEMOI	p =0.116 slope = 0.003	p = 0.349 slope = -0.025	p =0.257 slope = -0.033	p =0.203 slope = -0.041	p =0.397 slope = -0.030
River Runoff	p = 0.404 slope = 0.022	p = 0.836 slope = 0.005	p =0.694 slope = -0.011	p =0.716 slope = -0.011	p =0.874 slope = -0.005
Mean SST	p = 0.170 slope = 0.054	p = 0.0937 slope = 0.073	**p =0.032 slope = 0.100**	p =0.128 slope = 0.072	p =0.338 slope = 0.048
Winter SST	**p = 0.017 slope = 0.111**	**p =0.012 slope = 0.129**	**p =0.012 slope = 0.145**	**p =0.041 slope = 0.126**	p =0.105 slope = 0.111

A2.2. Assessing Complementarity and Redundancy of Indicators

To assess complementarity and redundancy of indicators, correlations between ecological indicators were examined (following the method applied in Coll *et al.*, 2016). Correlations were identified using Spearman's non-parametric rank order correlation coefficient, assessing the statistical dependence between two variables ranging between -1 and 1. A matrix was created for each time period to summarise any positive or negative indicator correlations along with their significance, in order to assess overall redundancy (Table A2.2 a & b). This allowed us to ascertain whether it was necessary to retain the full suite of indicators, and whether and how correlated indicators should be weighted. The "Proportion of Predators" and "Mean Lifespan" showed a significant correlation during the first period, alongside "Biomass" and the "Proportion of Predators". Although a significant correlation between "Mean Lifespan" and the "Proportion of Predators" was not observed in Period 2 it was quite close to being a strong correlation (Fowler *et al.*, 1998). The correlation between "Biomass" and the "Proportion of Predators" was, however, observed in Period 2. As the correlation between "Mean Lifespan" and the "Proportion of Predators" was also consistent with that found across all time periods in Lockerbie *et al.* (2016), the decision was made to apply weightings to these indicators as well as to the "Biomass" indicator over both time periods to account for possible redundancies.

Table A2.2a & b: Spearman's non-parametric rank order correlation coefficients (values below the diagonal) and associated p-values (values above the diagonal) of ecological indicators for **(a)** Period 1 and **(b)** Period 2. Significant correlations are highlighted in bold.

(a) Period 1: 1978-1990

Indicators	Mean Length	Mean Lifespan	Biomass	% Predators	TLsc	TLmc
Mean Length		**0.006**	0.068	0.113	0.075	0.068
Mean Lifespan	**0.707**		0.025	**0.000**	0.008	0.687
Biomass	-0.520	-0.614		**0.007**	0.015	0.713
Proportion of Predators	0.460	**0.878**	**-0.702**		0.015	0.583
Trophic Level of Surveyed Community	0.508	0.696	-0.651	0.651		0.202
Trophic Level of Modelled Community	0.519	0.123	0.112	-0.167	0.379	

(b) Period 2: 1991-2010

Indicators	Mean Length	Mean Lifespan	Biomass	% Predators	TLsc	TLmc
Mean Length		0.538	0.287	0.767	0.767	0.024
Mean Lifespan	-0.145		**0.000**	0.002	0.012	**0.000**
Biomass	-0.249	**-0.796**		**0.000**	0.003	0.048
Proportion of Predators	-0.070	0.648	**-0.732**		0.201	0.074
Trophic Level of Surveyed Community	-0.070	0.551	-0.637	0.297		0.086
Trophic Level of Modelled Community	0.506	**-0.771**	0.449	-0.409	-0.393	

The indicators utilised in the IndiSeas project have undergone a rigorous selection process

and have been selected for well-considered reasons (e.g. Shin *et al.*, 2010a; Shin *et al.*,

2010b). It is for these reasons that the decision was made to retain all indicators for this study. Weighting were modified from those used in Chapter 2 in order to account for the inclusion of the biomass indicator, attributing a weighting of 0.19 to non-correlated indicators and 0.155 to correlated indicators.

A2.3. Ecosystem Scores Not Adjusted for Increased Fishing Pressure

A preliminary analysis was conducted identifying overall ecosystem scores if fishing pressure was not adjusted to account for the impacts of IUU fishing practices (Table A2.3). Without adjusting the overall fishing pressure indicator, it was found that the ecosystem could be classified as possibly improving over both time periods.

Table A2.3: Scores of ecological indicators showing adjustments for the impacts of fishing pressure and the environment in the South Catalan Sea. Scores here are calculated making use of official fishing pressure data.

Indicator	Period 1						Period 2				
	Original Score	Fishing pressure trend - Does fishing pressure explain trend?	New score	Can trends in environmental indicators explain trend?	Final Score	Original Score	Trend - Does fishing pressure explain trend?	New score	Can trends in environmental indicators explain trend?	Final Score
Mean Length	5	Decreasing - No	5	Yes	3.33	4	Decreasing - No	4	Yes	2.66
Mean Lifespan	5	Decreasing - No	5	Yes	3.33	1	Decreasing - Partially	0.75	Yes	0.5
Biomass	3	Decreasing - Partially	2.25	Partially	1.8	5	Decreasing - No	5	Yes	3.33
Proportion of Predators	3	Decreasing - Partially	2.25	Partially	1.8	3	Decreasing - Partially	2.25	Partially	1.8
Trophic level of surveyed community	5	Decreasing - No	5	Yes	3.33	1	Decreasing - Yes	0.5	Yes	0.33
Trophic level of modelled community	5	Decreasing - No	5	Yes	3.33	5	Decreasing - No	5	Yes	3.33
Mean Score	4.3				2.82	3.2				1.99
Weighted Mean					2.5					2.06

A2.4. Individual Group Trends

In order to fully understand the trends observed in indicators, time series of surveyed (Table A4) and modelled biomasses (Table A5) of individual species and groups were analysed. In terms of surveyed biomass, anglerfish, juvenile and adult hake, sardine and mackerel were all observed to influence several ecological indicator trends.

Table A2.4: Trends in surveyed biomass of individual groups and species for Period 1 and 2 in the South Catalan Sea. Dark grey = significant negative trends, light grey = significant positive trends.

Group	Period 1(1978-1990)	Period 2 (1991-2010)
Mullets	p = 0.9657 slope = -0.00386	p = 0.2074 slope = -0.04979
Anglerfish	p = 0.002783 slope = -0.21626	p = 0.9167 slope = -0.004224
Flatfishes	p = 0.8844 slope = -0.01307	p = 5.224e-05 slope = 0.13164
Juvenile Hake	p = 0.01184 slope = 0.19320	p = 0.0001125 slope = 0.12792
Adult Hake	p = 0.0001075 slope = -0.24670	p = 1.46e-05 slope = 0.13703
Demersal Sharks	p = 5.48e-09 slope = -0.27324	p = 3.208e-07 slope = -0.14870
European Anchovy	p = 0.9896 slope = 0.001176	p = 0.1818 slope = 0.05259
Adult Sardine	p = 0.823 slope = 0.02009	p = 0.000844 slope = -0.1159
Horse Mackerel	p = 0.4408 slope = -0.06824	p = 0.01983 slope = 0.08724
Mackerel	p = 0.8497 slope = 0.01703	p = 0.0007047 slope = -0.11715
Atlantic Bonito	p = 0.3151 slope = 0.08797	p = 0.0001735 slope = -0.12563
Swordfish & Tuna	p = 0.02401 slope = 0.17843	p = 0.9877 slope = -0.0006234

In terms of modelled biomass, it is likely that benthic invertebrates, anglerfish, hake and other small pelagic are likely to have influenced trends in the "Trophic Level of the Modelled Community" indicator.

Table A2.5: Trends in modelled biomass of individual groups and species for Period 1 and 2 in the South Catalan Sea. Dark grey = significant negative trends, light grey = significant positive trends.

Group	P1 (1978-1990)	P2 (1991-2010)
Shrimps	p = 0.673 slope = -0.03332	p = 0.3206 slope = 0.03956
Crabs	p = 0.3811 slope = -0.06811	p = 0.5913 slope = 0.02160
Norway Lobster	p = 0.6868 slope = 0.03180	p = 0.000 slope = -0.15353
Benthic Invertebrates	p = 0.01174 slope = 0.17276	p = 0.00185 slope = 0.11016
Benthic Cephalopods	p = 0.8061 slope = 0.01941	p = 0.000 slope = 0.14372
Benthopelagic Cephalopods	p = 0.4188 slope = 0.06304	p = 0.004771 slope = -0.10215
Mullets	p = 0.9285 slope = 0.007101	p = 0.2074 slope = -0.04979
Conger Eel	p = 0.03709 slope = 0.14933	p = 0.0009306 slope = 0.11524
Anglerfish	p = 0.0005721 slope = -0.21096	p = 0.9167 slope = -0.00422
Flatfishes	p = 0.7783 slope = -0.02226	p = 0.000 slope = 0.13164
Poor Cod	p = 0.1156 slope = -0.11757	p = 0.01388 slope = -0.09136
Juvenile Hake	p = 0.07793 slope = 0.12984	p = 0.0001125 slope = 0.12792
Adult Hake	p = 0.000 slope = -0.23089	p = 0.000 slope = 0.13703
Blue Whiting	p = 0.801 slope = 0.01993	p = 0.01344 slope = -0.09172
Demersal Fish 1	p = 0.6326 slope = 0.03765	p = 0.5292 slope = -0.02527
Demersal Fish 2	p = 0.4188 slope = 0.06304	p = 0.004771 slope = -0.10215
Demersal Fish 3	p = 0.1485 slope = 0.10894	p = 0.0004271 slope = -0.12038
Demersal Sharks	p = 0.000	p = 0.000

	slope = -0.25377	slope = -0.14870
Benthopelagic Fishes	p = 0.000 slope = 0.24564	p = 0.1363 slope = 0.05831
European Anchovy	p = 0.6603 slope = 0.03465	p = 0.1818 slope = 0.05259
Adult Sardine	p = 0.6811 slope = 0.03241	p = 0.000844 slope = -0.1159
Other Small Pelagics	p = 0.000 slope = 0.24052	p = 0.001248 slope = 0.23512
Horse Mackerel	p = 0.6528 slope = -0.03546	p = 0.01983 slope = 0.08724
Mackerel	p = 0.7561 slope = 0.02454	p = 0.0007047 slope = -0.11715
Atlantic Bonito	p = 0.9762 slope = -0.002365	p = 0.0001735 slope = -0.12563
Swordfish and Tuna	p = 0.07491 slope = 0.13099	p = 0.07491 slope = 0.13099
Loggerhead Turtle	p = 0.861 slope = 0.01380	p = 0.000 slope = 0.000
Audouins Gull	p = 0.04396 slope = 0.14522	p = 0.02254 slope = -0.08569
Other Sea Birds	p = 0.01646 slope = 0.16656	p = 0.5118 slope = -0.02634
Dolphins	p = 0.1046 slope = -0.12083	p = 0.0000 slope = 0.000
Fin Whale	p = 0.09077 slope = -0.12528	p = 0.000 slope = -0.167788

A2.5. Environmental Trends

Trends in standardized environmental indicators were calculated for Period 1 and 2 (Figure A2.1 and A2.2).
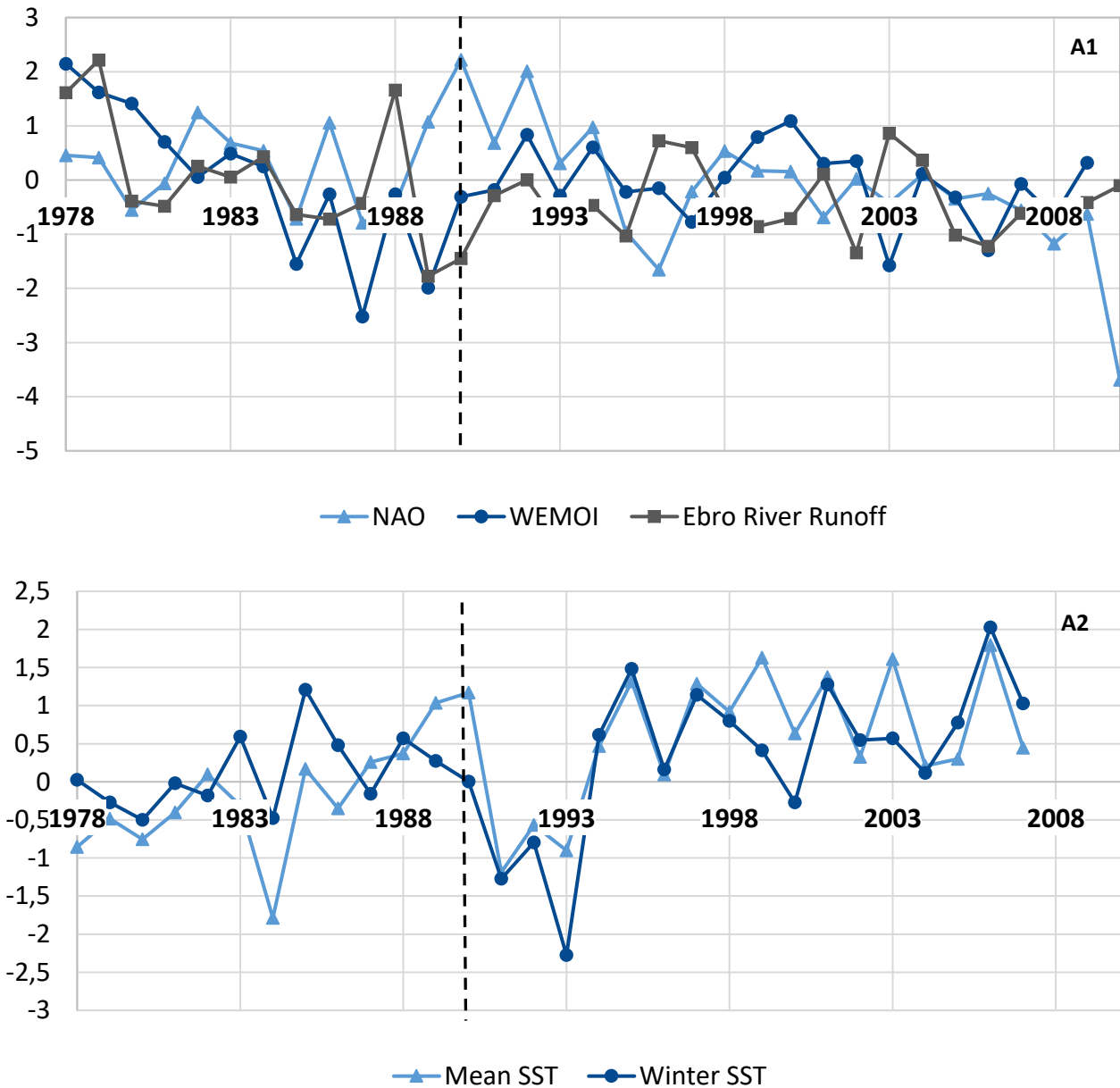
Figure A1 & A2: Trends in environmental indicators over entire time period. Dashed line identifies time period separation.

A2.6. Comparison of Indicators Across Time Periods

To identify changes across the entire time series, Welch's two sample t-tests were utilised to identify the difference between mean indicators values in Period 1 and 2.
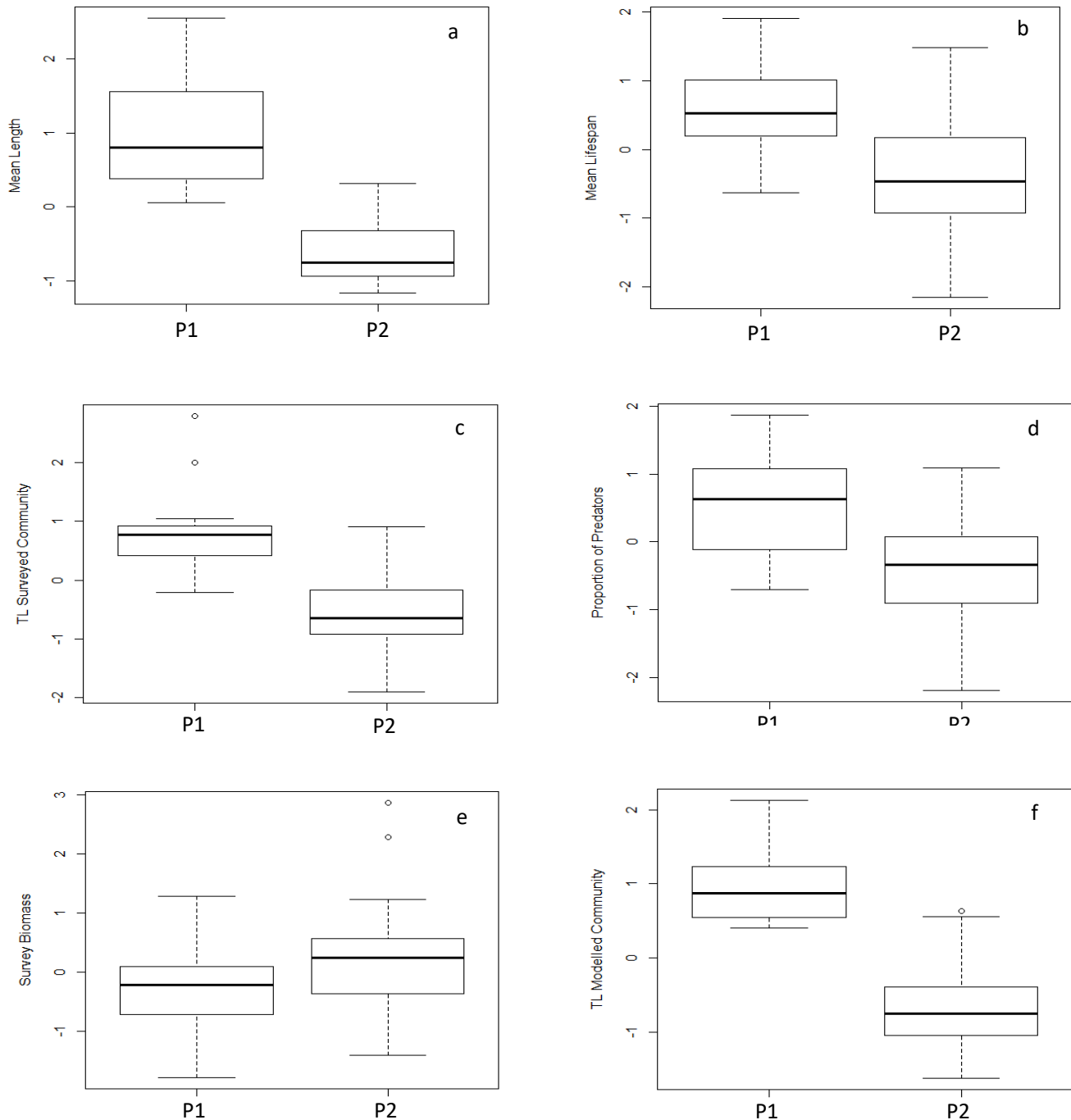
Figures A3 a-e: Comparison of means in of indicator means between Period 1 and Period 2 for (a) Mean length, (b) Mean lifespan, (c) Survey biomass, (d) Proportion of predators, (e) Trophic level of the surveyed community and (f) Trophic level of the modelled community with corresponding p values.

Appendix 3: The North Sea Ecosystem

A3.1. IndiSeas Indicators and the Corresponding European Marine Strategy Framework Directive (MSFD) Attributes

The indicators utilised in the IndiSeas project compliment those used in the MSFD

IndiSeas Indicator	MSFD Attribute
Mean Length	Population condition, proportion of selected species at the top of food webs, population age and size distribution
Mean Lifespan	Population condition, abundance/distribution of key trophic groups/ species
Survey Biomass	Population size, habitat condition
Proportion of predatory fish	Proportion of selected species at the top of food webs, abundance/ distribution of key trophic groups/species, ecosystem structure
Trophic level of the community	Habitat condition, proportion of selected species at the top of food webs, abundance/distribution of key trophic groups/species
Marine trophic index (TL>=3.25)	Proportion of selected species at the top of food webs, level of pressure of the fishing activity
Trophic level of landings	Ecosystem structure, Level of pressure of the fishing activity
Intrinsic vulnerability index	Habitat condition, level of pressure of the fishing activity
Inverse fishing pressure	Level of pressure of the fishing activity

approach. Each selected IndiSeas indicator can be linked to the attributes desired by the MSFD approach to assess good environmental status (GES), Table S1.

Table A3.1: IndiSeas indicators and the corresponding attributes of the MSFD for assessing good environmental status. Adjusted from MEECE (2011).

A3.2. Correlations

The method developed in Coll *et al.* (2016) was adopted to identify complementarity and redundancy of ecological indicators. Spearman's non-parametric rank order correlations were conducted to assess the statistical dependences of two indicators. A matrix was created for each period to summarise positive and negative correlations (grey), as well as their significance (white) (Tables A3a, b & c). It can be assumed that indicators which correlate with one another may be somewhat redundant. As the majority of indicators were observed to correlate with one another, particularly in Period 3 the decision was made that the weightings applied to mean length and mean lifespan in Chapter 2 (Lockerbie *et al.*, 2016) and Chapter 3 (Lockerbie *et al.*, 2017) would not be necessary for this study.

Tables A3.2a-c: Spearman's non-parametric rank order correlation coefficients (values below the diagonal, grey) and associated p-values (values above the diagonal, white) of ecological indicators for **(a)** Period 1, **(b)** Period 2 and **(c)** Period 3. Significant correlations are highlighted in bold.

(a)	Period 1 (1983-1994)					
Indicator	Mean Length	Mean Lifespan	Survey Biomass	Proportion of Predators	TLsc	TLmc
Mean Length		0.099	0.457	0.499	0.886	
Mean Lifespan	-0.497		0.063	0.031	0.956	
Survey Biomass	0.237	-0.549		0.921	0.075	
Proportion of Predators	0.216	-0.619	0.034		0.022	
TLsc	-0.048	0.017	0.538	-0.664		
TLmc						

(b)	Period 2 (1995-2003)					
Indicator	Mean Length	Mean Lifespan	Survey Biomass	Proportion of Predators	TLsc	TLmc
Mean Length	■	0.493	0.708	0.911	0.981	0.385
Mean Lifespan	0.266	■	0.493	**0.010**	0.4933	0.911
Survey Biomass	0.150	-0.266	■	0.212	0.162	**0.002**
Proportion of Predators	-0.050	**-0.816**	0.466	■	0.085	0.677
TLsc	-0.016	0.266	-0.516	-0.616	■	0.249
TLmc	0.333	0.05	**0.900**	0.166	-0.433	■

(c)	Period 3 (2004-2010)					
Indicator	Mean Length	Mean Lifespan	Survey Biomass	Proportion of Predators	TLsc	TLmc
Mean Length	■	**0.023**	**0.034**	**0.024**	0.235	╱
Mean Lifespan	**-0.857**	■	0.235	**0.000**	**0.034**	╱
Survey Biomass	**0.821**	-0.535	■	0.235	0.906	╱
Proportion of Predators	**0.857**	**-1.00**	0.536	■	**0.034**	╱
TLsc	-0.535	**0.821**	-0.071	**-0.821**	■	╱
TLmc	╱	╱	╱	╱	╱	■

In this study, strong correlations with a coefficient value larger/smaller than +/- 0.7 are included with a p value less than 0.05 (based on Fowler *et al.*, 1998).

A3.3. The Application of Time Lags to Data Series

Time lags are necessary to allow ecosystem components to react to changes in both the environment and in management. Therefore, a decision was made to analyse ecological and fishing indicators to determine whether it was necessary to apply a time lag to the data set in order to detect the expected changes. Ecological fishing indicators were therefore analysed with (Figure S4a) and without (Figure S4b) the application of time lags. It was observed that the smallest time lag which allowed expected trends, particularly in fishing pressure indicators, was two-year (Figure S4b) and therefore it was decided that a two-year lag be applied to all ecological and fishing indicators. This is consistent with biological understanding.

Tables S4a & b: Analysis of ecological and fishing indicators without (a) and with a two-year time-lag (b).

a.

Indicator	Period 1 (1983-1992)	Period 2 (1993-2001)	Period 3 (2002-2010)	Period 1 vs Period 3
Mean Length	p = 0.641 Slope = 0.054 Score: 3	p = 0.363 Slope = -0.116 Score: 3	p = 0.003 Slope = -0.343 Score: 5	→ p = 0.488
Mean Lifespan	p = 0.648 Slope = -0.042 Score: 3	p = 0.987 Slope = 0.001 Score: 3	p = 0.166 Slope = 0.220	↓p = 0.051
Survey Biomass	p = 0.154 Slope = 0.155 Score: 3	p = 0.699 Slope = 0.051 Score: 3	p = 0.044 Slope = -0.276 Score: 5	→ p = 0.566
Proportion of Predators	p = 0.156 Slope = -0.159 Score: 3	p = 0.339 Slope = 0.110 Score: 3	p = 0.122 Slope = -0.194 Score: 3	↑ p = 0.042
TLsc	p = 0.000 Slope = 0.282 Score: 1	p = 0.020 Slope = -0.202 Score: 5	p = 0.182 Slope = 0.118 Score: 3	↓ p = 0.001
TLmc		p = 0.168 Slope = 0.152 Score: 3	p = 0.009 Slope = -0.240 Score: 5	
Landings	p = 0.219 Slope = 0.058 Score: 3	p = 0.070 Slope = -0.134 Score: 4	p = 0.003 Slope = -0.261 Score: 5	→ p = 0.709
MTI		p = 0.081 Slope = -0.239 Score: 4	p = 0.278 Slope = 0.071 Score: 3	
TL Landings	p = 0.000 Slope = -0.274 Score: 5	p = 0.093 Slope = -0.169 Score: 4	p = 0.663 Slope = 0.052 Score: 3	→ p = 0.908
IVI		p = 0.775 Slope = 0.049 Score: 3	p = 0.002 Slope = 0.376 Score: 1	
Overall Fishing Pressure	3.5	3	3	

Indicator	Period 1 (1983-1994)	Period 2 (1995-2003)	Period 3 (2004-2010)
Mean Length	p = 0.586 Slope = 0.043 Score: 3	p = 0.203 Slope = 0.181 Score: 3	p = 0.014 Slope = -0.426 Score: 5
Mean Lifespan	p = 0.229 Slope = -0.080 Score: 3	p = 0.950 Slope = 0.006 Score: 3	p = 0.025 Slope = 0.520 Score: 1
Survey Biomass	p = 0.109 Slope = 0.131 Score: 3	p = 0.036 Slope = 0.246 Score: 1	p = 0.669 Slope = -0.065 Score: 3
Proportion of Predators	p = 0.462 Slope = -0.059 Score: 3	p = 0.405 Slope = 0.097 Score: 3	p = 0.076 Slope = -0.357 Score: 4
TLsc	p = 0.010 Slope = 0.172 Score: 3	p = 0.068 Slope = -0.151 Score: 4	p = 0.036 Slope = 0.274 Score: 1
TLmc		p = 0.006 Slope = 0.357 Score: 3	p = 0.181 Slope = -0.224 Score: 3
Landings	p = 0.285 Slope = 0.033 Score: 3	p = 0.042 Slope = -0.167 Score: 5	p = 0.036 Slope = -0.224 Score: 5
MTI		p = 0.003 Slope = -0.294 Score: 5	p = 0.078 Slope = 0.176 Score: 2
TL Landings	p = 0.003 Slope = -0.187 Score: 3	p = 0.600 Slope = 0.065 Score: 3	p = 0.219 Slope = -0.144 Score: 3
IVI		p = 0.358 Slope = -0.108 Score: 3	p = 0.007 Slope = 0.498 Score: 1
Overall Fishing Pressure Score	2	2.5	3

A3.4. Comparison of Indicators Across Time Periods

To identify changes across the entire time series, Welch's two sample t-tests were utilised
to identify the difference between mean indicators values in Period 1 and 3.
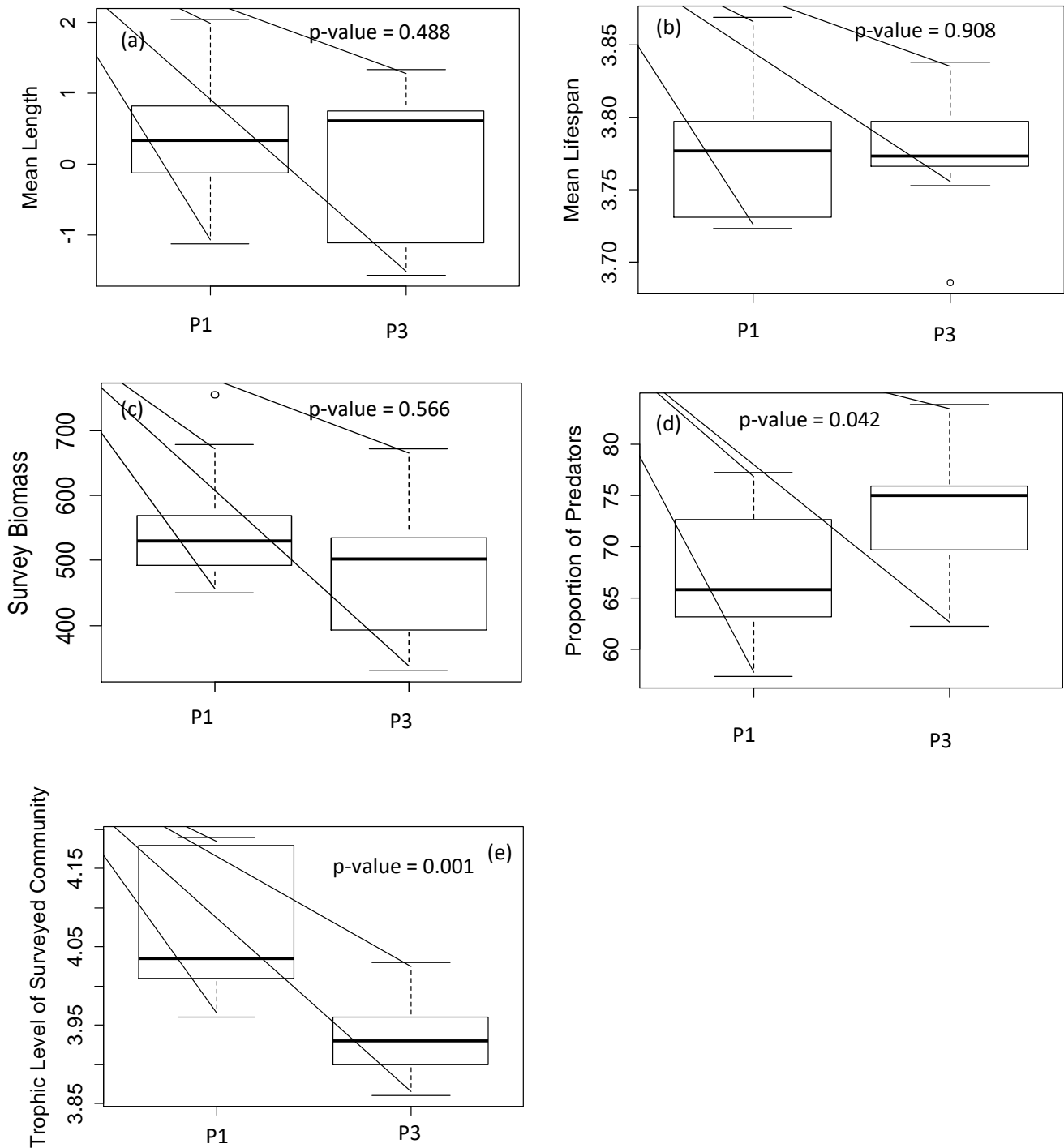
Figures A3.4 a-e: Comparison of means in of indicator means between Period 1 and Period 3 for (a) Mean length,
(b) Mean lifespan, (c) Survey biomass, (d) Proportion of predators and (e) Trophic level of the surveyed
community with corresponding p values.

A3.5. Analysis of the Entire Data Set as a Single Period

An analysis was also conducted of the entire data set to determine long term trends within the ecosystem. Following weighting of fishing pressure indicators an ecologically significant decline in fishing pressure was observed over the entire period. A decision tree was developed to analyse the influence of fishing pressure on ecological indicators (Figure S6a). There was a lack of trends observed in several ecological indicators, however some unexpected trends were observed, in particular a significant decrease in mean lifespan and TLsc occurring over the entire period. These trends suggest a decline in fish size in the ecosystem over time. This relates to the observed increase in smaller fish over the final two time periods observed when the data set is divided, a trend which is masked in other indicators if only the entire time series is considered.

Over the entire data set environmental indicator trends were more significant, as would be expected, with only the NAO lacking a significant trend (Table S6a). Following the adjustment of indicator scores to account for the impacts of fishing pressure and the environment on ecological indicators the ecosystem was classified as possibly improving (Table S6b).

A3.6. Breakpoint Analysis