Pham Viet Anh

Application of ecosystem modelling for decision support in marine fisheries management in Vietnam

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Academic promoter 1: Prof. Dr. ir. Peter Goethals

Laboratory of Environmental Toxicology and Aquatic Ecology

Department of Applied Ecology and Environmental Biology

Ghent University, Belgium

Academic promoter 2: Prof. Dr. ir. Frederik De Laender

Research Unit in Environmental and Evolutionary Biology

Biology Department

University of Namur, Belgium

Local promoter: Dr. Chu Tien Vinh

Former Deputy Director General of Directorate of Fisheries

Ministry of Agriculture and Rural Development, Viet Nam

Dean

Prof. dr. ir. G. Van Huylenbroeck

Rector

Prof. dr. Anne De Paepe

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AC: Ratio of Ascendancy to Development Capacity

B: Biomass

BA: Biomass Accumulation

CE: Constraint Efficiency Index

CPUE: Catch Per Unit Effort

D-FISH: Directorate of Fisheries

DECAFIREP: Department of Capture Fisheries and Resources Protection

EE: Ecotrophic Efficiency

EEZ: Exclusive Economic Zone

EwE: Ecopath with Ecosim

FAO: Food and Agriculture Organization of the United Nations

FCI: Finn's Cycling Index

FiB: Fishing in Balance

FWE: Food Web Efficiency

HP: Horsepower

LIM: Linear Inverse Model

MARD: Ministry of Agriculture and Rural Development

MSVPA: Multi-Species Virtual Population Analysis

MSY: Maximum Sustainable Yield

MTI: Marine Trophic Index

MTL: Marine Trophic Level

P/D: Pelagic/Demersal Ratio

RIMF: Research Institute for Marine Fisheries

SEAPODYM: The Spatial Environmental Population Dynamic Model

Sub-DECAFIREP: Sub-Department of Capture Fisheries and Resources Protection

WCPFC: Western and Central Pacific Fisheries Commission

Y: Yield from Fisheries

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Chapter 1: General introduction and structure of the thesis

1. Chapter 1: General introduction

1.1. Introduction of Vietnamese fisheries

Vietnam has a coastline of more than 3260 km and an economic exclusive zone (EEZ) of more than 1 million km^2. Due to geographical characteristics of the Vietnamese marine ecosystem and to ensure effective fisheries management, fishing zones are divided into different zones as indicated in the Figure 1.1 (Vietnamese Government 2010). These include the inshore are (grid area), coastal area (horizontal stripes) and offshore area (i.e. outside the coastal zone but within Vietnamese EEZ) (Figure 1.1). Accordingly, the inshore area only allows for small fishing vessels with capacity less than 20 HP or vessels without the engine to be fished. The vessels with capacity from 20-90 HP can only be operated in the coastal and offshore areas and the vessels higher than 90 HP can only be operated in the offshore and high sea areas.

Since early 1990s, marine capture fisheries have developed significantly and rapidly in Vietnam. In 1990, there were only 41,266 fishing vessels with a total capacity of 727,500 horsepower (HP), primarily operating in coastal areas with total catches of about 672,130 tons (DECAFIREP 2013). In 2013, the total number of fishing vessels reached more than 120,000 units (DECAFIREP 2013), the total engine capacity has increased to over 7 million HP and total catches reached 2.23 million tons. Value of export product from fisheries sector reached over $6.7 US billion in 2013 contributing for about more than 3% of total gross domestic product (GDP) and 24% in GDP of agricultural sector (MARD 2014). Actually, fisheries sector creates direct employment opportunities for over one million people (DECAFIREP 2013).

Despite the socio-economic importance of fisheries in Vietnam, it is questionable what the impact and sustainability is of the fisheries and how sustainable development can be achieved due to lack of effective fisheries management. According to recent assessments, catches have by far exceeded the maximum sustainable yield in the Vietnamese coastal waters and many marine fish stocks have been seriously reduced (Pomeroy et al. 2009). Average catch per horsepower (HP) estimated in the 1980s was around 1.1 tons·HP^{-1} and this number was reduced to about 0.35 tons·HP^{-1} in recent years (Figure 1.2) (Anh et al. 2014a).

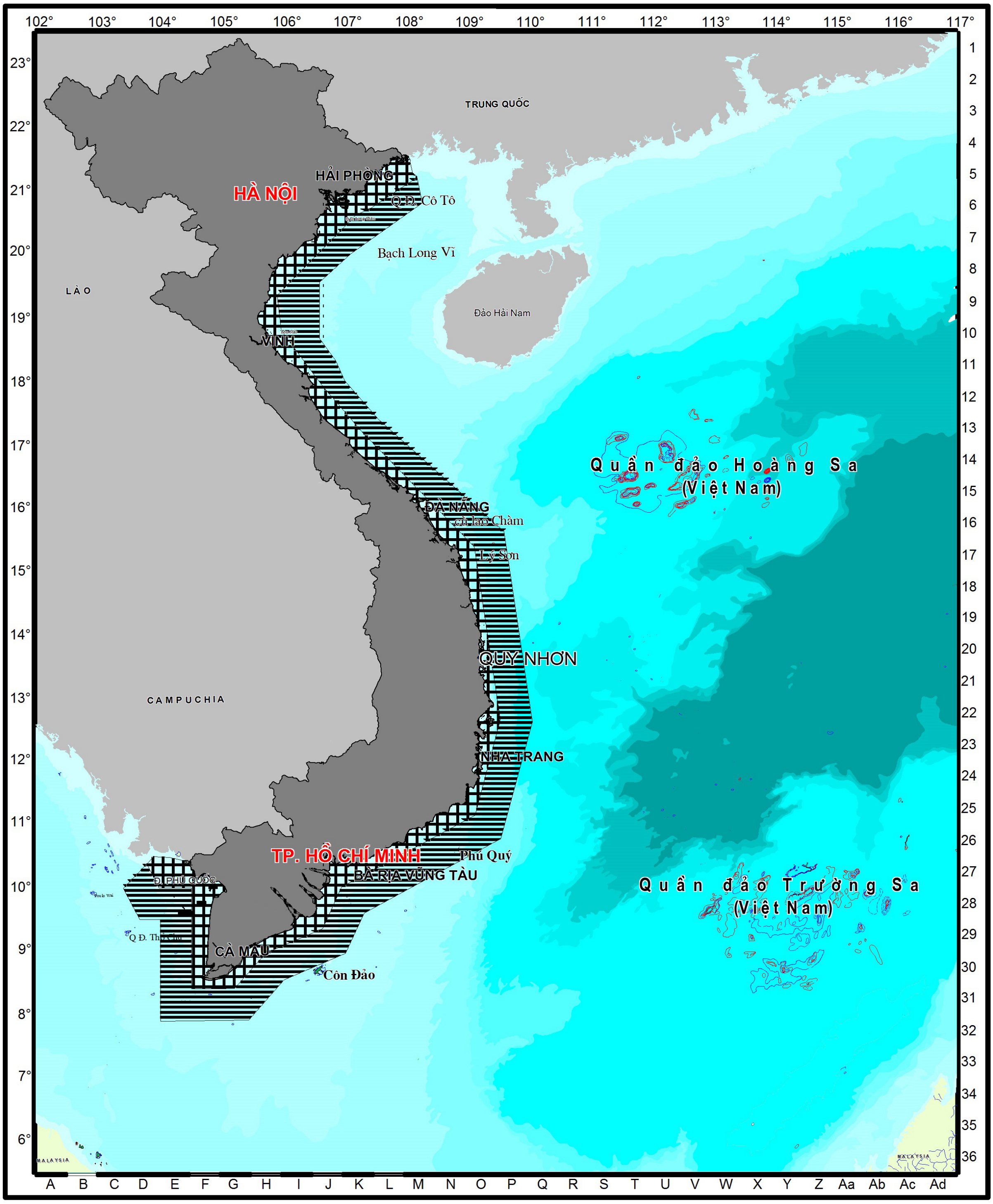

Figure 1.1. Division of the Vietnamese fisheries management regions. The gridded and horizontal striped areas denote the inshore and coastal zones, respectively. Note: a common fishing ground, the coastal zone in the Tonkin Gulf sharing between Vietnam and China, is not included.

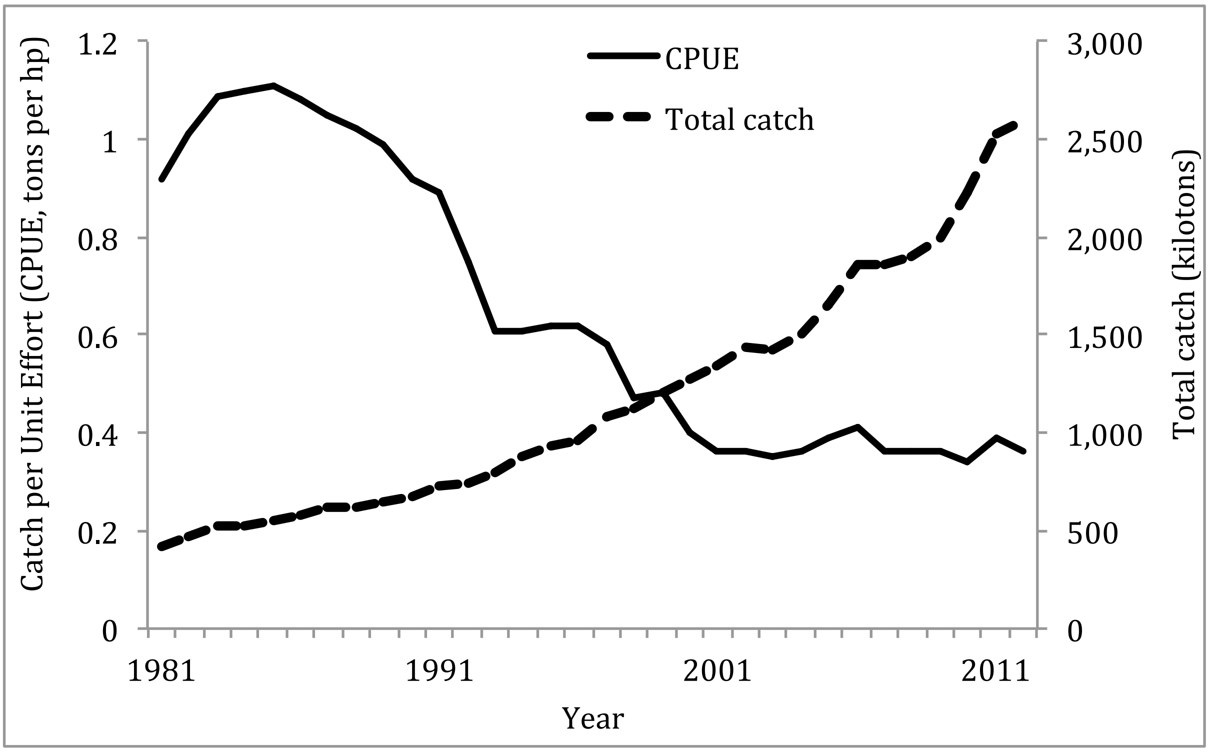

Figure 1.2. Catch of capture fisheries of Vietnam and productivity index (catch per unit effort, CPUE). Adapted from an official report of Department of Capture Fisheries and Resources Protection, Vietnam (DECAFIREP 2013).

Fisheries management agencies in Vietnam are complexly structured. The Ministry of Agriculture and Rural Development (MARD) is the highest level management agency at national level responsible for the agriculture, forestry and fisheries sectors. Under the MARD, Directorate of Fisheries (D-FISH) is responsible for all fisheries management-related issues. Under the D-FISH, there are several management departments and branches at provincial level to conduct monitoring, controlling and surveillance activities and fisheries data collection.

Marine resources are declining corresponding to the increasing of number of fishing boats. Consequently, the higher the effort is, also means the higher the cost per day is and the lower the income per unit of effort. To keep income stable, fishermen have to intensify fishing capacity (e.g. by increasing fishing duration, increasing number of gear operations, and reducing mesh size) and this leads to a worse situation whereby resources are getting exhausted. Therefore, there is a need to have proper management approaches to guide a sustainable development of the marine fisheries.

In addition to fishing pressure, environmental pollution is also one of the key problems for the Vietnamese fisheries sector (Tran 2011). Many ecosystems components such as mangroves, estuaries, coastlines and coral reefs have been destroyed by fishing activities and this seriously affected breeding and nursery habitats. Oil sludge and waste of fishing boats being regularly discharged into the sea is a common action in any fishery village in Vietnam. As a result of all these factors, fish and other aquatic life are adversely affected in the marine ecosystems.

Currently, some fisheries management measures have been applied, such as fishing closed seasons and protected areas, as well as mesh size restrictions in Vietnam (MOFI 2006, MARD 2011). However, these management policies have been established using only information from single-species assessments. Consequently, interactions among species are not assessed and this remains a limitation for the accurate improvement of fisheries management in Vietnam. There have not been attempts to integrated ecosystem approaches to fisheries management system due to lack of scientific evidence and outcomes of holistic evaluations tools.

1.2. Ecosystem-based fisheries management

Single-species fisheries management is an approach to manage a stock-by-stock basis and try to maximize fisheries production, economic benefits, employment or revenues. However, it is widely accepted that the single-species management has often led to unsustainable exploitation because social, economic and ecological objectives could not be met simultaneously (FAO 2008). In addition, the single-species fisheries management approach could not be used to take into account the interactions in the ecosystem (FAO 2008). It is gradually being realized that the historical impacts of fishing have been large, dramatic and difficult to reverse (Harvey et al. 2003). Fishing not only has direct effects, but also affects other components of the ecosystem (Griffiths et al. 2010). For example, there is often bycatch of non-targeted species, sometimes due to food-chain effects, and physical damage to habitats. Therefore, fishing can cause changes in the structure and functioning of ecosystems (Griffiths et al. 2010). To achieve sustainability and address limitations of the single-species fisheries management approach, an ecosystem approach to fisheries management (EAFM) that is integrating multiple drivers in a common framework is therefore needed (FAO 2003, FAO 2008). Implementation of EAFM requires the application of scientific methods and tools that also go beyond the single-species level, to a large extent, the exclusive sources of scientific advice (FAO 2008). Managers and decision-makers must find solutions that consider the wider range of societal objectives as recommended under an EAFM. It is also recognized that without quantitative evaluations of ecosystem changes under alternative fishing policies, marine ecosystems could be devastated through changes in ecosystem structure and functioning.

To implement EAFM, FAO (2003) established guidelines to guide fisheries scientists and managers worldwide. The guidance is to translate the economic, social and ecological policy goals and demands on sustainable development into operational objectives, indicators and performance measures. The guideline has also provided principles and concepts to fisheries management under EAFM and stated as follows:

- Fisheries should be managed to limit their impact on the ecosystem to the extent possible;
- Ecological relationships between harvested, dependent and associated species should be maintained;
- Management measures should be compatible across the entire distribution of the resource (across jurisdictions and management plans);
- Because knowledge on ecosystems is incomplete and thus the precautionary approach should be applied; and
- Governance should ensure both human and ecosystem well-being and equity.

EAFM has widely been accepted as legislation framework for fisheries management. At the national level, the United States of America has established an 'ecosystem principles' advisory panel and this panel has recommended to apply ecosystem principles, goals and policies to fisheries management and to develop Fisheries Ecosystem Plans (Fluharty 1999). In response to these recommendations, several ecosystem management plans have been implemented including Chesapeake Bay fishery ecosystem plan, South Atlantic Fishery Ecosystem Plan, Western Pacific Fishery Ecosystem Plan, etc. (Fluharty 1999).

Australia is one of the leading countries making good progress in implementing many elements of the EAFM. Australia has implemented some specific integrated elements such as: undertaking ecological risk assessment and developing a risk management response, implementing large-scale spatial management; enhancing fishery data collection; and enhancing liaison and community capacity (OECD 2010).

In Canada, the Ocean Act has been developed since 1997 (Canadian Government 1997) and provided legislation basic for ecosystem approach and precautionary approach principles (OECD 2010). In 2002, an Ocean Strategy was established and then an Ocean Action Plan was also developed to describe detail of an ecosystem approach to the management of human activities. Especially in 2007 Canada published a science framework for applying ecosystem approach to integrated management for fisheries, ocean, aquaculture and species at risk management.

At regional level, EAFM is implemented very effective at the Benguela Current Large Marine Ecosystem with cooperation of FAO and three countries in the region (i.e. Angola, Namibia and South Africa). In addition, regional fisheries management organizations (RFOMs) are responsible for playing a key role in managing fisheries resources beyond national jurisdiction. RFMOs have also adopted specific management measures such as bycatch reduction and habitat protection and marine protected areas (UN, 2006). Especially, the European Commission (EC) is also working towards implementation of EAFM through various instruments in the region. EC has established several Regional Advisory Councils (European Commission 2004a) such as in the North Sea (European Commission 2004b), in the north-western European waters (European Commission 2005), in Baltic Sea (European Commission 2006) to integrate ecosystem-based management and precautionary principles into management advices. The Marine Strategy Directive of the Commission recognizes the EAFM as one of the most important issues in the European contexts (European Commission 2008).

At international level, the concepts and principles of EAFM have been integrated in many international instruments, agreements and conferences such as the 1992 Convention on Biological Diversity, the 1995 United Nations Fish Stocks Agreement, the 1995 FAO Code of Conduct for Responsible Fisheries, etc. (FAO 2003). In June 2006, FAO held an Expert Consultation on economic, social, institutional considerations of applying EAFM and during the consultation participants recommended that the FAO publish technical guidelines on economic, social and institutional aspects of EAFM. This recommendation was taken into consideration in a document of FAO in 2008 (FAO 2008).

It is clear that EAFM has received considerable attention by fishery managers and scientists due to its potential to support comprehensive management decisions (Murawski 2007, Griffiths et al. 2010). However, questions on what are the potential gains of implementing EAFM, and how EAFM can be implemented in data-poor areas need still to be considered and addressed by fisheries scientists and managers.

1.3. Fisheries impacts on marine ecosystems

There are several human impacts to the marine ecosystem such as contamination, habitat degradation, and eutrophication. Also, fishing has also been considered to impact marine ecosystems and generate probably irreversible structural and functional changes (Estes et al. 2011). As indicated in Figure 1.3, increasing fishing pressure and habitat degradation have caused wide and strong impacts on ecosystems worldwide, which are reflected as changes in predator-prey interactions, food web (e.g. spatial distribution, productivity, and structure of exploited communities (Myers and Worm 2005, Lotze et al. 2006) and decline in mean trophic level (Pauly et al. 1998). These impacts on community structure and function have been widely documented and quantified in many marine ecosystems (Pauly et al. 1998, Lotze et al. 2006).

At the stock level, fishing can reduce target and non-target stocks, spawning potential and, possibly, population parameters such as growth rate, maturations, etc. (Garcia et al. 2003). Fishing can also modify age and size structure, sex ratio, genetics and species composition of target species, and of non-target species (Garcia et al. 2003).

At the ecosystem level, Ferretti et al. (2010) detected strong ecological effects, such as trophic cascades and changes in ecosystem control equilibrium such as top-down, bottom-up, wasp-waist controls by using the relationship between fishing and possible alterations of direct and indirect trophic relationships within impacted ecosystems. On the other hand, overfishing can shift original stable, mature and efficient ecosystems into a status that is immature and stressed (Garcia et al. 2003). These impacts can be caused in several ways such as targeting and reducing the abundance of high trophic level species, modifying the trophic chain and flows of biomass and energy across the ecosystem (Pauly 1979). Fishing can also induce changes in habitats by destroying and disturbing bottom topography, the associated habitats and benthic communities (Garcia et al. 2003).

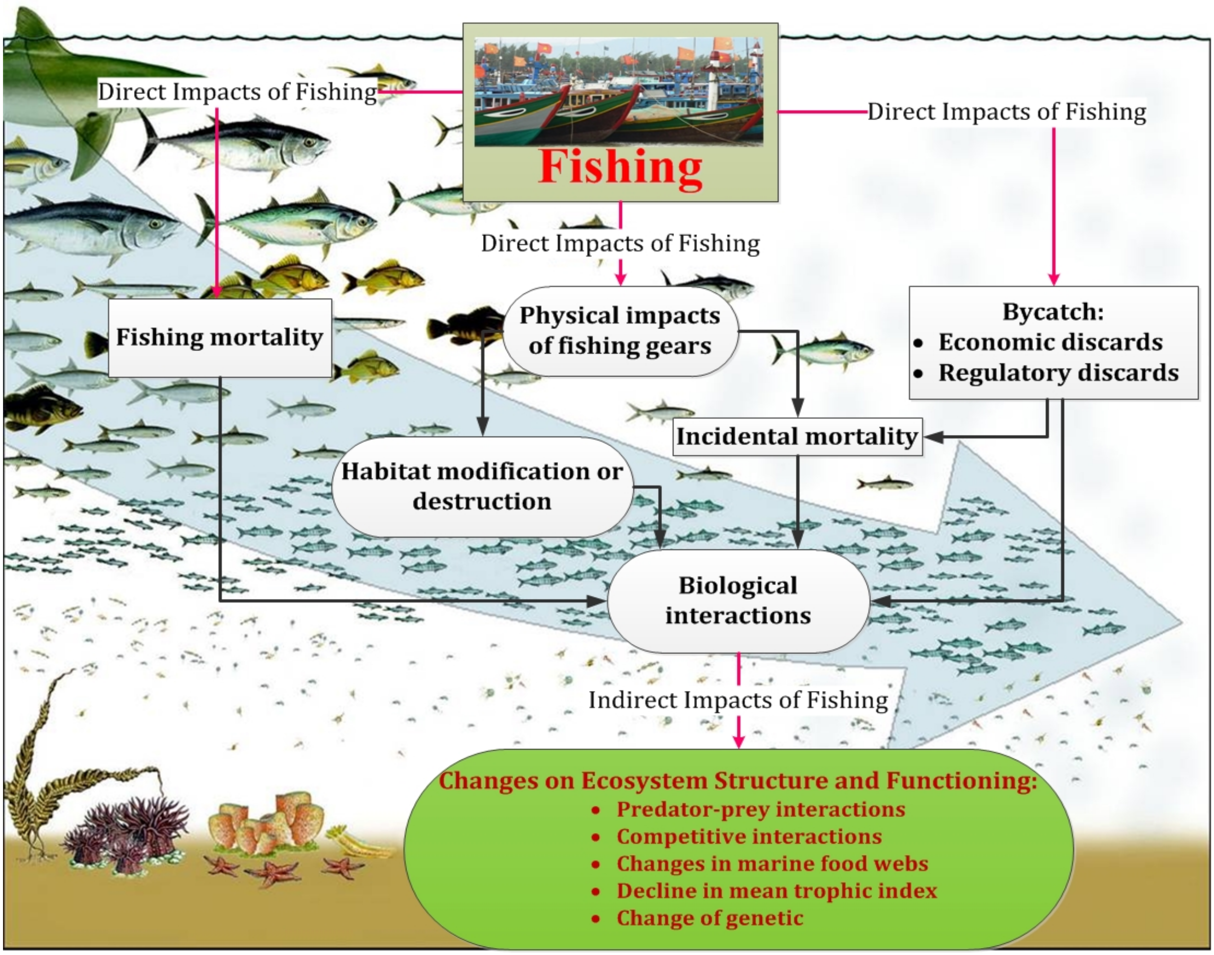

Figure 1.3. Impacts of fishing on marine ecosystem structure and functioning. The big arrow indicates phenomenon of the fishing down the marine food web that may occur if fishing at high intensity takes place over longer time. Adapted from Pauly et al. (1998).

1.3.1. Direct effects of fishing

According to FAO statistical data, global marine fishery production have increased from only 20 million tons in 1950 to more than 80 million tons in the 2010s (FAO 2014). Despite increasing levels of fishing effort 1990s to 2000s, the global yield of fish remained relatively constant for the period of 2010s (FAO 2014). This may be a signal of overexploitation causing resources to be at an unsustainable status (Pauly and Christensen 1995). In fact, the proportion of marine fish stocks that are within biologically sustainable levels declined from 90% in 1974 to 71.2% in 2011 (FAO 2014). In other words, 28.8% of fish stocks were estimated as being harvested at a biologically unsustainable level.

In addition, fishing gears (both active and passive gears) can also directly affect non-target organisms, typically referred to as 'bycatch' (Jennings and Kaiser 1998). The global bycatch proportion annually estimated was 40 % in the total annual global marine catch (Davies et al. 2009). According to recent assessment of Western and Central Pacific Fisheries Commission (WCPFC) at Pacific Ocean, billfish stock (i.e. bycatch of tuna fisheries) is overfishing (WCPFC 2014). Regarding to bycatch issue, FAO has developed international guidelines on bycatch management and discard reduction (FAO 2010) and has been urged to provide support in capacity building for their implementation within the ecosystem approach. However, bycatch remains a major concern affecting fisheries management (Davies et al. 2009, FAO 2014).

Impacts of fishing on physical disturbance are possible (Jennings and Kaiser 1998). Active fishing gear such as trawlers which are towed on seabed can cause extensive modification of seabed habitats and their associated benthic communities (Jennings and Kaiser 1998). The effects of disturbance of otter trawling on a benthic community were investigated with a manipulative field experiment by Schwinghamer et al. (1998). They concluded that tracks made by trawl doors were readily visible on the sea floor immediately after trawling, 10 weeks or even after 1 year. Their observations also revealed that organisms and shells tended to be organized into linear features parallel to the corridor axis. They also demonstrated that trawling reduces both surficial biogenic sediment structure and the abundance of flocculated organic matter.

Another direct effect of fishing is 'ghost fishing' that is caused by losing fishing gear at sea and this can lead to fishing gear lost continues to catch fish (Garcia et al. 2003). This is well known for passive fishing gears such as gillnet, traps, pots, trammel net, etc. Although the impacts of ghost fishing are basically unknown, there are indications that their effects are not negligible (Goni 1998). Not only fish but also sea birds, marine mammals, and sea turtles can be affected by ghost fishing (Garcia et al. 2003).

In conclusion, fishing has a number of direct effects on marine ecosystems because it is responsible for increasing mortality of target and bycatch species. Moreover, bottom trawling causes a major physical impact on the habitat of benthic organisms. Thirdly, ghost fishing via lost gear affects the marine ecosystem. All of these must be considered appropriately under implementation of EAFM.

1.3.2. Indirect effects of fishing

1.3.2.1. Effects of fishing on the structure of trophic networks

Structural attributes that appear to be constant, or at least regular, throughout the planet's latitudinal range are displayed in the trophic networks (Navia et al. 2012). The stability of networks and their capacity can be linked to these regularities to respond to different types of environmental stressors (Bascompte et al. 2005). These structural attributes are mainly created by some interactions between predators and preys, proportional abundance of predators, intermediate species and primary species, as well as the number of species at different trophic levels.

Despite their importance, few studies on the impact of fisheries to structural properties of trophic networks are available. Lotze et al. (2011) found significant changes in the constant proportions that must exist among top predators, intermediate species and primary species, known as "species scaling laws" (Briand and Cohen 1984). In addition, based on other structural indicators of trophic networks (link density, connectivity, cannibalism), Lotze et al. (2011) concluded that the trophic network in the Adriatic Sea has been subject to overfishing of high trophic levels, leading to its structural simplification, progressively becoming less connected and complex. This type of structural changes directly affects the capacity of the network to respond to species lost and increases the likelihood of secondary extinctions, even with low values of species reductions, leading the network to structural collapse more easily (Dunne et al. 2004).

Fishing can also cause change in the composition of fishery landings and have shifted from large piscivorous fishes toward small invertebrates and planktivorous fishes, a process now called "fishing down the marine food webs" (Pauly et al. 1998). This fishing phenomenon is known as an important effect related to the structure and composition of trophic networks. Given the interpretation to this phenomenon, fishing has substantially modified trophic networks, from being dominated by large predators of high trophic level to small species of lower trophic levels, fishing down marine food webs was initially considered an effect of negative consequences. It was documented both at a global (Pauly et al. 1998) and regional (Pauly and Palomares 2005) scale, as well as at a local scale in countries such as Thailand (Christensen 1998), Canada (Pauly et al. 2001), America (Steneck et al. 2004), Brazil (Freire and Pauly 2010), and many others.

On the other hand, Essington et al. (2006) proposed another effect caused by fishing and called this “fishing through the marine food webs”. This phenomenon is caused by an increasing harvest of low trophic levels in marine networks (sequential addition of new fisheries), even when catches of high trophic level species remain constant or increase (Essington et al. 2006). Litzow and Urban (2009) reported that the historical periods of decrease in the trophic levels of catches in Alaska obeyed to fishing through the food web and not fishing down the food web, adding as an argument that declines in the trophic level of catches are caused in many cases by temporary additions of fisheries targeting low trophic level species (e.g. crustaceans). Litzow and Urban (2009) concluded that it is clear that commercial exploitation has had profound effects on marine ecosystems in Alaska, but that due to the complexity of connections in marine trophic networks it is difficult to understand these effects. In terms of the ecological interpretation of fishing through the food web, Essington et al. (2006) noted that although they found increases in the catches of high trophic level species, this does not mean that these stocks are healthy and that their findings should not be used to make population inferences since they worked with species categories grouped by trophic level.

1.3.2.2. Effects of fishing on the functioning of trophic networks

Fishing does not only affect network structure but also the functioning of the trophic network. Different levels of fishing pressure can generate multiple effects on the function of species and their interactions. These effects are much more difficult to detect and assess than structural effects and often cause the largest changes in ecosystems because they link the different types of ecosystem control spreading across trophic networks. These mechanisms are referred to as top-down, bottom-up, and wasp-waist control (Pace et al. 1999).

Since fisheries have mostly targeted large species, which exert predatory functions within trophic networks. The most well known effects to date are those based on the decrease in abundance of these large species. A growing body of literature has reported a strong relationship between fishing and decreases in abundance of populations of top predators (Myers et al. 2007, Baum and Worm 2009, Griffiths et al. 2010). These reductions have been documented in coastal, benthic, demersal, and pelagic environments and are associated with different fisheries (Myers et al. 2007, Baum and Worm 2009, Griffiths et al. 2010). The decrease in top predator abundance has allegedly led to community restructuring, with their composition (richness and abundance) now being dominated by medium-sized species with lower trophic levels (Ellis et al. 2005, Myers et al. 2007, Lotze et al. 2011).

The on-going debate on whether top-down, bottom-up or wasp-waist processes controlling marine ecosystems is fundamental to understand how drivers of change affect ecosystem dynamics. The bottom-up control is a process with controlling of species at low trophic levels to higher trophic level upward (Frederiksen et al. 2006) and the top-down control is an opposite situation with controlling of top predators to their preys (Chase et al. 2002). On the other hand, an ecosystem is controlled by species at intermediate trophic level controlling the abundance of their predators through a bottom-up interaction and the abundance of prey through a top-down interaction called ''wasp-waist'' control (Cury et al. 2000).

When adopting the bottom-up control view, climate change seems to be considered as the major process behind recent changes in marine ecosystems (Beaugrand 2004, Frederiksen et al. 2006). In contrast, when adopting the top-down control view, Frank et al. (2005) reported that shifts in marine ecosystems are mainly caused by overfishing of top predators. In the 'wasp-waist' control, species the intermediate level that links zooplankton and top-predators is usually occupied by dominating a few pelagic forage fish species that has been suggested to control the marine ecosystem (Cury et al. 2000).

Perturbations at the base of the food web (bottom-up control processes) will propagate upward through the food web. Because a perturbation at the top is unlikely to cascade down the food web, such systems are relatively robust with respect to harvesting (Petrie et al. 2009). While bottom-up processes generally enhance ecosystem resilience, top-down interactions may result in trophic cascades and internal positive feedbacks within the food web (Frank et al. 2005). An ecosystem subject to strong top-down forcing is therefore expected to exhibit several alternative stable states under the same external conditions such as little complexity, low species richness, strongly interconnected and highly dependent on trophic interactions which make them more vulnerable to fishing (Frank et al. 2005).

In contrast, ecosystems in tropical latitudes seem to be somehow more resistant to the effects of harvesting since time series studies on composition, diversity, and volume of catches show much weaker effects than those recorded in cold and temperate ecosystems (Harris and Poiner 1991).

Although wasp-waist ecosystem control has been proposed for ecosystems (Cury et al. 2000), few studies assessed the structural significance of this control. This is because model ecosystems under wasp-waist are very sensitive to effects on key species (Jordan et al. 2005). There are two reasons explaining for this sensitivity of the model under wasp-waist control. First, because interactions between wasp-waist species (i.e. anchovies and sardines) are stronger than those between other species pairs and even if these two species do not have direct interactions between them, they share a large number of predators and prey. This allows for the change in their abundance to expand indirect effects such as "apparent competition" or "exploitation competition" (Menge

1995). Second, wasp-waist species have higher population self-regulatory values than those of other species and thus, they could cause cyclical (Hassell et al. 1976) and chaotic dynamics and unpredictable oscillations in nature (Bakun and Broad 2003).

1.4. Can we implement the ecosystem approach to fisheries management under data-poor conditions?

Many fisheries worldwide have limited data, particularly but not exclusively in developing countries. In some circumstances even catch statistics might not be reliable and effort statistics may not be available (Forrest 2008). Many of these fisheries often have only very general or no clear management objectives, and infrastructure and resources are insufficient to support comprehensive and continuous data collection, scientific research, and fishery management (Pilling et al. 2008).

Moreover, data-poor problems increase in long time series and macro-scale studies, which have the main purpose of examining changes in the dynamics of a whole ecosystem as the principles of EAFM require. Poor communication and coordination in policy formulation and the implementation of management plans among responsible management agencies and stakeholders are common problems that result in data-poor fishery situations (Hilborn and Walters 1992).

However, implementing an EAFM in these fisheries should still be feasible because the first need of the EAFM implementation is to set the objectives realistically and to develop management strategies (FAO 2003). This can be done effectively even with limited information in the face of uncertainty. Having clear management objectives, ecosystem-based holistic assessment tools using limited data and knowledge corresponding to that management objectives are required (Trenkel et al. 2007).

One approach, proposed for fisheries management that are limited in data and knowledge, is the development and selection of a suitable set of ecological indicators that can provide easily understood outcomes with a cost-effective manner (FAO 1999). The ecological indicators aim to describe as much of a system as possible in as few points as possible, in order to understand, evaluate and improve it (Pauly and Palomares 2005). In order to be useful for management purposes, ecological indicators should be sensitive to changes in ecosystem integrity through space and time, easily measured, understandable, informative and based on accessible data (FAO 1999).

At the macro-scale level, countries can use indicators to produce a holistic picture of the fisheries sector and its environment, while at the micro-scale level, indicators provide an operational tool in fisheries management, as a bridge between objectives and management actions. Like any reductionist approach, an indicator must be understood within its context. An indicator rarely captures the complete richness and complexity of an ecosystem, but a set of indicators may do, especially in all the data-poor cases in which they are the only tools that can be used.

There are several modelling tools that are available for data-poor areas. These models can be applied to allow extrapolating the missing data from the literature or from available sources (e.g. ecopath model) (Polovina 1984) or to use eco-physiological constraints from the literature and from site-specific knowledge (e.g. inverse model) (Vezina and Platt 1988).

1.5. Research gaps and objectives

Impact of fishing can cause inevitable changes on the ecosystem in many regions worldwide (Navia et al. 2012). Apart from some descriptive assessments and analyses using single-species approaches, no investigation of the sustainability of existing fishing patterns has been made at the ecosystem level for the fisheries of Vietnam (Anh et al. 2014a). A holistic fisheries management such as EAFM is currently lacking in Vietnam (Anh et al. 2014b). Sound and comprehensive scientific knowledge of the state of the ecosystem, the effects of human impacts and the vulnerabilities of ecosystem components and habitats are essential prerequisites for such approach (FAO 2008). Hence, it can achieve societal goals for both human developments and the health of the ecosystem. The EAFM also requires that the ecological, economic and social aspects of any activity or decision are simultaneously taken into account in a process that integrates all relevant sectors and stakeholders. Thus, to ensure that decision-making supports a sustainable use of ecosystem services and resources in an efficient and equitable way, it is fundamental that the social, economic and ecological impacts of fisheries are identified and quantified both on the short and long term. The main objective of this research thesis is to:

1. Evaluate the impacts of fishing on ecosystem functioning and structure of Vietnamese coastal ecosystem;

2. Investigate sustainability of existing fishing practices with consideration of economic, social and ecological aspects in the Vietnamese coastal fisheries, and

3. Propose suitable management policies and strategies for future considerations.

In the present research I provide tools and insights useful for implementing an EAFM in Vietnam. Important constraints such as economic, social and ecological aspects were also considered to support the decision-making processes of fisheries managers in the future.

1.6. Spatial and temporal scopes, and organization of thesis

Spatial and temporal scopes of this thesis were as follows: In Chapter 3, catch and effort data were reconstructed from 1981 to 2012 for entire Vietnamese EEZ and for all fisheries including inshore, coastal and offshore fisheries (Table 1.1). These reconstructed catch data were used as an input data source of Chapter 4 to calculate fishery-based indicators. In Chapter 5, only data set of 1990-1995 and 2000-2005 were selected and only for inshore and coastal fisheries of Vietnam (Table 1.1). Similarly, data set of inshore and coastal fisheries from 2000-2005 extracted from all fisheries was used as input data of Chapter 6.

Table 1.1. Overview of the spatiotemporal scale of the studied data.

	Chapter 3	**Chapter 4**	**Chapter 5**	**Chapter 6**
Studied temporal scale	1981-2012	1981-2012	1990-1995 and 2000-2005	2000-2005
Studied spatial scale	Entire Vietnamese EEZ	Entire Vietnamese EEZ	Limited on coastal ecosystem	Limited on coastal ecosystem
Fisheries data considered for	All inshore, coastal and offshore fisheries	All inshore, coastal and offshore fisheries	Only inshore and coastal fisheries	Only inshore and coastal fisheries

The thesis is organized in seven chapters (Figure 1.4).

Chapter 2 reviews existing tools to EAFM and proposes suitable tools for Vietnamese fisheries management when resources are limited.

Chapter 3 reconstructs catch and effort statistics from the Vietnamese fisheries to describe the fisheries in terms of developments on annual catch and effort and to explore if there were any changes in the fish community underlying trends in catch composition. In this chapter, we present different methodological approaches to recover time series catch and effort of the fisheries. These methodologies are useful in data-poor situations, when available data sources are only in a certain period with specifying by gears and species groups.

Chapter 4 evaluates the impact of fishing on the ecosystem structure based on simple fisheries-based indicators. This approach is suitable in assessment and management of fisheries in data-poor areas. We use a set of selected fishery-based indicators (i.e. marine trophic index (MTI), fishing in balance (FiB) and pelagic/demersal (P/D) ratio) derived from landing data and trophic level information to assess the ecological balance of marine ecosystems.

In **Chapter 5** we verify whether ecosystem functioning is affected by fishing intensity. In addition, we study the development level of this ecosystem and its state of maturity. This facilitates understanding of the function of the whole ecosystem for analysing the impact of human influences.

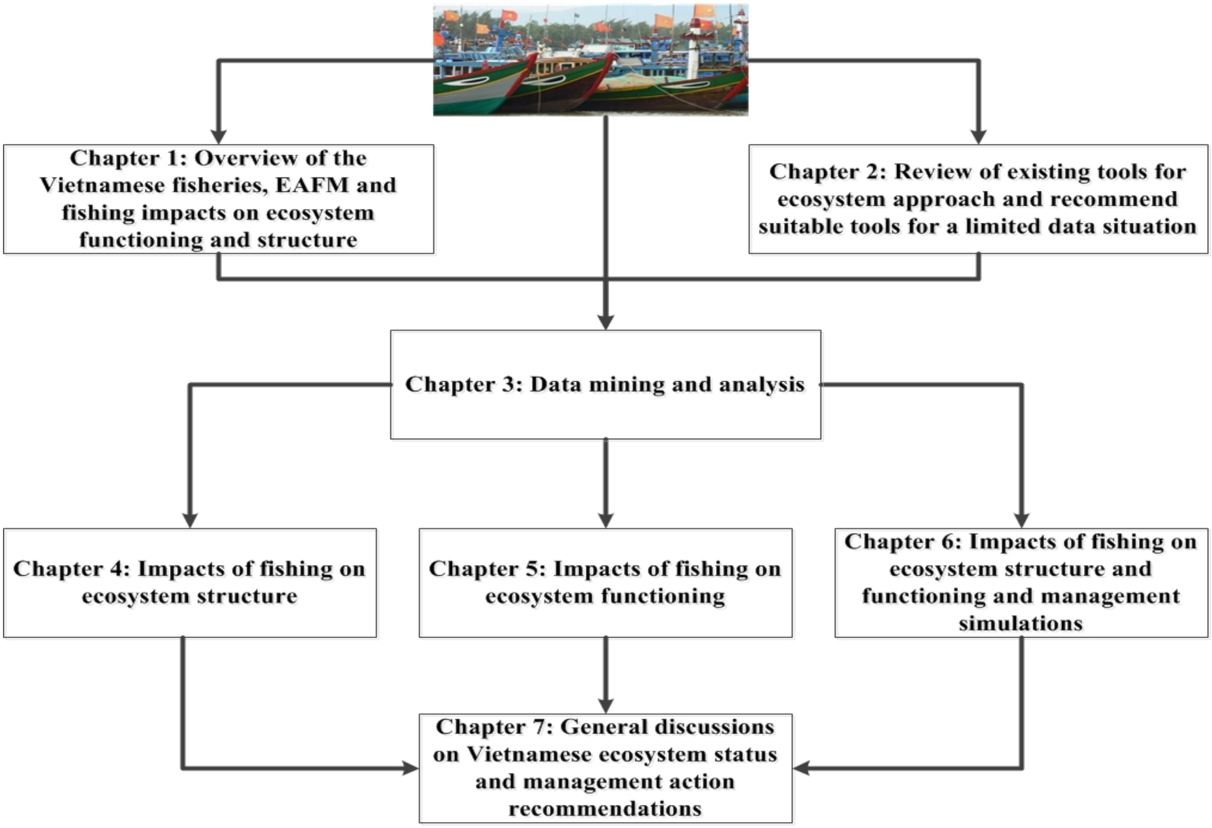

Figure 1.4. Flowchart of organization of chapters in the thesis.

In **Chapter 6**, an Ecopath with Ecosim trophic model is constructed to describe the structure and functioning of the coastal ecosystems supporting the Vietnamese fisheries. Some fishing scenarios are used to determine whether the existing levels of fishing could possibly be sustained via the consideration of social, economic and ecological aspects, and three different predator-prey controls. Suitable management scenarios were proposed for the Vietnamese coastal ecosystem management.

Chapter 7 provides general discussions, recommended management actions, and further research options.

Chapter 2: Towards an ecosystem approach for Vietnam's fisheries management: A review of existing tools

Redrafted from

Anh, P. V., F. De Laender, G. Everaert, C. T. Vinh and P. Goethals (manuscript in preparation). Towards an ecosystem approach for Vietnam's fisheries management: A review of existing tools.

2. Chapter 2: Towards an ecosystem approach for Vietnam's fisheries management: A review of existing tools

Abstract

Ecosystem approaches has been recognized as a necessary approach in fisheries assessment and management. Many tools exist to support an ecosystem approach to fisheries management. Tools for ecosystem approaches allow a better understanding of how species are influenced by each other and by human activities. They can be used both to quantitatively describe the structure and functioning of marine ecosystems and provide an indication of how these are likely to change in response to different ecological perturbations. Here, we provide an overview of the main six approaches that can be used to support EAFM and we selected three of them (i.e. ecological indicators, inverse model and Ecopath with Ecosim) to assess fishing impacts on ecosystem structure and functioning of Vietnamese marine ecosystem. The selections were based on their suitability on data-poor conditions in Vietnam but can sufficiently assess impacts on the Vietnamese marine ecosystem.

2.1. Introduction

In recent years, there has been a growing recognition of ecosystem approaches in fisheries assessment and management (EAFM) (FAO 2003). Many tools exist to support an ecosystem approach to fisheries management. These tools allow including interactions between species and assessing the impact of fisheries in a more relevant manner compared to single-species assessment approaches (FAO 2008). However, many of these EAFM tools are only suitable for data sufficient fisheries. In many developing countries, including Vietnam, the application of such tools is not straightforward due to the scarcity of data, lacking capacity of fisheries scientists/managers and the complexity of marine ecosystems. First, we give a general overview of six existing ecosystem tools that can be used to support EAFM. Final selections for application in Vietnam is based on criteria such as data and skill requirements, advantages and disadvantages. Three tools were selected and recommended as suitable under data-poor conditions, such as the Vietnamese marine ecosystem.

2.2. Existing tools for ecosystem approaches

Tools for ecosystem approaches allow for a better understanding of how species are influenced by each other and by human activities. They can be used both to quantitatively describe the structure and functioning of marine ecosystems, and to provide indications of how these are likely to change in response to different ecological perturbations. Due to limitation of fisheries data in Vietnam, it is necessary to develop holistic tools to support for EAFM that require less input data. Here, we provide an overview of the main types of approaches that can be used to support for EAFM. It includes a critical analysis of the methodologies, the advantages, disadvantages and limitations of each approach for limited data fisheries.

2.2.1. Fishery-based indicators

Understanding ecological interactions is a key point for an EAFM (Cury and Christensen 2005). As indicated in Chapter 1, ecological processes such as trophodynamic interactions, i.e. predation and competition, have been identified to be of paramount importance in fish population dynamics (Bax 1998). This involves two major problems of decreases in food resources and the indirect effect of decreasing fish biomass on the functioning of ecosystems (Cury and Christensen 2005). Therefore, there is a need for indicators (Murawski 2000) to reflect and describe the complex interactions between fisheries and marine ecosystem (Pauly and Watson 2005). Ecological indicators can be used to describe these in simpler terms that can be understood and used even by non-scientists for policy making. Therefore, indicators could be used to support the implementation of an EAFM by providing information on the status of the ecosystem, the extent and intensity of effort and mortality and the progress regarding to management objectives.

Pauly et al. (1998) have proposed to used marine trophic index (MTL) as an indicator for fisheries management. Although the MTL of catches is the index most frequently used to assess the status of marine ecosystems, it has been widely questioned (Caddy et al. 1998, Branch et al. 2010) because economic interests in the different fisheries influence it. In fact, in ecosystems where fisheries simultaneously harvest species at different trophic levels (multi species fisheries), changes in the MTL become masked and the index remains more or less stable with time, potentially giving the impression of a sustainable fishery through time (Perez-Espana et al. 2006). Therefore, MTL is suggested to use in combination with other indicators to identify structural changes in trophic networks and to detect possible consequences of these changes on network function.

In addition, the fishing in balance (FiB) index is also proposed as a relevant index to reflect ecosystem changes related to catch composition (Garcia and Staples 2000). This index considers whether the increase in landings because of fishing on lower TL corresponds the ecological appropriate increases (determined by the transfer efficiencies between TL's). The FiB index will stay constant if a decrease in the average trophic level is matched by a sufficiently large increase in the catch. When the FiB index decrease there may be indication that fisheries take out so much biomass from the ecosystem that its functioning is impaired (Pauly and Watson 2005). A decline in FiB will also indicate that discarded catches has not been reflected in the total catches that are used to calculate FiB (Pauly and Watson 2005). An assumption required in calculation of the FiB index is that transfer efficiency is constant (and known sufficiently well) across trophic levels (Pauly and Watson 2005).

2.2.2. Multi-species virtual population analysis model

Multi-species virtual population analysis (MSVPA) is an approach that has been proposed by the International Council for the Exploration of the Sea (ICES) to manage commercially important stocks in the North Sea and the Baltic Sea since the 1980s (Lewy and Vinther 2004). Required input data of the model include estimates of catch-at-age in numbers (C), fishing mortality rates (F) in the terminal year and oldest age classes, and residual natural mortality rates (M1). Requirements of such data inputs (e.g. fisheries catch-at-age) is one of the main advantages of MSVPA development because it is similar as the data needed for standard single species models such as SSVPA. Model outputs can be directly compared to those of single species approaches, which simplifies their incorporation into fishery management (Garrison and Link 2005). The MSVPA model is more realistic than SSVPA, because SSVPA assumes that the natural mortality rate does not change over time and throughout age classes, while MSVPA splits the natural mortality into two components: predation (M2), which depends on time and age because of variations in predator abundance and prey selection, and residual mortality (M1), which depends on additional non identified factors. MSVPA can be run on Windows PC using Microsoft Office Tools (Excel).

The disadvantages of MSVPA approach is that it provides an incomplete picture of ecosystem processes and dynamics (Garrison and Link 2005). For example, only exploited species are included while other components of the ecosystem are not considered or are included implicitly as fixed inputs of biomass (Garrison and Link 2005, Garrison et al. 2010).

2.2.3. Spatial environmental population dynamic model

The spatial environmental population dynamic model (SEAPODYM) was initially developed as a model for investigating spatial tuna population dynamics in the Pacific Ocean, under the influence of both fishing and environmental effects (Lehodey et al. 1998). It is an age-structured population and 2D coupled physical-biological interaction model (Bertignac et al. 1998, Lehodey et al. 1998). The model also includes a description of multiple fisheries and predicts the spatio-temporal distribution of catch, catch rates, and length-frequencies of catch based either on observed or simulated fishing effort, allowing respectively to evaluate the model or to test management options (Lehodey 2005). The model is a combination of a movement model which is based on a diffusion–advection equation in two horizontal dimensions (Lehodey 2001) with an age-structured population model of the targeted species (tuna) (Lehodey et al. 2003). Environmental and spatial constraints are used to determine the movement and the recruitment of tuna. The movement of adult tuna in the model has been obtained on tagging studies to analyse their movements. The movement of small forage organisms, tuna larvae and juveniles is modelled using advection by ocean currents in two horizontal dimensions (Lehodey 2001). Detailed equations of transport for tuna populations are given in Bertignac et al. (1998) and Lehodey et al. (1998). Parameters such as sea surface temperature, ocean currents and primary production can be retrieved from coupled physical-biogeochemical models, as well as from satellite data (Lehodey et al. 2003). In addition, data about fishing effort and catch by different fleets are used as input information. The model outputs are information on recruitment and biomass by different regions and fisheries to allow realistic prediction of the large-scale distribution of the species (Lehodey 2001).

For technical aspects, SEAPODYM is programmed with source code in the language C++ that makes it less convenient for practical use by marine system managers (Lehodey

2005). In addition, SEAPODYM only represented a small subset of the species in the ecosystem at high trophic level and thus it is impossible to explore trophic interactions at all levels. It also requires a lot of data, making it costly and difficult to use (Whippe et al. 2000).

2.2.4. Atlantis model

Atlantis has been initially developed by Fulton et al (2004) as an ecosystem model that is intended to support and evaluate management strategies. The model has been used by several other authors such as Cochrane et al. (1998), Butterworth and Punt (1999), Sainsbury (2000) and it has been applied to multiple marine systems (from single bays to millions of square kilometres) in Australia and the United States (Fulton et al. 2004). Atlantis consists of multiple modules (submodels) which include several steps in the management strategy as well as adaptive management cycles (Fulton et al. 2004). Modules include biological, physical and fisheries models and they have a varying degree of complexity. The main component of Atlantis is the biophysical module that is deterministic and spatially-resolved in three dimensions. This model simulates the nutrient (usually nitrogen and silica) flows through the main biological groups in the marine ecosystem of interest. The primary ecological processes considered in the model are consumption, production, waste production and cycling, migration, predation, recruitment, habitat dependency, and mortality (Fulton et al. 2004). Atlantis also features a detailed exploitation submodel that can deal with the impact of pollution, coastal development and broad-scale environmental change, but is focused on the impact of fishing fleets. The exploitation module interacts with the other modules called the sampling and assessment module to simulate fisheries dependent and independent data with realistic levels of measurement uncertainty (bias and variance). By doing so, a wide range of combinations of fisheries and survey information can be simulated. The model requires the most intensive data collection efforts in comparison with other models for fisheries management. It needs phytoplankton production parameters such as maximum temperature-dependent growth rate, light limitation factors and half saturation constants. Moreover, the model also requires configuration of food web connections. For each species, the model needs the parameters consisting of abundance per area, individual growth rates, length weight conversions, maximum age and age at maturity, general habitat preferences, dispersal and/or migratory characteristics, within and outside model, diet data and recruitment parameters. The model outputs include species/group biomass by different scenarios and these outputs are used as an input for the management module that is typically a set of decision rules and management levers. The management model in Atlantis is currently only detailed for the fisheries sector. The model is coded in C++ and could run on Linux and PC but required advanced hardware configurations.

Due to huge data requirements, Atlantis model was evaluated as the most unsuitable model in terms of suitability to be applied on data-poor areas.

2.2.5. Inverse model

To quantify food webs we rely on the crucial merge of field observations and models, and these are called inverse data assimilation techniques (van Oevelen et al. 2010). Data assimilation refers to data integration within a model structure and inverse means that data from field observations are used to reconstruct the underlying model parameters. Inverse methods are highly considered in geophysical sciences (Lary 1999, Wang et al. 2000), where data inferences can only be made indirectly.

Advanced applications of ecological data assimilation in food web models are mainly found in the marine realm (e.g. Vallino (2000)) and rely on fitting model parameters to observed time series data or spatial distribution patterns, after which the food web flows are recovered. These are called non-linear inverse technics. However, these non-linear inverse problems are solved with complex numerical techniques with long computer time. These technics are also complicated to implement and do not always retrieve the optimal parameter set (Soetaert et al. 2002). In addition, there are some additional reasons limiting the use of these mechanistic modelling approaches in ecology (Gaedke 1995). Firstly, ecological sciences typically deal with poorly or partly understood mechanisms to describe complex processes. Secondly, parameter values are often not known accurately enough to determine a start range from which they can be fitted. Thirdly, data sets are usually too sparse to constrain all parameters of a mechanistic model.

In contrast to these mechanistic models, the linear inverse models (LIM) require less data and describe the food web as a linear system of flows that interconnect the compartments. The flows are quantified by solving a mass balance equation supplemented with site-specific and literature data. Their simplicity allow these LIM as a widely applied technique, which are also known as inverse analysis (Klepper and Van de Kamer 1987, Vezina and Platt 1988).

The general structure of an inverse model includes (i) compartment mass-balance equations, (ii) data equations, and (iii) constraints (Savenkoff et al. 2007, De Laender et al. 2010)). The mass-balance equations specify that, for each compartment, the sum of inflows (consumption for each consumer group) is balanced by the sum of outflows (production, respiration, and egestion for each consumer group). In addition, the data equations attempt to fix the value of certain flows or combination of flows (i.e., incorporate the observations into the model that coincide with the period/region for which a solution was derived), whereas the constraints incorporate general knowledge into the model.

The inverse model is solved by assuming steady state of all compartments. Under the steady state assumption, consumption representing the input must balance the sum of the outputs consisting of production, respiration, and egestion (flux of unassimilated food: faeces or detrital flow) for individual compartment (Vezina and Pahlow 2003). Because of a relatively low data requirement, the inverse model is considered as a suitable tool in data-poor areas.

2.2.6. Ecopath with Ecosim model

The Ecopath and Ecosim modelling tool (EwE) is composed of a core mass balance model (Polovina 1984, Pauly et al. 2000, Christensen and Walters 2004) from which temporal and spatial dynamic simulations can be developed (Christensen and Walters 2004). This tool has been widely used to quantitatively describe aquatic systems and the ecosystem impacts of fishing (Christensen and Walters 2004).

Trophic flows between discrete trophic levels are the basis for the food chains (Lindeman 1942) and thus species or groups are assigned to distinct trophic levels and positions in a food web. Polovina (1984) developed the first Ecopath model in the Northwestern Hawaiian Islands based on the above theory and using the concept of mass balance and energy conservation. Christensen and Pauly (1992) further developed the model including fractional trophic levels of species that feed across a range of trophic levels. After that, the scope of Ecopath has been more developed by introducing the trophodynamic simulation model Ecosim to conduct multispecies simulations to explore ecosystem structure and functioning, the impact of fishing, policy exploration (Christensen 2005).

EwE was designed for straightforward construction, parameterization and analysis of mass-balance trophic models of aquatic and terrestrial ecosystems (Christensen 2005). Since its initial development in the early 1980s (Polovina 1984), EwE was the first model to apply a type of statistics called "path analysis" to the field of marine ecology. It has now been widely used for constructing food web models of marine and other ecosystems (Christensen and Pauly 1992, Christensen 2005).

The input parameters of an Ecopath model requires diet composition, fishery parameters (landings and discards by gear type) and input of three of the following four basic parameters: B, P/B or its equivalent total mortality rate (Z), Q/B, and EE for each group (i) in a model. Normally B, P/B and Q/B are entered for all groups and EE is estimated since it is impossible to obtain this parameter from its field estimation. The EE expresses the proportion of the production that is used in the system and can be considered an expression of model uncertainty rather than a meaningful ecological term (Christensen and Walters 2004). For parameterization, Ecopath sets up a series of linear equations for each group to solve the unknown or missing parameters establishing mass-balance in the same operation. Ecosim is a dynamic simulation tool (Walters et al. 1997). It has been developed to test the effects of given modifications on the ecosystem (new policies, increased fishing effort, etc.). Its goal is to help select the best alternative for the ecosystem as a whole, and not only for a single species.

EwE is simultaneously developed in same platform called EwE. With simplicity and ability to accurately identify ecological relationships, EwE has revolutionized scientists' ability worldwide to understand complex marine ecosystems. EwE requires identifying and quantifying feeding relationships between various living resource stocks in an aquatic system. The feeding relationships require estimates of biomass of each living resource and feeding rates of a predator on a prey item or group. Mortalities from predation as well as harvested catches from fisheries and any other death terms are also important to predict yields of each trophic level; factors that alter these trophic levels can be assessed for impacts on any organism or group of the ecosystem. These factors, for example, might be the management policies that can be applied in the system (fishing limits, gear types, etc.) or natural control through events such as storms, hurricanes, disease, and parasitism. It is important to realize that applicability of the EwE approach is to answer simple, ecosystem wide questions about the dynamics and the response of the ecosystem to anthropogenic changes. Thus, management policies can be designed for implementing ecosystem based management principles, and can provide insight into the changes that have occurred in ecosystem over time.

2.3. Tools recommended for ecosystem approach to fisheries management in Vietnam

There is the increased emphasis by the fisheries management community to move away from single species management and towards an ecosystem approach (FAO 2008). Inherent in this effort is the need for an understanding of the interactions between harvested species, their prey, predators, and competitors. Many ecosystem tools have been developed to address current fisheries questions. However, each of them has advantages, disadvantages and limitations (Whippe et al. 2000). It is necessary to evaluate these techniques for ecosystem-based management. In the current fisheries management context in Vietnam, there are some criteria to evaluate regarding which models might be most suitable in terms of their application to ecosystem-based management. There is initially a need to address research and data availability questions as: (1) ability of the tool to identify suitable management measures, (2) ability to use in poor data situations, and (3) applicability to multi-gear and multi-species conditions.

Table 2.1. Overview of some general aspects to apply tools to support ecosystem approach to fisheries management.

Tools	Data requirements	Requirement of programming skills	Requirement of mathematical skills	Advantages	Disadvantages
Fishery-based indicators	Low; only landing data	No	Low	Lower data requirement, easy to calculate, easy to interpret, can be used to partly evaluate changes on structure and functioning of ecosystem	Subjective interpretation; should be interpreted from several combined indicators
MSVPA	Detailed stomach content data input to model makes it unsuitable for most regions	High	Fairly high	Large concerted effort concentrated on approach with attendant large sampling effort and studies to test underlying assumptions plus subsequent efforts to improve and modify approach	Data hungry; lack of statistical structure to take account of uncertainty in parameter estimates
SEAPODYM	Data intensive hence not suitable for data-poor areas	High	Medium level required	Attempts to incorporate environmental data directly into a spatial population dynamics simulation model; novel movement model	Insufficient resolution of mid-trophic levels to explore trophic interactions at all levels
Atlantis	High including biomass, production, consumption, diet composition, nutrient and climate data.	Very high	Fair level required	Spatially-explicit biomass dynamics response to different fisheries management scenarios	Data intensive and no easy user interface
Inverse model	Low; can be referred from related ecosystems or solved from the model	Moderate	Low	Low data requirement because of ability to refer from related ecosystems, Detail representation on all trophic levels	No potential consideration on environmental variation, Lack of ability to perform management scenarios
Ecopath with Ecosim model	Low; can be referred from related ecosystems or solved from the model	No	Low	Moderate data requirement, easy to use, input parameters easily inherited, very good on conducting fisheries management measures	No potential consideration on environmental variation

2.3.1. Ability to identify suitable management measures

Fisheries management aims to regulate fishing mortality rates over time and to achieve defined sustainability objectives. Modelling will play an important role for providing insights into what level these maximum mortality rates should be, and how they should be adapted over time. The impacts of alternative exploitation patterns can be explored using different approaches in the ecosystem models like EwE or linear inverse model. By exploring impact of exploitation by different patterns, fishing mortalities can be indicated by either sketching over time and evaluate the results, or by a formal optimization routine to evaluate the fishing effort over time that would maximize particular performance measures for management (Christensen and Walters 2004). These assessments can range from impacts of bycatch and habitat damage effects by some fishing activities, to impacts of fishing on capabilities of stocks to support other valued species. When policies of harvest controls have been based only on reference points from single species assessments, even including bycatch mortality effects, these policies ignore ecological interactions entirely because typical single-species models make particular assumptions about how natural mortality and recruitment rates somehow remain stable despite the fact that it can be changed in the ecosystem (e.g., changes in predation risk and food availability). Recognizing these restrictive assumptions, modellers have invested in the development of models that account explicitly for at least some major trophic interaction effects.

In Vietnam, since current policies for fisheries management are based on only information from single-species assessments, these policies will be very restricted in their application to the situation of multi-species and complex ecosystem. Thus, it is necessary to develop ecosystem approaches in order to evaluate entire marine ecosystems and establish reasonable fisheries management policies for sustainability of these marine ecosystems.

2.3.2. Data requirements and suitability for data-poor situations

Data and information are the basis inputs to ecosystem models. These models can help to evaluate whether management measures are appropriate or not. Because EAFM is a broad management process to manage many components in the ecosystem and with participation of different stakeholders, the data and information requirements are also more demanding. However, it is important to stress that immediate action should be based as much as possible on data and information that already exist. In Vietnam, some information has already been available in reports and statistics from various research institutes, agencies and ministries. However, these data and information have not been integrated for overall fisheries monitoring and evaluation purposes and therefore, there is a need to collect and use that available information for developing relevant EAFM tools.

In data-poor situations, evaluation tools such as ecological indicators, Ecopath with Ecosim (EwE) and inverse models can be suitable. In fact, ecological indicators have been successfully applied for some areas under data-poor conditions (e.g. Vivekanandan et al. 2005, Bhathal and Pauly 2008, Babouri et al. 2014). Of those, Babouri et al. (2014) also used reconstructed catch data to calculate several ecological indicators for Algerian fisheries. They have found that ecosystems at both national and local levels were excessively exploited, and were altered by overexploitation and probably eutrophication. These foundations have provided suitable advices for fisheries management in Algeria.

Inverse models have successfully been applied, for example in Canada (Savenkoff et al. 2001, Savenkoff et al. 2004, Savenkoff et al. 2007), the Arctic Ocean (Forest et al. 2011, van Oevelen et al. 2011), the North Atlantic (Daniels et al. 2006), the northeast subarctic Pacific (Vezina and Savenkoff 1999). In the studies in Arctic Ocean, van Oevelen et al. (2011) used linear inverse modelling to decipher carbon flows among the compartments of the benthic food web. Forest et al. (2011) also used inverse models to evaluate the food web at the Amundsen, Beaufort Sea that is covered by ice from October to late December. Therefore, it is note that as an effective tool in data-poor areas, inverse models can be used to quantify food webs in limited data conditions, for instance on deep-seas or ice covered areas where sampling methodologies are difficult to deploy.

EwE has been applied in almost all ecosystem types in the world such as lakes, aquaculture systems, estuaries, small bays, coastal systems and coral reefs, shelf systems, upwelling systems, and open seas (Morissette 2007). It has also been applied in all fisheries over the world from rich to poor data fisheries.

The development of ecological indicators can be based on simple data such as landings that are usually available at many fisheries. The EwE and inverse models allow modellers to extrapolate the missing data from the literature or from existing available sources (e.g. diet composition referred from Fishbase). This will be necessary and useful for fisheries monitoring and management activities in Vietnam where fisheries data are usually limited due to lack of financial and human resources.

2.3.3. Applicability to multi-gear and multi-species situation

There is an increased emphasis by the fisheries management community to move away from single species management and move towards an ecosystem approach to fisheries management especially in the situations of multi-gear and multi-fisheries as in tropical countries. Inherent in this approach is the need for an understanding of the interactions between harvested species and their preys, predators, and competitors. With the need to understand in detail the nature and dynamics of exploited marine ecosystems, and more precisely the complexity of species interactions, the development of the ecosystem approach for management of marine systems is becoming more and more important.

Ecosystem tools allow for a better understanding of how species are influenced by each other and by human activities. They can be used both to quantitatively describe the functioning of marine ecosystems, and to provide indications of how these are likely to change in response to different ecological perturbations. The situation is complex because the multi-species resource is selectively and non-selectively exploited by making use of a large number of diverse fishing gears.

The Vietnamese marine ecosystem is located at a tropical area and thus it is characterized by very high biodiversity and complexity (Spalding 2001). However, the complex food web of the marine ecosystem and its functioning are basically unknown. Although the biology and population dynamics of some key marine species have been studied individually providing estimations of population parameters, mortality, stock size and recruitment (RIMF 2005a), these studies provide insufficient information to plan sustainable resource management for this ecosystem. Moreover, no attempt has been made to understand the population dynamics at an ecosystem level through trophic links in the Vietnamese marine ecosystem. Therefore, there is an increasing demand to develop ecosystem models to consider interactions between and among compartments (Christensen and Pauly 2004). These tools also can be used to evaluate multi-gear and multi-species situation in Vietnamese fisheries.

2.4. Conclusions

In tropical countries such as in Vietnam, fisheries data are often scarce (Pilling et al. 2009) and only landings data of target species are available to assess the status of fisheries. This is because of the large heterogeneity of fishing activities, comprising of a wide variety of target species, gear types, landing sites, and distribution channels (Lunn and Dearden 2006). Consequently, data on catch and effort often lack essential details. If collected, catches were mostly identified to only higher taxonomic levels. Despite above numerous shortcomings, judicious selection and use of suitable assessment tools applicable for limited data situation may still provide useful insights on assessment of status and changes of the ecosystem (Christensen et al. 2009). In this chapter, we selected three tools (i.e. ecological indicators, inverse model and Ecopath with Ecosim) to assess fishing impacts on ecosystem structure and functioning of the Vietnamese marine ecosystem. In chapters 3 and 4 fisheries data are used covering the entire Vietnamese EEZ between 1981 and 2012 (Table 1.1). In chapters 5 and 6 only coastal data were used. The used methodologies aim to address and assess past, present and future impacts of fishing on both structure and functioning of the Vietnamese coastal ecosystem.

Chapter 3: Reconstruction of effort and catch data of Vietnamese fisheries

Redrafted from

Anh, P. V., F. De Laender, G. Everaert, C. T. Vinh and P. Goethals (manuscript in preparation). Reconstruction of effort and catch data of Vietnamese fisheries.

3. Chapter 3: Reconstruction of effort and catch data of Vietnamese fisheries

Abstract

Reliable and disaggregated time-series catch and effort data are crucial and fundamental for marine stock assessment and management. However, such data are not always available in developing countries due to resource capacity limitation in data collection. Several approaches were used in this study for the reconstruction of time-series on fishing effort and catch from 1981 to 2012 for the Vietnamese fisheries. Data reconstruction indicates that there was a stable trend on number of fishing vessels for all gear types during 1981 to 1985 and a slight increase from 1990 to 1995. Total number of fishing vessels for all gear combined increases four times from 1981 to 2012 (from 30,000 to 120,000 vessels). Total catch by gear types and ecological groups was also reconstructed and the trend on this reconstructed catch was similar to the total number of fishing vessels. Although there is a level of uncertainty associated with reconstructed data, outcomes from this study will provide the necessary input for further studies attempting to assess and manage the Vietnamese marine ecosystem.

3.1. Introduction

Fisheries resources managers must rely on several important factors such as stock abundance to determine the status of a fishery for management policy making (Sparre 1991). Abundance estimates themselves rely heavily on catch and effort statistics (Sparre 2000). Catch statistics are also used to monitor quotas, estimate fishing mortality and in conjunction with effort data to calculate catch per unit effort (CPUE). Reliable total catch and effort data, jointly with biomass and fish productivity estimates, are therefore crucial components of effective assessment and management of fisheries (Sparre 2000). Underestimation or overestimation of these variables can lead to biased stock assessment models (Punt et al. 2006). This is especially important for fisheries that are managed under annual catch limits and/or catch share programs, requiring timely and accurate total catch information to ensure that allocated harvest amounts are not exceeded.

Fisheries data can be collected in two ways. A first way is via the fishing industry, i.e. logbooks provided by fishers, observer data from fishing vessels and landing data collected at landing sites (Figure 3.1). These data are called 'fisheries dependent data'. This is the cheapest way to collect fisheries data, but they do not provide oceanographic environmental data and sometimes fishing locations recorded by fishers in logbooks are incorrect because they do not want to disclose their fishing grounds. Although these problems can be supplemented and crosschecked by using data vessel monitoring system (VMS), it is impossible for Vietnamese fisheries because it is not obligated to install the VMS in such small-scale vessels. A second way is to obtain fisheries data from independent surveys ('fisheries independent data'). When using this approach, fisheries scientists conduct independent surveys at sea using research vessels. However, for Vietnamese fisheries, both methods are very difficult to implement. Lack of compliance is a main challenge when collecting fisheries dependent data. Many of the local (artisanal) fishers do not provide the logbooks although this has been imposed by the legislation (MARD 2013). The fisheries independent surveys are a very expensive method and therefore it is not always a practical way in Vietnam. Even if a data collection system is set up, it is difficult to be maintained because of the instability on structure of fisheries management agencies and constraints on both human and financial resources. This makes it difficult to obtain time series data in many developing countries. As a consequence, there are inconsistencies and gaps in the available data which hampers sound stock assessment and management in Vietnam. In Vietnam, although both fisheries independent and fisheries dependent data collection programs are actually implemented, the later is being paid more attention to due to lower expensive to collect than fisheries independent data. In this reconstruction work, we also used fisheries dependent data as the basic data source and fisheries independent data as a supplemented source.

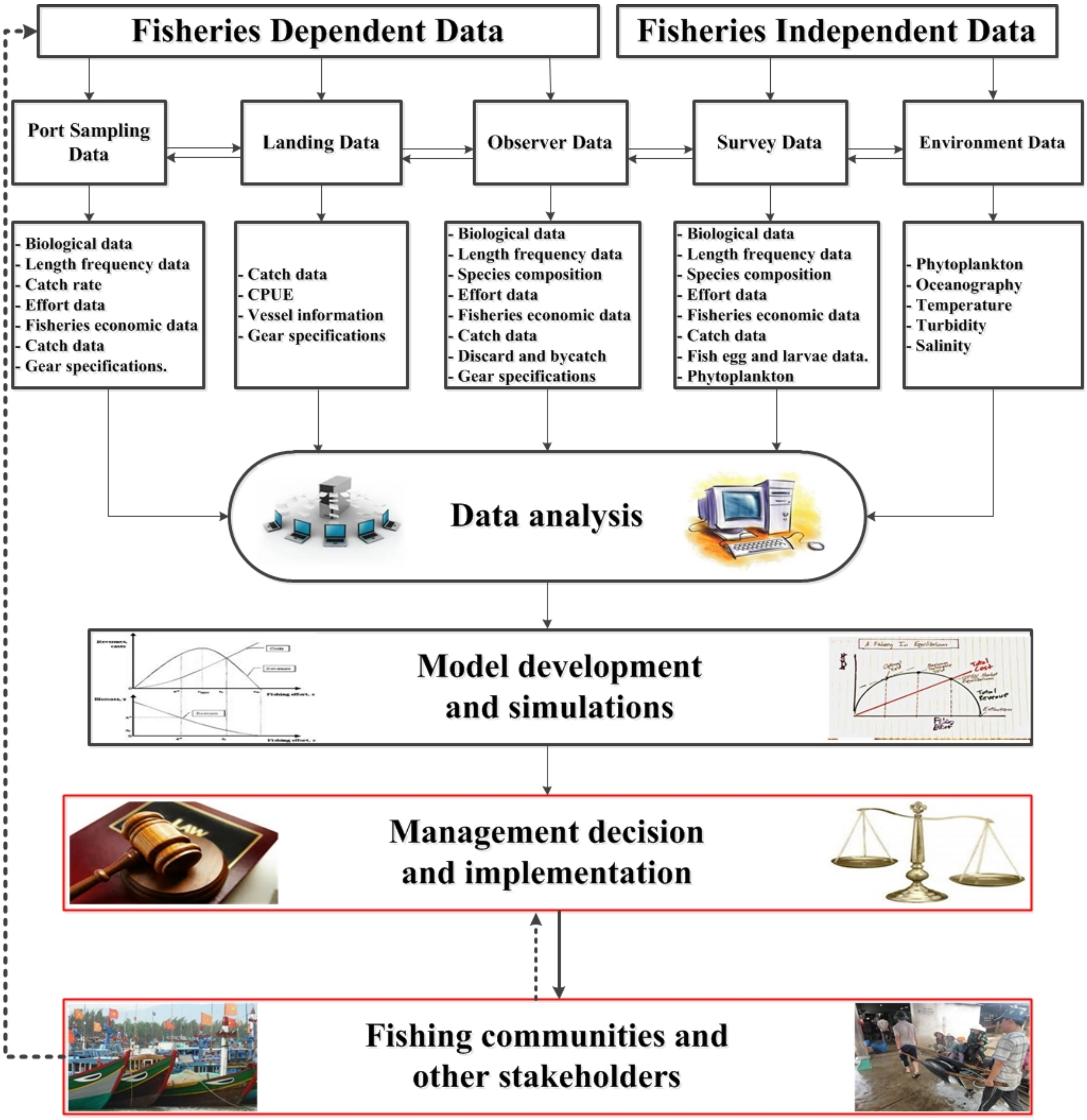

Figure 3.1. A theoretical framework of data sources and fisheries management processes in Vietnam. The dotted lines indicate feedback loops.

Vietnamese marine fisheries have been developing rapidly and have significantly contributed to socio-economic development and food stability in Vietnam. However, it is questionable how sustainable fisheries management can be achieved in a context of lacking scientific data (Tuan 2012). Although local authorities in 28 coastal provinces (Sub-Department of Capture Fisheries and Resources Protection, Sub-DECAFIREP) of Vietnam have recorded the total number of vessels operating entire Vietnamese EEZ, details of the number of vessels classified by gear have only become available in recent years (DECAFIREP 2013). Catches of Vietnamese fisheries are often not recorded or only recorded as a lump sum of all combined species by local authorities or several single national projects.

Reconstruction of time series catch data in the past has been carried out in many regions and countries in the world (Zeller et al. 2006, Zeller et al. 2007, Thurstan et al. 2010, Tesfamichael and Pauly 2011, Ulman et al. 2013). Zeller et al. (2006) have reconstructed the coral reef fisheries catches in American Samoa from 1950 to 2002 using long-term interpolation and concluded that there was a relationship between consumption of per capita catch and total catch of communities. Tesfamichael and Pauly (2011) have used four different methods to reconstruct total catches of fisheries in the Red Sea. They used catch data inferred from processed seafood products, on-board observation of discards, formal and informal purchasing information from markets, and interviews. Whereas, Watson and Pauly (2001) used statistical modelling to relate oceanographic data to fisheries production. In summary, there is no definite procedure to reconstruct or estimate catch but there is only a common purpose from different approaches to get as close to accurate numbers as possible (Tesfamichael and Pauly 2011).

The main objective of this chapter is to reconstruct fisheries effort and catch data of the Vietnamese fisheries from 1981 to 2012 for entire Vietnamese EEZ and for all fisheries fishing in inshore, coastal and offshore areas except oceanic tuna fisheries. Here, we grouped fishing gears by seven main types (i.e. fish and shrimp trawls, purse seine, gillnet, fish and squid handlines, and mixed gears) based on the current vessel register system implemented in 28 coastal provinces. Details of number of vessel by fishing gears are indicated in the Appendix 3.1. Then, the catch composition was calculated for each gear category. Dividing the catch by gear is based on practical availability of the information from fisheries surveys, observer and logbook data. The output of the present study will be a time series of effort data and standardized catch landed in Vietnam for Vietnamese vessels. Despite the above, no attempt is made in this chapter to draw inferences on the state of the fisheries resources. Such an attempt is made in following chapters, using different approaches, based on the catch data presented here, and time series of fishing effort aggregated by gear type.

3.2. Materials and methods

3.2.1. General method of reconstruction

The main methodology in catch and effort reconstruction is searching for different sources reporting from the provinces, critically analysing them, and organizing them to a common standard, which can be used for comparison and carrying out analyses for the assessment of the resources. The sources include peer-reviewed published papers (if available) and grey literature (mainly from the government and consultant). The information collected was enriched by the insights of local experts and colleagues who provided data through personal communications.

3.2.2. Data sources

3.2.2.1. Reconstruction of effort data

Effort records (number of vessels) by gear types were not available for the 1980s and the early 1990s. Although there were estimates of the total active number of vessels, these data were aggregated by all vessels from 1981-1995. Total vessel data were only disaggregated by gear type since 1996, when a project funded by the Danish Government started (Tuan 2012). From 1996 up to 2010s, the enumeration of the total annual fishing vessels has been conducted by the provincial Department of Capture Fisheries and Resources Protection (Sub-DECAFIREP) at 28 coastal provinces in Vietnam. Therefore, the total number of fishing vessels by gear type was reconstructed for 1981-1995 and for some missing data years (i.e. 2006 and 2008) using both linear interpolation and extrapolation from the years that effort data were available (i.e. 1996-2012 excluding 2006 and 2008), and total number of combined vessels collected from vessel registration system. The total number of combined vessels was previously reported by local authorities. Although, total vessels by gears were previously reported for 2006 and 2008 but these figures were estimated from uncertain sources, the effort data of these years were rejected and were reconstructed in the present study.

Total effort reconstruction was broken down for seven gears of fish trawl, shrimp trawl, gillnet, purse seine, fish handline, squid handline, and mixed gears denoting lift net, stick net and traditional fisheries.

3.2.2.2. Reconstruction of catch data

Fisheries catches have been successfully reconstructed in other regions of the world (Zeller et al. 2006, Zeller et al. 2007). Here, we applied the following approaches to reconstruct historic marine fisheries data for Vietnam:

a) Identify existing data sources (logbook, landings, observer data), reported aggregated catch times-series data;

b) Develop data anchor points that are used to interpolate or extrapolate;

c) Interpolate or extrapolate for time periods between data anchor points.

There were three data types (logbook, landings, observer data) used to reconstruct total catch data. These data were derived from different national projects. For the periods from 1996 to 2005, we estimated total annual catch from landing data collected previously using a standard method of the Food and Agriculture Organization (RIMF

2005a, RIMF 2005b). In this method, information on the catch per unit effort (CPUE), the boat activity coefficient (BAC, the probability that a boat is active on a given day during a given month) and the active days per fishing fleet (A) was collected monthly using questionnaires (RIMF 1996, FAO 2002). The number of fishing vessels (FS) was based on reconstructed effort data as described in the previous section. The total catch expressed in tons (C) by the fisheries was then estimated by:

$C = FS \cdot CPUE \cdot BAC \cdot A$ (Eq. 3.1).

Ecological groups were grouped based on similarities in their ecological and biological features (e.g., feeding, habitat, mortality), and data availability for reconstruction (Table 3.1). Then total catch by the ecological groups was reconstructed using catch rate data from fisheries surveys during 1996 – 2005 (Appendix 3.2; Table 3.2). In addition, data from observers and logbooks on fishing boats were used to fill in species composition data that were missing in the fisheries surveys (Appendix 3.2; Table 3.2).

Table 3.1. List of species included in total catch reconstruction from 1981 to 2012.

No.	Ecological Groups	Included species/taxa
1	Tuna	*Katsuwonus pelamis*, *Euthynnus affinis*
2	Large predators	Carcharinidae, Scombridae (*Scomberomorus commerson* and *Scomberomorus guttatus*)
3	Large demersal fish	Ariidae, Cepolidae, Cynoglossidae, Drepannidae, Fistularidae, Gobiidae, Holocentridae, Meneidae, Monocanthidae, Nemipteridae, Muraenidae, Ostraciidae, Paralichthidae, Pegasidae, Platycephalidae, Plotosidae, Polynemidae, Priacanthidae, Rhinobathidae, Sciaenidae, Syngnathidae, Synodontidae, Tetraodontidae, Lethrinidae, Serranidae, Scorpaeinidae
4	Other demersal fish	Bothidae, Cynoglossidae, Gerreidae, Haemulidae, Mullidae, Nemipteridae, Presttodidae, Siganidae, Sillaginidae, Soleidae, Sparidae, Teraponidae, Sciaenidae
5	Reef fish	Chaetodontidae, Labridae, Pomacentidae
6	Other pelagic fish	Carangidae (*Atule mate, Alepes kalla, Alepes djedaba, Megalaspis cordyla, Scomberoides spp., Selaroides leptolepis, Seriolina nigrofasciata*, Theraponidae, *Lactarius lactarius* and Selar *crumenopthalmus*, Caesionidae, Scombridae (*Rastrelliger spp.*), *Decapterus maruadsi*, *D. russelli*, *D. kurroides*
7	Medium pelagic fish	Trichiuridae, Stromateidae
8	Small pelagic fish	Clupeidae
9	Anchovy	*Stolephorus commersoni, Encrasicholina heteroloba, E. punctifer, Stolephorusindicus, E. devisi*
10	Cephalopods	Includes squids (*Loligo spp.*), cuttlefish (*Sepia spp.*) and octopus (*Octopus spp.*)
11	Shrimp	Penaeidae, Palaemonidae, Scyllaridae, Soleidae, Solenoceridae, Squillidae
12	Crustaceans	Portunidae, Palinuridae

Another data source was the collection of official reports of local authorities at 28 coastal provinces of Vietnam. However, total monthly catch data reported by the local authorities were only aggregated by commercial groups such as fishes, shrimp, squid, and octopus. These data were broken down into ecological groups in this study. Total catch reconstruction was separated into 12 ecological groups of tuna, large predators, large demersal fish, other demersal fish, other pelagic fish, medium pelagic fish, small pelagic fish, anchovy, cephalopods, shrimps and crustaceans. Annual official landing reports conducted by the General Statistic Department (GSO) were used to compare with the reconstructed catch data in this study. These data of GSO contain the total catch of all fisheries from 1980 to 2012 but information on the gears used and species collected has not been specified.

Table 3.2. Catch rate (%) of ecological groups in total catches of Vietnamese fisheries used to reconstruct their corresponding total catch from 1981 to 2012.

Ecological groups	Fish trawl	Shrimp trawl	Purse seine	Gillnet	Fish handline	Squid handline	Mixed gears
Tuna	0.00	0.00	0.58	7.73	0.00	0.00	0.00
Large predators	0.30	0.00	6.97	10.52	0.00	0.00	0.00
Large demersal fish	51.33	9.39	0.00	0.00	0.00	0.00	0.00
Other demersal fish	31.43	15.30	0.00	0.00	47.23	0.00	0.00
Reef fish	3.94	13.61	0.00	0.00	0.00	0.00	35.14
Other pelagic fish	1.97	0.00	17.41	24.03	23.56	3.98	0.00
Medium pelagic fish	0.79	0.00	23.14	20.45	0.00	5.75	0.00
Small pelagic fish	0.00	0.00	44.09	32.51	25.96	0.00	43.45
Anchovy	0.39	0.00	2.91	0.00	0.00	0.00	0.00
Cephalopods	1.18	0.94	1.16	0.00	0.00	87.36	12.32
Shrimp	0.39	52.67	0.00	0.00	0.00	0.00	0.00
Crustaceans	1.18	2.36	0.00	0.00	0.00	0.00	0.00
Unidentified	7.09	5.73	3.74	4.76	3.25	2.91	9.09

3.2.3. Statistical analysis

3.2.3.1. Missing data

Between 1981 and 1995, in 2006 and in 2008, when data of fishing effort were missing, linear interpolations or extrapolations were made to fill data gaps. These were made based on of the assumption that increases or decreases of the number of vessels by each gear are proportional to the increase or decrease of all vessels.

3.2.3.2. Bootstrapping and smoothing methods

To evaluate the statistically variability of the reconstructed effort and total catch by gears and ecological groups, a bootstrap method (Efron 1979) was used. Bootstrapping is a computer-intensive approach that can provide measures of uncertainty (confidence intervals, standard errors, etc.) for a wide range of problems. It is based on the basic idea of repeated resampling with replacement from an original sample of data in order to create replicate datasets from which inferences can be made on the quantities of interest. When catch and effort data were reconstructed from different sources for different years, the highest and lowest values were used as upper and lower bounds, respectively. Similarly, where data were interpolated, the upper and lower bounds were calculated by linear interpolation between upper and lower neighbouring points. Then the true catch (x) was assumed to randomly fall between the lower (a) and upper range (b) estimates as following:

$$\mathrm{P}[\mathrm{a} \leq \mathrm{x} \leq \mathrm{b}] = \int_{\mathrm{B}}^{\mathrm{A}} \mathrm{f}(\mathrm{x})\mathrm{dx} = 1 \qquad \text{(Eq. 3.2)}$$

With values of x between a and b, the probability density function *f(x)* is given as following:

$$f(x) = \begin{cases} \frac{2(x-a)}{(b-a)(c-a)}, \ if\ a \leq x \leq c \\ \frac{2(b-x)}{(b-a)(b-c)}, \ if\ c \leq x \leq b \end{cases} \qquad \text{(Eq. 3.3)}$$

where c is the mode of the distribution, the peak of the triangle. Here, an asymmetrical triangular distributions was selected because the bounds were probably neither symmetrically nor normally distribution (Vose 2008). Then, the value of c was determined by experts who knew each fishery and its associated data, either as the most likely value of the catch and effort, or percentage of its range between the upper and lower bounds:

$$c = a + \big(d(b-a)\big) \qquad (Eq. 3.4)$$

where d is the percentage range that the mode will vary from the lower bound.

For fitting the curve to better visualize the trend of time-series catch and effort data, the smoothing method was applied. Here, the “loess” locally weighted regression method (Cleveland 1979) was used to fit the model. Smoothers are non-parametric estimators that produce smooth estimates of regression functions. In this study, the smoothing parameter was selected equal to 0.3 (span = 0.3). Means and confidence intervals of 95% obtained from 1000 bootstrap replicas were calculated for the loess smoothing.

3.3. Results

3.3.1. Effort reconstruction

In general, there was a stable trend for all gear types during 1981 to 1985 and then a slight increase trend from 1990 to 1995 (Figure 3.2). Stable trends were again found for all gear combined, fish trawl, purse seine, gillnet, fish handline and mixed gears from 2000 to 2005 (Figure 3.2A, B, D, E, F, H). The number of fishing vessels for all gear types operating in Vietnam waters largely changed between 1981 and 2012 (Figure 3.2). In 1981, there were only about 30,000 units for all gear types operating in Vietnamese waters but this figure increased to nearly 120,000 units in 2012 (Figure 3.2A). Disregarding mixed gear types, the gear types with the highest number of vessels were gillnet and trawl with more 20,000 units enumerated in 2012 (Figure 3.2B & E), whereas shrimp trawl and squid handline contributed less than 10,000 units of total fishing vessels in 2012 (Figure 3.2C & G). The number of trawlers gradually increased from about 6000 in 1981 to more than 23,000 units in 2012 (70% increased, Figure 3.2B) and a same situation found for the gillnet (Figure 3.2E).

3.3.2. Total catch reconstruction

Details on reconstructed catch data can be found in Appendices 3.4 – 3.10. The trend on the reconstructed catch data was similar with total reconstructed fishing vessel. There were stable trends from 1981 to 1985 and from 1995 to 2005 for almost gear types (Figure 3.3A, C, E and F). A slight increase trend was found in the end and early of 1990s. There were sudden increases on total catch of some gear types such as fish trawl, shrimp trawl, purse seine and gillnet between 2006-2008. When considering contribution of gear types on total catch of Vietnamese fisheries, the contribution of gillnet in the total catches of the Vietnamese fisheries was the highest, followed by fish trawling. Total reconstructed catches of gillnet reached around 1.5 million tons (t) in 2012, while this figure was only 0.2 million t in 1981 (Figure 3.3D). Fish trawl was the second largest contributor in the total catch of Vietnamese fisheries with total highest catches of about 0.7 million t in early years of 2010s (Figure 3.3A). The mixed gears also contributed significantly (about 0.3 million t in early 2010s) in total catch of Vietnamese fisheries (Figure 3.3G). Unfortunately, we could not separate more details on the mixed gears due to their complexity and hence data unavailability of these fishing gears. When total catch were grouped by ecological groups, there were stable trends from 1990 to 2005 for total catches of some ecological groups such as large demersal fish, other demersal fish, anchovy, cephalopods and crustacean (Figure 3.4A, B, C, D, E, I, K and M). Slight increase trends were found for other, medium and small pelagic fish, and shrimp (Figure 3.4F, G, H and L).

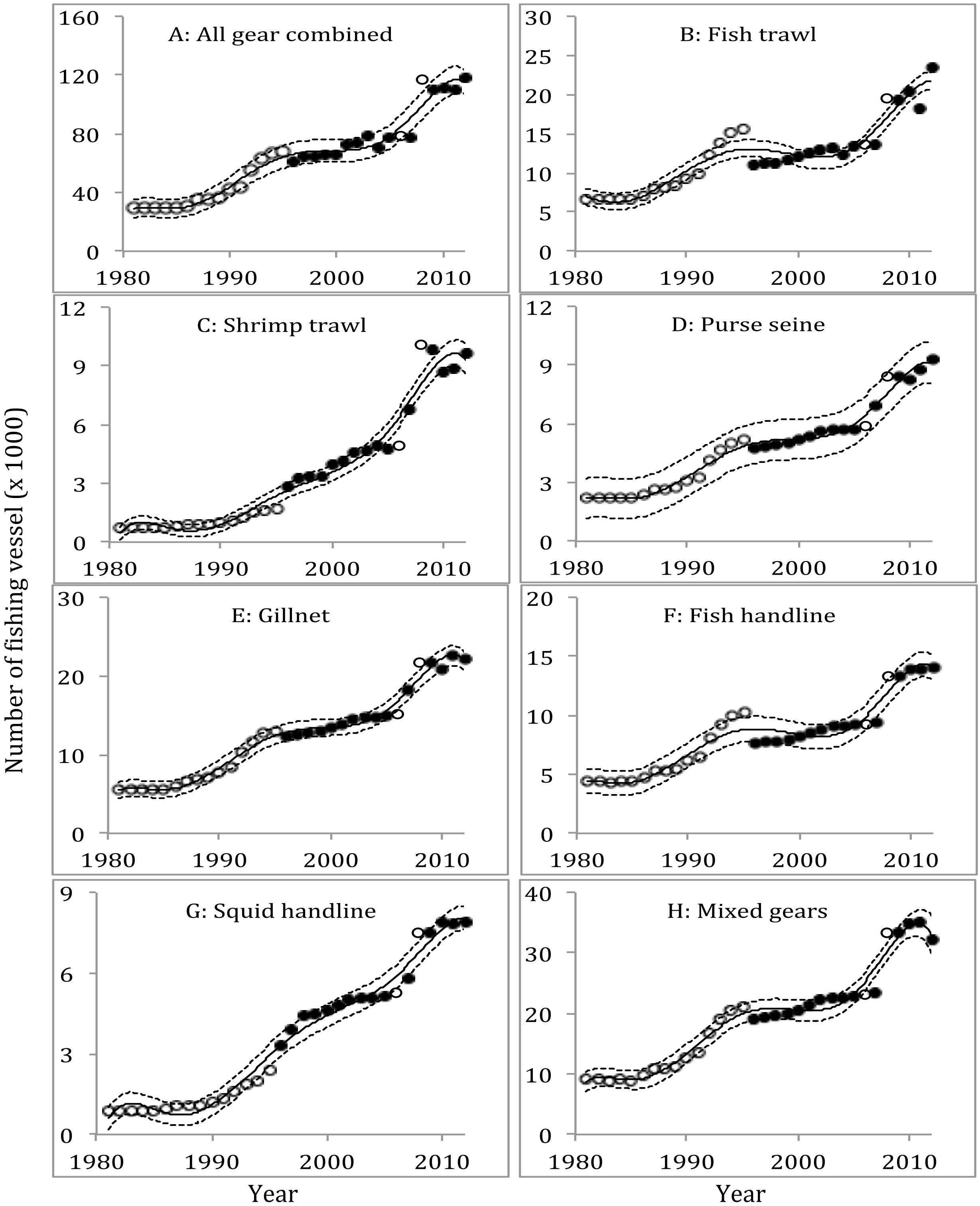

Figure 3.2. Total number of fishing vessels reconstructed for the main fisheries fishing in Vietnamese marine waters from 1981 to 2012. The continuous line represents the fit of number of fishing vessels and the dotted lines indicate the 95% confidence intervals. Locally weighted regression and bootstrapping was used to obtain the smoothing curve and confidence intervals. The open circles indicate extrapolated and interpolated data and solid ones are from collected data.

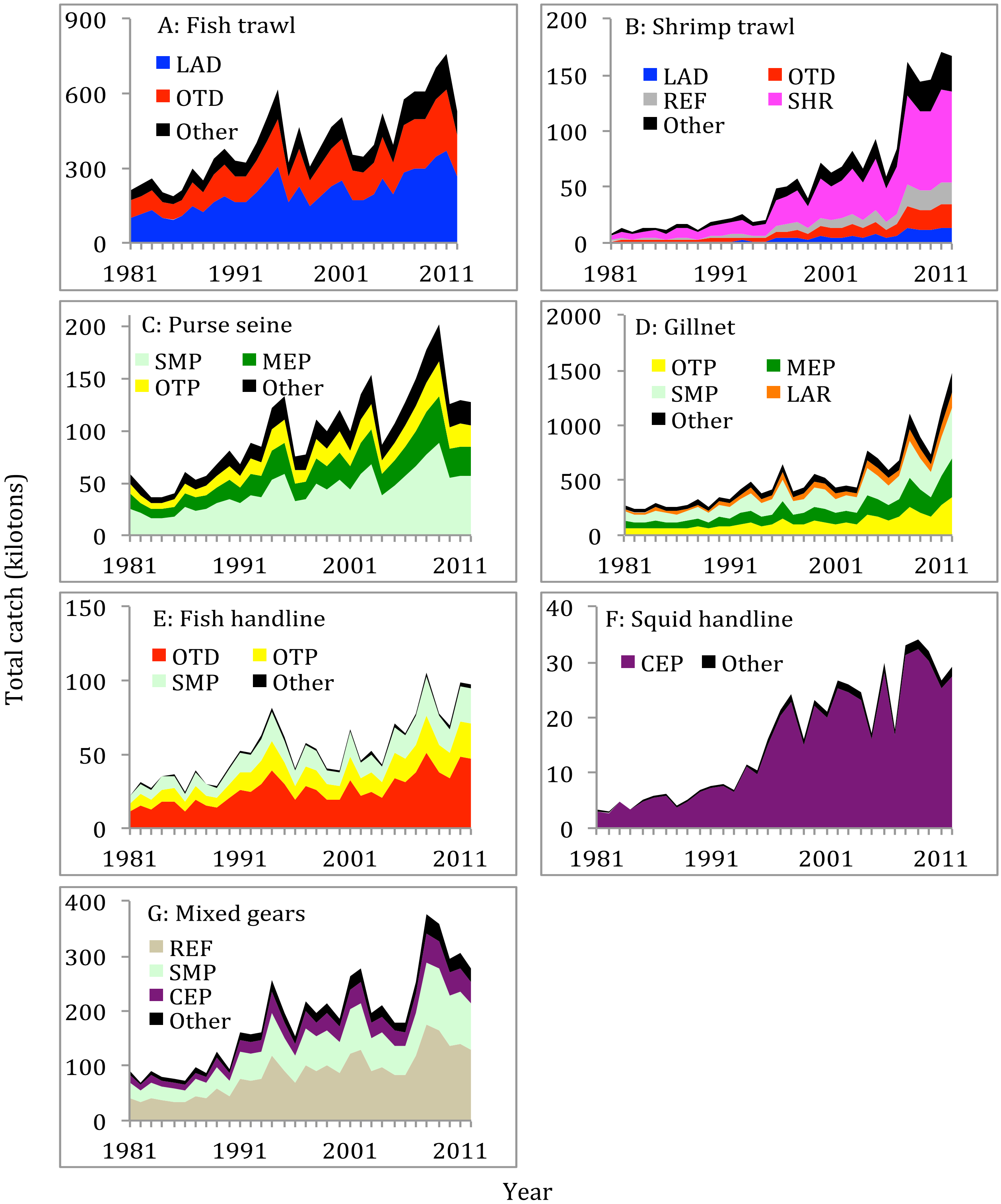

Figure 3.3. Catch composition by main gears of Vietnam from 1981 to 2012. "Other" stands for all groups with catches less than 10% of the total catch for the corresponding gear. LAD=Large demersal fish, OTD=other demersal fish, REF=Reef fish, SHR=Shrimp, SMP=Small pelagic fish, MEP=Medium pelagic fish, OTP=other pelagic fish, CEP-Cephalopods. Note: species composition rates were applied as in Table 3.2 for every year.

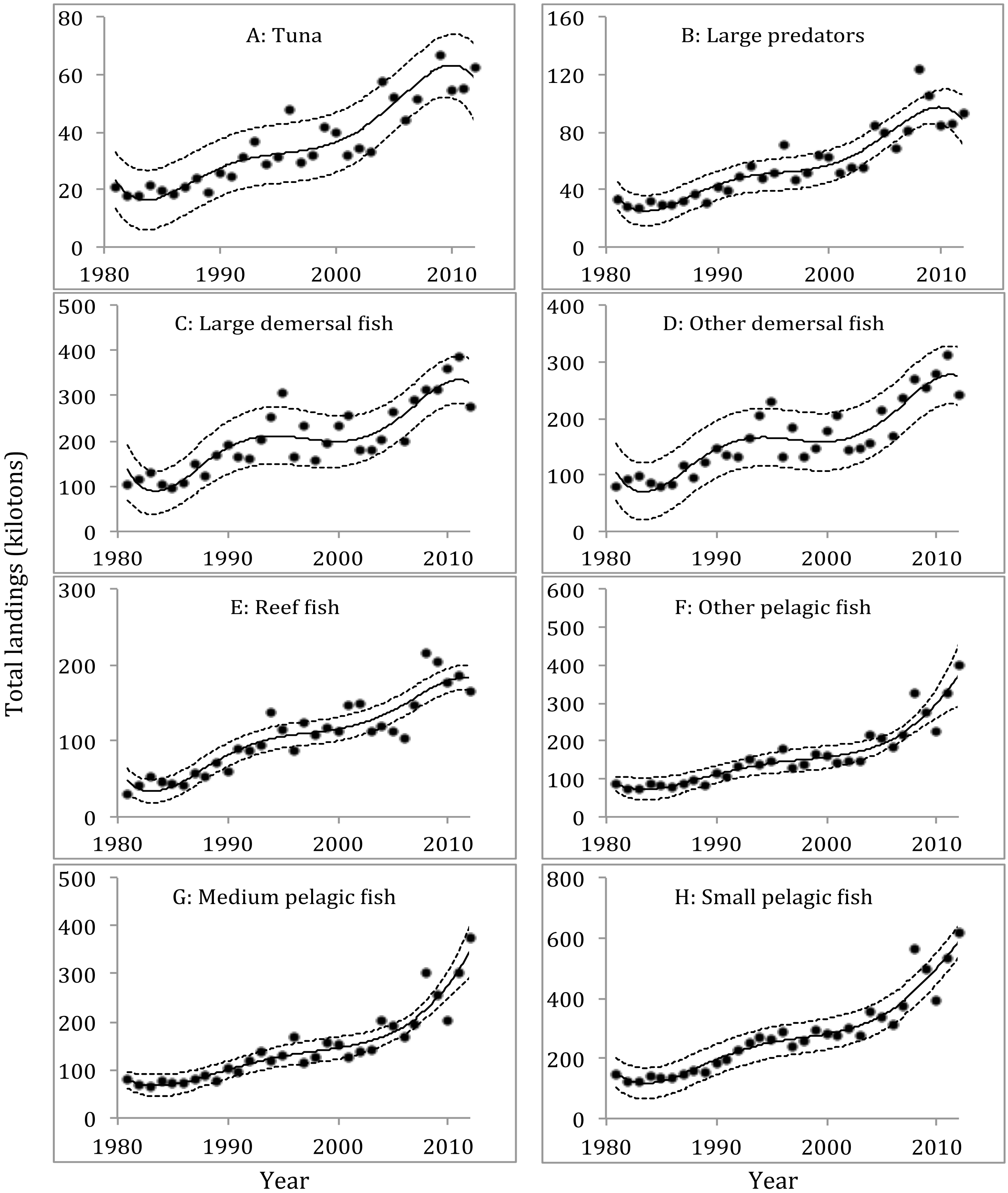

Figure 3.4. Landings of ecological groups caught in Vietnamese waters from 1981 to 2012. The continuous line represents the fit of landings and the dotted lines indicate the 95% confidence intervals. Locally weighted regression and bootstrapping was used to obtain the smoothing curve and confidence intervals. Note: species composition rates were applied as in Table 3.1 for every year.

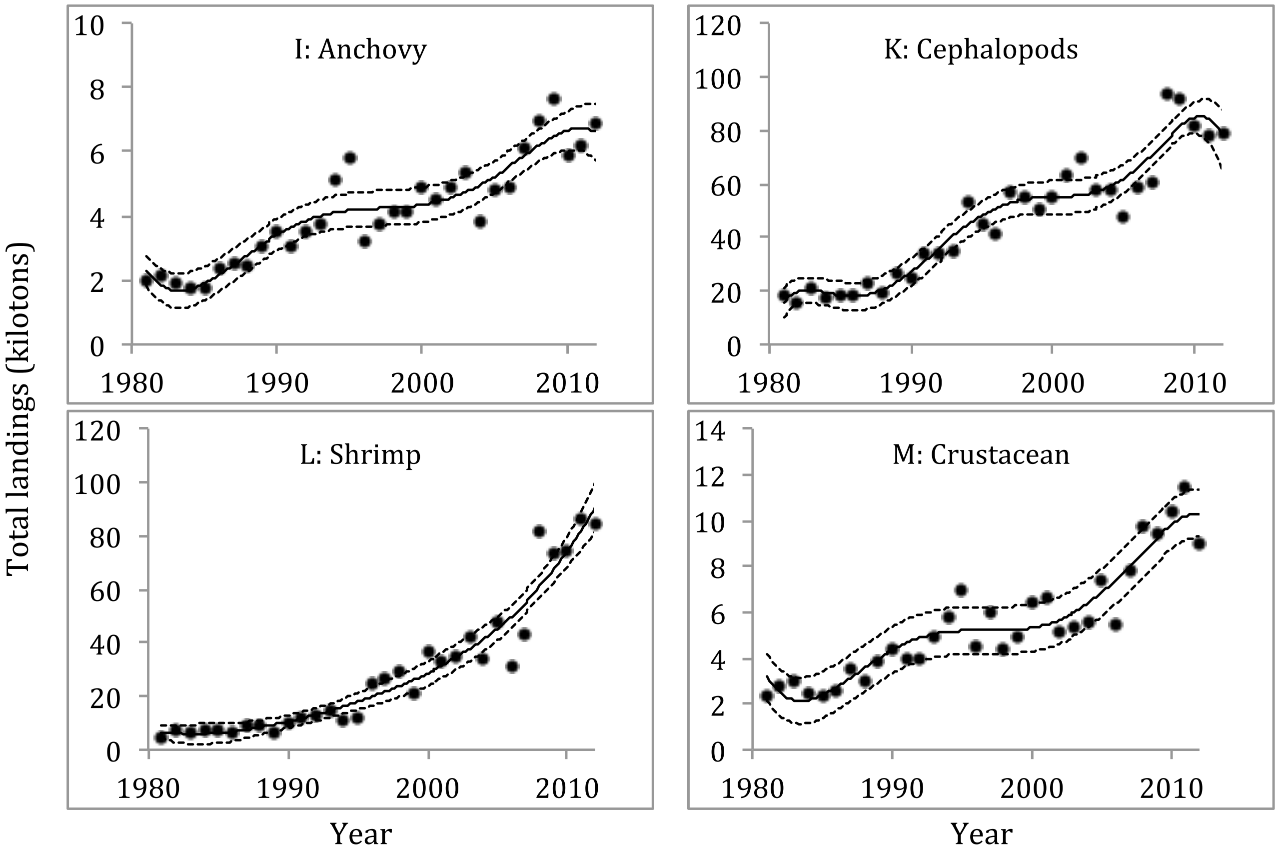

Figure 3.4. (cont.) Landings of ecological groups caught in Vietnamese waters from 1981 to 2012. The continuous line represents the fit of landings and the dotted lines indicate the 95% confidence intervals. Locally weighted regression and bootstrapping was used to obtain the smoothing curve and confidence intervals. Note: species composition rates were applied as in Table 3.1 for every year.

Based on the reconstruction in the present study, a comparison could be performed between the reconstructed total catch and the officially reported data. Both data sources were excluded oceanic tuna fisheries. Interestingly, there was a consistency between the reported official catch statistic and the statistic reconstructed in this study (Figure 3.5). Large differences were only found in some years. For example, in the period from 1991 to 1995, the reported catch statistics were lower than the average values of reconstructed catches (Figure 3.5). However, they are still lying between 95% of the confidence intervals of the reconstructed landings (Figure 3.5).

3.4. Discussion

In this study, we reconstructed effort and catch data for entire Vietnamese EEZ and for all fisheries fishing in inshore, coastal and offshore areas. There were several assumptions used in this study and these could cause potential uncertainties on final results. For instance, we used total number of fishing vessels to reconstruct total catch by gear referring expert knowledge. Therefore, expert knowledge can be impacted by several assumptions that have not taken into account in this study. In addition, sometime the number of registered vessels may not be representative for the real number of fishing vessels. However, these registered vessels were already used to reconstruct total landing in this study with an assumption that the registered vessels are active vessels. In fact, as concluded by Thurstan et al. (2010) landings are a product of fish availability, fishing effort and regulations on catches. Fishing effort is not a measure of the number of fishing vessels alone. Information on improved fishing technologies, and migration to new fishing grounds must simultaneously be accounted for when estimation of catch data (Caddy and Garibaldi 2000, Stergiou 2002, Thurstan et al. 2010). This is because it is possible that landings can keep at high level even when stocks decline. Therefore, the fishing power must be a measure of how fishers increase their catching power over time, for example, by improvements in gear, or their ability to detect fish (for example, higher fishing capacity, lighter nets and electronic fishing gear). These are called as technological creep. In fact, several previous studies have reported changes in vessel fishing power caused by the introduction of global positioning systems (GPS) (Robins et al. 1998, Eigaard et al. 2011). (Robins et al. 1998) found that boats that used a GPS alone had 4% greater fishing power than boats without a GPS. Eigaard et al. (2011) indicated that haddock CPUE increased significantly following the introduction of swivel line and skewed hooks. Therefore, the technological creep as introducing advanced gear technology and on-board equipment causing the increase of fishing capacity should be taken into consideration on future studies.

The number of fishing vessels of almost gears was suddenly increased in 2008 (Figure 3.2B, C, D, E, F, G, H). This can be explained that before 2008 many fishing vessels have not been counted in the national vessel registration system (Tuan 2012). However, in 2008 the Government enhanced the national vessel registration system from local to central levels and provided suitable incentive packages (DECAFIREP 2013). Therefore, there were many fishing vessel to become newly registered in the vessel statistical system. This could cause increases of total catch by fishing gears in 2008. In this study, we found a strong increase in gillnet vessel number of about 70% from 1981 to 2012. Gillnet vessels usually have higher power than fish and squid handline and thus the increase of this fishery will cause stronger impacts on Vietnamese marine ecosystem.

Reconstructed total catch of fish and shrimp trawl, purse seine and gillnet indicated a strong upward trend from 2007 to 2008 (Figure 3.3A, B, C, D). This can be explained by sudden increases of number of fishing vessel in this period. However, this increase on catch data can be affected by artifact errors by using expert knowledge that should carefully be reconsidered in future studies.

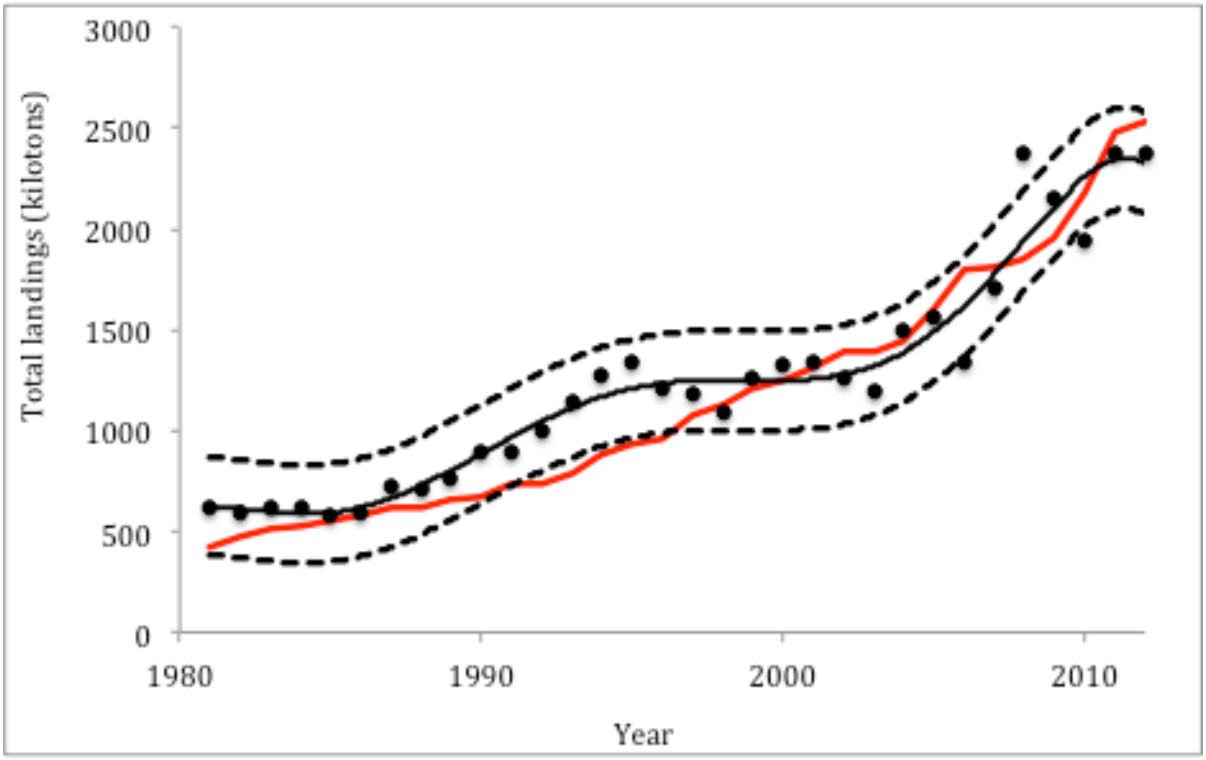

Figure 3.5. Comparison between total reconstructed landings and officially reported catch of Vietnamese fisheries from 1981 to 2012. The solid black line is the total reconstructed catch, the dotted black lines are 95% confidence intervals and the red line is the total officially reported catch (DECAFIREP 2013).

The historical catch reconstruction, presented in this research, is a first attempt for Vietnamese fisheries, taking into account different ecological groups and gear types by time series. Officially reported catch data indicated a continuous upward trend comparing to reconstructed total catch. However, it is also noted that estimation of officially reported catch data was only based on data collected every six months (DECAFIREP, 2013). In this estimation, CPUE and BAC of all fishing gears assumed were unchanged a six-month period. This assumption can cause potential uncertainties in estimated catch. In addition, officially reported catch data were a lumped sum by a total landings for the whole country (Tuan 2012). These are not useful for fisheries management, especially for ecosystem approaches.

Reconstructed catches originating from Vietnamese waters continuously increased over most of the studied period (Figure 3.5). In addition, reconstructed data show that the number of fishing vessels has significantly increased over the past three decades. The number of people relying on marine resources has also been increasing recently. In 1981 there were only 190,000 fishermen in Vietnam but this increased to more than 1,700,000 in the 2010s (DECAFIREP 2013). A combination of continuous increases of fishing vessels and population pressure suggests that overfishing could likely be occurring in the Vietnamese fisheries if they are not well managed. In this case, management measures that integrate all economic, social and ecological are necessary otherwise resources can be exploited in an unsustainable fashion. However, it is hard to apply management measures in developing countries to take people out of work without other alternatives to offer them. In this case, approaches such as community-based fisheries management or co-management can be effective (Beddington et al. 2007). These approaches involve the participation of communities and resource users in decision-making (Armitage and Marschke 2013) by encouraging compliance with regulations, promoting a sense of community ownership over fisheries (Gutierrez et al. 2011) and reducing conflict over scarce resources (Jentoft 2005).

Although there is a level of uncertainty associated with estimates in this study, the reconstructed data in this study can provide the information needed for investigating fishing pressure on fish communities and contribute to an ecosystem approach to fisheries management.

Chapter 4: Ecological indicators to assess impacts of Vietnamese fisheries

Redrafted from

Anh, P. V., F. De Laender, G. Everaert, C. T. Vinh and P. Goethals (manuscript in preparation). Ecological indicators to assess impacts of Vietnamese fisheries.

4. Chapter 4: Fishery-based indicators to assess impacts of Vietnamese fisheries

Abstract

The impacts of fishing on trophic structure of ecological groups were assessed using fishery-based indicators. In this study, we use a set of fishery-based indicators (i.e. mean trophic level of landing (MTL), the fishing in balance index (FiB), and the pelagic/demersal (P/D) ratio. We used these indicators to assess fishing impacts between 1981 and 2012 on twelve ecological groups. We found a slight decline in MTL of 0.01 trophic level per decade. FiB increased from 1985-1995 and 2005-2008 that is in line with fisheries development strategies of the Vietnam Government to expand further fishing grounds at that time. The P/D index varied from 1.3 to 1.4 suggesting that local fishers are targeting pelagic species fisheries using gillnets and purse seine. The use of fishery-based indicators in the evaluation of ecosystem offers an alternative to complex models requiring a huge amount of data that is very useful in the Vietnamese fisheries. However, the artifact problems on indicator calculation should carefully be taken into account in outcome results of future studies.

4.1. Introduction

Human activities result in contamination, habitat degradation, and eutrophication (Navia et al. 2012). Particularly, fishing has been identified as causing irreversible structural and functional changes in marine ecosystems (Estes et al. 2011, Lotze et al. 2011). The responses of aquatic ecosystems to these impacts are the result of complicated interactions and feedback mechanisms (Angelini and Moloney 2007). Thus, holistic approaches are increasingly needed to understand and manage these impacts. An ecosystem approach to fisheries management (EAFM) can be a potential approach because it can lead to more relevant and reliable objectives in management by taking account of the complexity between and among ecosystem components (FAO 2003). The goal of EAFM is to sustain healthy, productive and resilient condition of the ecosystem to continuously provide services and goods for human being (Link 2002). Understanding ecological interactions is a key point for an EAFM (Cury and Christensen 2005). The power of ecological processes such as trophic interactions has been defined to be extremely important in fish population dynamic (Bax 1998). To evaluate ecosystem complexity and then to implement such suitable approaches, tools must be developed to facilitate communication between managers, scientists and stakeholders. In this case, fishery-based indicators can be selected. The fishery-based indicators can reflect and describe the complex interactions between fisheries and the marine ecosystem (Pauly and Watson 2005). In addition, the fishery-based indicators can be used to monitor trends in ecosystem conditions over time and to provide an early warning signal of changes in the ecosystem. They can be used to detect spatiotemporal differences in the integrity of ecosystems (Babouri et al. 2014). In particular, the fishery-based indicators can describe the stages of exploitation of resources and analyse the state of the ecosystem relative to past periods for which there are no other data available except the amount of landings (Pauly and Watson 2005).

Pauly et al. (1998) observed a gradual transition in landings from long-lived, high trophic level, piscivorous bottom fish towards short-lived, low trophic level invertebrates and planktivorous pelagic fish. This phenomenon, known as fishing down the marine food webs (Pauly et al. 1998), occurs as species of greater size with long life cycles are most susceptible to collapse. Once these stocks become depleted, exploitation is directed toward smaller sized species with a faster growth rate, and thus the mean trophic level (MTL) of landings decreases (Pauly et al. 1998, Stergiou and Karpouzi 2001). However, Branch et al. (2010) demonstrated the potential danger of basing conclusions on a single indicator to assess the ecological health of an ecosystem. Therefore, using the MTL alone may be insufficient to evaluate the general status of marine ecosystems. Apart from the MTL, other indicators have also been developed which can be integrated to evaluate the state of ecosystems exploited by fishing. These include the “fishing-in-balance” (FiB) index developed by Pauly et al. (2000) to analyse the expansion and contraction of fisheries. These indicators, along with the ratio of pelagic and demersal fish landings (P/D) (Caddy 2000) can be calculated from data on commercial fisheries landings. Therefore, this type of indicators is well suited for studying the impact of fisheries on the marine ecosystem under data-poor circumstances such as in developing countries.

Landings of Vietnam's marine fisheries have increased rapidly since the 1990s. The total estimated fisheries catch was only about 700,000 tons in 1990 but reached more than 2 million tons in 2012 (Anh et al. 2014a). Average catch per horsepower (HP) estimated in the 1980s was around 1.1 tons·HP^{-1} and this number was reduced to about 0.35 tons·HP^{-1} in recent years. It is thus necessary to test if shifts in the trophic structure of the Vietnamese marine ecosystem with only emphasizing on fish communities occurred.

Implementation of EAFM needs to develop indicators to provide information on the state of the ecosystem, the extent and intensity of fishing effort and the progress of management in relation to objectives (Jennings 2005). There are many indicators to support EAFM but they are usually classified into three main types of ecological, social and economic indicators (Jennings 2005). However, in this study we only use fishery-based indicators calculated from landing data (i.e. mean trophic level of catch, fishing in balance and pelagic/demersal ratio) of inshore, coastal and offshore fisheries to assess changes on trophic structure of some ecological groups (only fish communities) in data-poor situation. Selection of these indicators is due to data availability in a data-poor situation of Vietnam.

4.2. Material and methods

4.2.1. Database

4.2.1.1. Catch data

We used reconstructed catch and effort data (1981-2012) of inshore, coastal and offshore fisheries to calculate ecological indicators for the Vietnamese marine ecosystem (cf. Chapter 3). In this study, catch data were grouped in 12 ecological groups (Table 4.1), based on similarities in their ecological and biological features (e.g., feeding, habitat, mortality), and the importance of the species for fisheries. Note that categories that are representing "unidentified species" as indicated in the Chapter 3 were not included in this analysis due to impossibility of assigning them a precise trophic level. In this study, because there is no information on discard on the Vietnamese fisheries, we assumed that all catches were landed and hence catches are equal to landings. Details of catch by group are indicated in the Appendix 4.1.

4.2.1.2. Trophic level

In this study, the trophic levels (TLs) were assigned according to Fishbase (Froese and Pauly 2006) for fishes and in SeaLifeBase (Palomares and Pauly 2013) for cephalopods, shrimp and other crustaceans. Mean TL was calculated for each ecological group. If more than one TL estimate was available for any species (or of species group), the mean of all available values was used. A general result was indicated in the Table 4.1.

Table 4.1. The trophic levels (TL) for ecological groups landed between 1981 and 2012 in Vietnam and used to calculate fishery-based indicators.

No	Ecological groups	TL
1	Large predators	4.20
2	Tuna	4.10
3	Medium pelagic fishes	3.50
4	Small pelagic fishes	3.33
5	Other pelagic fishes	3.45
6	Anchovy	3.10
7	Cephalopods	3.77
8	Large demersal fishes	4.15
10	Other demersal fishes	3.85
9	Reef fish	3.45
11	Shrimp	3.21
12	Crustaceans	3.30

4.2.2. Fishery-based indicators

Ecological indicators quantify the magnitude of stress, degree of exposure to the stress, or the degree of ecological response to the exposure. These indicators are intended to provide a simple and efficient method to examine the ecological composition, structure, and function of complex ecosystems (Dale and Beyeler 2001). In order to be useful, ecological indicators should be sensitive to changes in ecosystem integrity through space and time, easily measured, understandable, informative and based on accessible data.

Following these criteria, we have selected different fishery-based indicators that are relevant to assess fisheries' impacts on fish communities. Some priorities are summarized as in Appendix 4.2.

4.2.2.1. Mean trophic level

The mean trophic level of landing (MTL) was proposed at the Conference of the Parties to the Convention on Biological Diversity (CBD) as one of the indicators to recognize changes of biodiversity (CBD 2004). The index was developed based on the assumption that the decline on mean trophic level of fisheries catches is generally caused due to a fishery-induced reduction of biomass and biodiversity of vulnerable predators (Pauly et al. 1998). The MTL is calculated from a combination of fisheries landings and diet composition data for the landed fish species. It is computed, for each year as:

$$MTL_k = \frac{\sum(TL_j)(Y_{jk})}{\sum Y_{jk}} \quad \text{(Eq. 4.1)}$$

where MTL_k is the mean trophic level of all landings in year k, Y_{jk} refers to the catch (landings) of species j in year k and TL_j is the trophic level of species j. Changes in the landings of high trophic level species could be predicted by changes in this index (Pauly et al. 1998). Indeed, fishing often targets the highest TL species, allowing their prey to expand in number, in turn leading to excessive grazing of the latter on their prey. These trophic cascades, in extreme circumstances, can be disastrous for marine ecosystem (Daskalov 2002).

4.2.2.2. Fishing in balance

In marine ecosystems, the average efficiency of energy transfer assumed is 10% between trophic levels (Pauly and Christensen 1995) as ca. 90% of the energy is consumed for maintenance, reproduction and other activities of animals in the systems (Pauly and Christensen 1995). To study this energy transfer efficiency (TE) Pauly et al. (2000) introduced an index called the fishing in balance (FiB) index:

$$FiB_k = \log[\,Y_k \cdot \left(\frac{1}{TE}\right)^{MTL_k}] - \log\,[Y_o \cdot \left(\frac{1}{TE}\right)^{MTL_o}] \quad \text{(Eq. 4.2)}$$

where Y is landings in year k, MTL is the mean TL of the landings in year k, TE is the transfer efficiency (here set at 0.1 following Pauly et al. (2000). A value of year k = 0 refers to a baseline to normalize the index (Pauly et al. 2000) and of 1981 in this study.

The FiB index makes the relation between increases in landings due to focusing on lower TLs and the transfer efficiencies between TL's and verifies whether it corresponds to the ecological appropriate increase. The FiB index will stay constant (FiB = 0) if fishery is balanced. It means that all trophic level changes are matched by ecological equivalent changes in catch. When the FiB index increases (FiB > 0), there may be two possibilities: (1) occurring bottom-up effects with increase in low trophic level groups (Caddy et al. 1998) and/or (2) geographic expansion of the fishery to new areas which in effect expands the ecosystem exploited by the fishery (Pauly and Watson 2005). A decline in FiB (FiB < 0) will be observed if discarding took place that is not represented in the catch, or if the ecosystem functioning is impaired by the removal of excessive levels of biomass (Pauly and Watson 2005). An assumption required in calculation of the FiB index is that transfer efficiency is constant (and known sufficiently well) across trophic levels (Pauly and Watson 2005). The FiB index is supposed to provide a better indicator of ecosystem change that catch or catch composition due to its integrative nature (Garcia and Staples 2000).

4.2.2.3. Pelagic/demersal ratio

Changes in the trophic composition of marine communities can be tested in terms of large trophic groups such as planktivorous, benthivorous, or piscivorous animals (Caddy 2000). The expected effect of fishing (although not exclusive) is a decrease in the proportion of piscivorous fish. This is an easily understood indicator that can be estimated based on the biology of the species present in the community. In this context, an index that has been proposed is the ratio of pelagic species landing (P) to demersal species landing (D) (P/D) in weight (Caddy 1993, Caddy 2000). A decrease of this ratio indicates that resources are more abundant and domination of pelagic species and vice versa. This can be caused by both the eutrophication effect favouring the development of pelagic groups and high fishing pressure on demersal groups (Caddy 1993, Caddy 2000). In fact, pelagic species are positively influenced by the increase in nutrients that stimulate primary production (Caddy 1993), whereas demersal species are negatively affected by the hypoxia arising from the excess of primary production.

The P/D ratio is also a suitable indicator for studying the overall evolution of the fisheries. In fact, a high demand for demersal species and an increase in the P/D ratio can be explained by an over-exploitation of demersal species (Pennino and Bellido 2012). In addition, similar as other catch-based indicators, its sensitivity to the evolution of target species and fishing methods is high (Pennino and Bellido 2012). Consequently, this ratio is used to compare with landings of certain species groups that are important to the fisheries and to the evolution of the fishing fleets.

4.2.3. Statistical analysis

To better visualize the trends in the MTI, FiB and P/D ratio throughout the studied period, both smoothing method and linear regression models were performed. For relationships between the MTI, FiB, P/D ratio with total landings, we only used a linear regression model. For the smoothing method, we applied the “loess” locally weighted regression method (Cleveland 1979) to fit the model. Smoothers are non-parametric estimators that produce smooth estimates of regression functions. In this study, the smoothing factor was selected equal to 0.3 (span = 0.3).

For the linear regression model, we used a Spearman's rank correlation (Spearman 1904) to test if the linear regression models are statistically significant.

4.3. Results

4.3.1. Ecological indicators

There was large variation on the mean trophic level (MTL) of the Vietnamese fisheries within a year. However, when the MTL compared in the studied period, there was a slight downward trend of 0.03 trophic level between 1981 and 2012 (Figure 4.1A). It seemed that the MTL was divided into two groups (i.e. 1981 to 1995 and 1996 to 2012). Of those, the trend of the first period was relatively higher than the later one (Figure 4.1A). However, this downward trend was unclear if caused by fishing or by data variability.

The fishing in balance index (FiB) showed negative values in early years of the series (Figure 4.1B). After that, there were stable trends between 1995 to 2005 and 2009 to 2012. The FiB showed increase trends between 1985 to 1995 and 2005 to 2008 (Figure 4.1B).

The pelagic/demersal (P/D) ratio was always higher than 1 for all studied time series (Figure 4.1C). By fitting linear regression model, the P/D ratio indicated an unchanged trend during the studied period and slightly varied from 1.3 to 1.4 (Figure 4.1C).

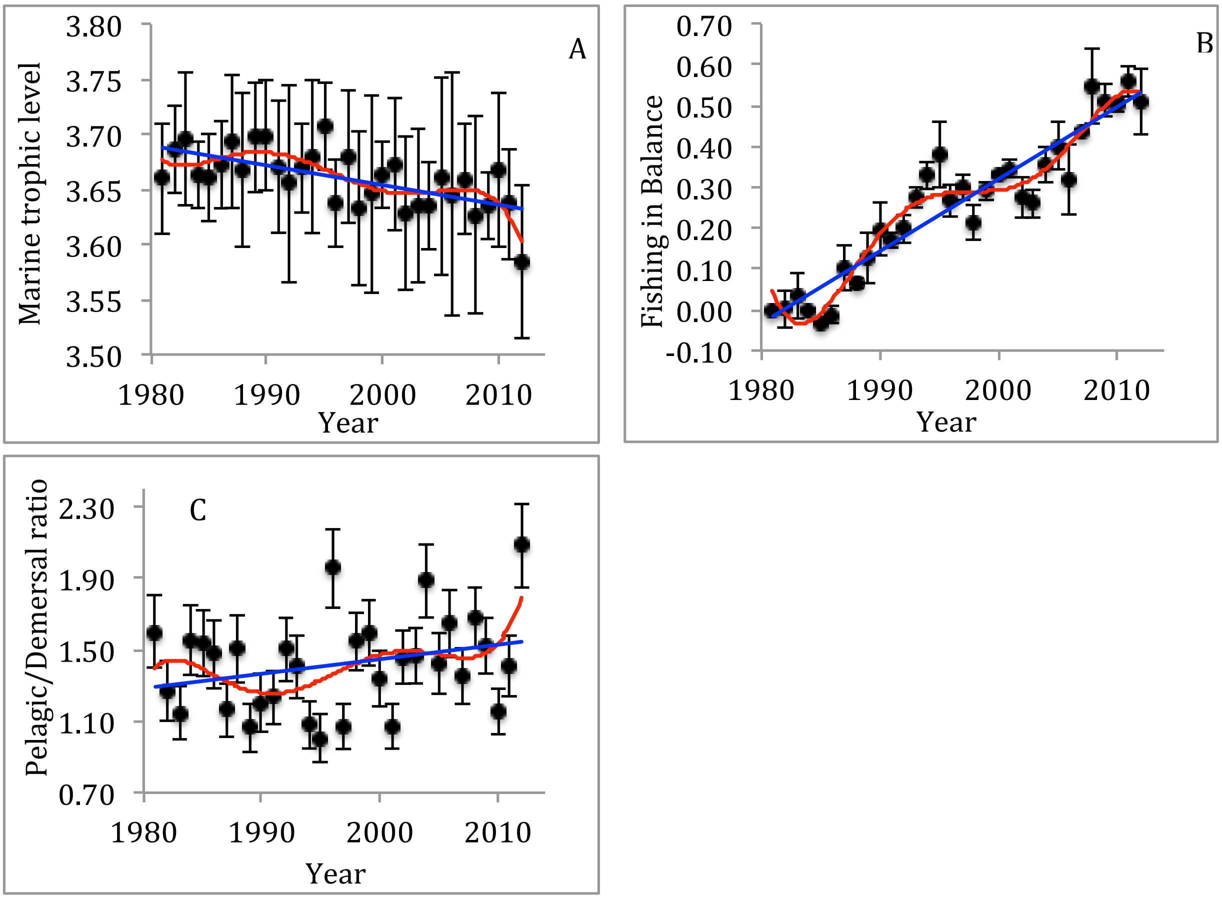

Figure 4.1. Trend of fishery-based indicators of Vietnamese fisheries from 1981 to 2012. A: Marine trophic index, B: Fishing in Balance and C: pelagic/demersal ratio. Means and 95% confidence intervals were shown. The red lines are fitted by smoothing methods and blue lines indicate linear regression models.

4.3.2. Relationship between fisheries landings and indicators

The MTI of fisheries catches was negatively correlated with total landings ($r = -0.61$, $p < 0.05$, Figure 4.2A & Appendix 4.3). In contrast, a positive correlation was revealed between the fishing in balance index and total landings in the Vietnamese fisheries from 1981 to 2012 ($r = 0.95$, $p<0.05$, Figure 4.2B & Appendix 4.3). However, there was no significant relationship found between pelagic/demersal ratio and total landings ($r = 0.08$, $p>0.05$, Figure 4.2C & Appendix 4.3).

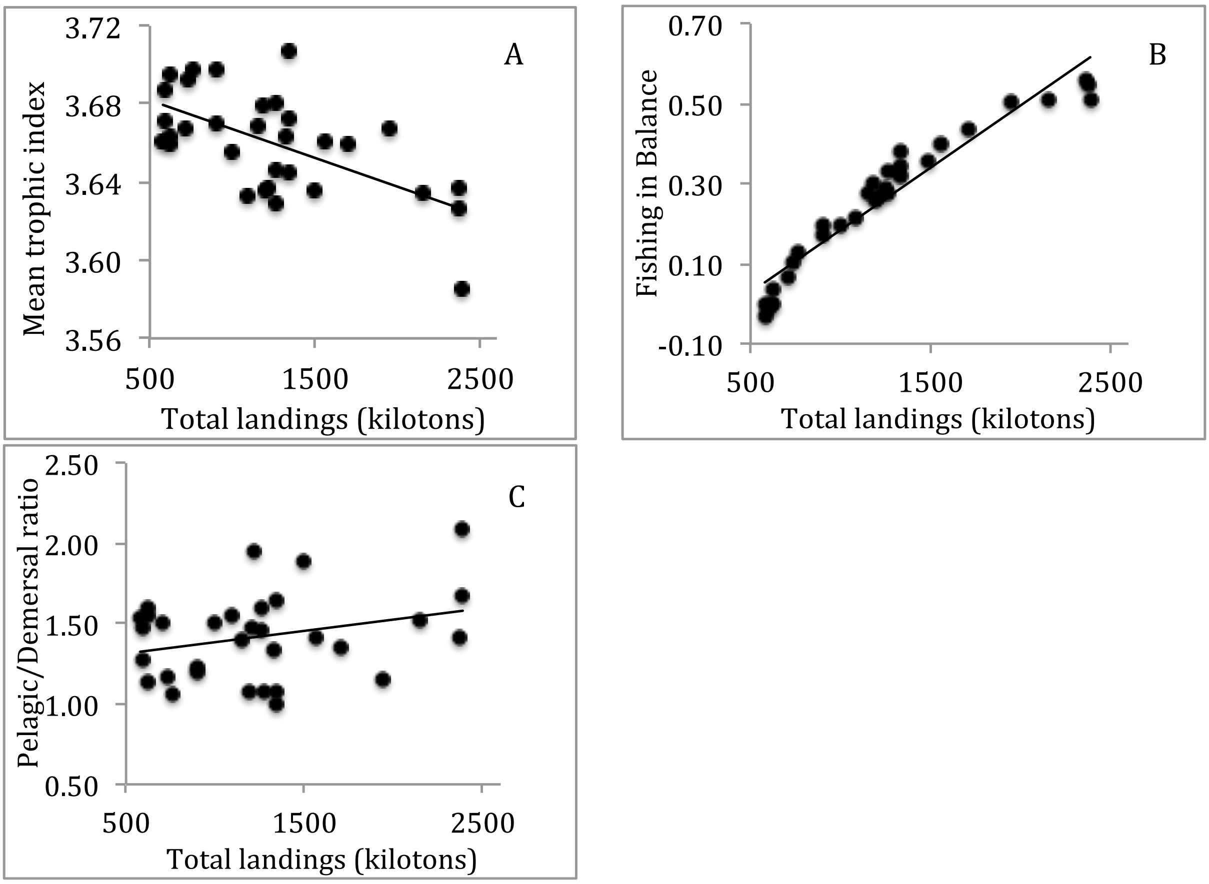

Figure 4.2. Relationship between total landings and the mean trophic index (A), Fishing in Balance (B) and Pelagic/Demersal ratio (C) of Vietnamese fisheries from 1981 to 2012.

4.4. Discussion

In this study, the mean trophic level (MTL) was divided into two groups (i.e. 1981 to 1995 and 1996 to 2012) with the trend of the first period of relatively higher than the later one. This was in agreement with the increase trends of catch. The MTL decline identified in this study was very low with only 0.03 trophic level in the studied period (0.01 per decade) compared to other studies worldwide (Table 4.2). There are some potential limitations in our analysis. One of the problems in the calculation of ecological indicators in this study was over-aggregated catch statistic. In fact, the number of ecological groups included in this study is small when compared to similar published analyses (e.g. Pauly et al., 1998; Bhathal and Pauly 2008). As indicated by (Pauly and Watson 2005), when the detailed fisheries statistics used to calculated marine trophic index, a declined rate of this indicator (a indication of fishing down the marine food web) was clearly found. This could indicate that over-aggregation of species composition can mask the trend of fishing down the marine food web and it should thus disappear when the taxonomic information in a dataset is lost (Pauly et al. 2001).

Table 4.2. Declines of trophic level in other regions of the world. Adapted from Freire and Pauly (2010).

Country/region/area	Studied period	TL/decade	Source
Thailand Gulf	1965-1997	0.05 - 0.09	Christensen (1998)
Chines EEZ	1970-1998	0.20	Pang and Pauly (2001)
Northern Gulf of Mexico	1950-2000	0.02	Pauly and Palomares (2005)
Eastern Canada	1950-1997	0.10	Pauly et al. (2001)
Indian	1950-2000	0.01 - 0.08	Bhathal and Pauly (2008)
East coast, USA	1950-2000	0.04	Chuenpagdee et al. (2006)
Vietnamese marine ecosystem	1981-2012	0.01	This study

Disaggregation and aggregation of landings data by areas and species can also affect estimations of fishery-based indicators (Freire and Pauly 2010, Moutopoulos et al. 2014). Moutopoulos et al. (2014) demonstrated that Greek's fisheries are fishing up the marine food web with when landings data were aggregated by a large area. However, a downward trend was found with disaggregated landings data by subareas. Effects of landings data aggregation were also shown in studies of the Brazilian marine food webs (Freire and Pauly 2010). It was indicated that once the disaggregation of landings data was performed, trophic levels showed opposite trends comparing with aggregated landings data. In the present study, landings data were aggregated for entire areas of Vietnam. Use of disaggregated detail data by areas was not possible due to lack of information for disaggregation of landings data by areas. In addition, in this study catch rate assumed were the same in every year of the studied periods. This assumption can affect the estimation of ecological indicators based on landing data.

In addition, we also consider that the estimation of MTL of the catch could be the factor that incorporates the highest uncertainty into our analysis. MTL values were estimated using individual trophic levels of each ecological group and these individual trophic levels were referred from the literature. This can be a problematic since diet composition, and consequently in trophic level, can be changed by studied areas (Caddy et al. 1998).

In this study, to calculate the fishery-based indicators we used reconstructed catch data for a long time series (1981-2012). There may be artifacts of our pre-processing of the catch data as discussed in the Chapter 3 that can affect the present results. Finally, the future inclusion of unidentified species may change the results of this analysis as they reach high proportions in Vietnamese fisheries catch (Chapter 3, Appendix 3.4 – 10). In addition, we used an assumption that there was no discard on the Vietnamese fisheries. This assumption can be a problem because discard on fisheries can also have damaging ecological effects (Catchpole et al. 2008) by (i) the loss of future yields incurred through the discard mortality of commercial species and (ii) the ecological impacts caused by discarding target and non-target species. These have not taken into account when calculate fishery-based indicators.

Phenomenon that fishing cause gradual transition in landings from high trophic level species towards low trophic level species is called "fishing down the food web" (Pauly et al. (1998). Unfortunately, the present results did not confirm very clearly this phenomenon. This is in agreement with conclusions of Harris and Poiner (1991) and Navia et al. (2012). These studies concluded that ecosystems in tropical latitudes (i.e. in this study area) seem to be a little more resistant than temperate ecosystems. In other words, trophic networks in temperate ecosystems have a higher likelihood of being impacted by fishing effects and cause phase shifts with little probability of return than those in tropical ecosystems.

Nevertheless, it is necessary to take all above mentioned issues into account in future studies to reduce potential uncertainties when estimate fishery-based indicators.

Although, the "fishing down the food web" was often referred to in literature (Pauly et al. 1998, Pauly et al. 2001), it was also criticized in some studies (Caddy et al. 1998, Myers and Worm 2003). Especially the fact that the downward MTL trend can be masked due to geographic expansion of the fisheries of a given region or country has occurred, which enables them to maintain or even augment their catch of high-trophic level species (Kleisner and Pauly 2011). To compensate the conclusions about the MTL in this study, the fishing in balance index was also used. This index was developed to address what may occur when the decline in the MTI is attributable to the deliberate choice of targeting low trophic level species. FiB stayed constant between 1995 to 2005 and 2009 to 2012 that shows that all trophic level changes of catch were matched by ecological equivalent changes in catch (Pauly et al. 2000). FiB increased from 1985 to 1995 and 2005 to 2008. This is in line with fisheries development strategies of Vietnam by the Government at that time (Tuan 2012). In fact, with innovations to develop offshore fisheries (Tuan 2012), Vietnamese fisheries have been shifting from coastal resource-targeted fisheries into offshore fisheries and fishing technologies are advanced. The modernisation of small- and large-scale fishing fleets (i.e., larger boats, of higher tonnage and engine horsepower, improved fishing gears, use of high-technology equipment, etc.) led to the expansion of fishing in open sea areas, previously largely inaccessible by fishing vessels because of unfavourable natural conditions (e.g. strong winds) and in deep water areas. As a result, new 'resources' started to be exploited, mostly at high trophic levels. These can cause changes in the marine food web and can alter due to a change in impacts as a result of the advances in fishing technologies (Caddy and Garibaldi 2000).

The P/D index was not much changed and varied from 1.3 to 1.4 in the studied period. This can verify long-term fishing tradition of the Vietnamese fishers focusing on pelagic species fisheries such as gillnet, purse seine and other traditional fishing gears without changing targeting species. However, targeting on same ecological groups in the long term can cause harmful effects on structure of these groups and thus new fishing strategies should be considered by policy makers. The P/D index in fisheries catches might be an indicator of eutrophication rather than exploitation (de Leiva Moreno et al. 2000). The pelagic fish are positively influenced by nutrient enrichment when it stimulates the plankton production (Caddy 1993), while the demersal fish are influenced by the dynamics of benthic community, which generally responds negatively to the conditions of excessive enrichment. It follows that a positive trend over time in the P/D index may depend both on the eutrophication both from the overexploitation of resources (Libralato et al. 2004). In addition, like other fishery-based indicators, it will be sensitive to changes in the fishing targets and methods.

The use of fishery-based indicators in the evaluation of ecosystem caused by fishing offers an alternative to complex models requiring a huge amount of data that are not always available. This is particularly the case of Vietnamese fisheries for which data are often scared. Consequently, we would like to encourage other studies of this kind together with outcomes from modelling to evaluate all fisheries to obtain the best possible results.

Chapter 5: Inverse modelling of trophic flows throughout an entire ecosystem to support sustainable coastal fisheries management in Vietnam

Redrafted from

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5. Chapter 5: Inverse modelling of trophic flows throughout an entire ecosystem to support sustainable coastal fisheries management in Vietnam

Abstract

Fishing effort in the Vietnamese coastal ecosystem has rapidly increased from the 1990s to 2000s, with unknown consequences for local ecosystem structure and functioning. Using ecosystem models that integrate fisheries and food webs we found differences in the production of six functional groups, the food web efficiency, and eight functional food web indices between the 1990s (low fishing intensity) and the 2000s (high fishing intensity). The functional attributes (e.g. consumption) of high trophic levels (e.g. predators) were lower in the 2000s than in the 1990s while primary production stayed invariant, causing food web efficiency to decrease up to 40% with time for these groups. The opposite was found for lower trophic levels (e.g. zooplankton): the functional attributes and food web efficiency increased with time (22 and 10% for the functional attributes and food web efficiency, respectively). Total system throughput, a functional food web index, was about 10% higher in the 1990s than in the 2000s, indicating a reduction of the system's size and activity with time. The network analyses further indicated that the Vietnamese coastal ecosystem in the 1990s was more developed (higher ascendancy and capacity), more stable (higher overhead) and more mature (higher ratio of ascendancy and capacity) than in the 2000s. In the 1990s the recovery time of the ecosystem was shorter than in 2000s, as indicated by a higher Finn's cycling index in the 1990s (7.8 and 6.5% in 1990s and 2000s, respectively). Overall, our results demonstrate that the Vietnamese coastal ecosystem has experienced changes between the 1990s and 2000s, and stress the need for a closer inspection of the ecological impact of fishing. However, it is not very clear to what extent the changes between two studied periods were only caused by fishing or other causes. Thus, future studies should integrate a variety of disturbance sources into more representative models to quantify the relative contribution of several potential drivers to changes in the structure and functioning of the Vietnamese coastal ecosystem.

5.1. Introduction

At the end of last century, fisheries management used single-species stock assessment methods to quantify fish stocks (Caddy and Cochrane 2001). Unfortunately, this approach had – from an ecological perspective – strong limitations and shortcomings as ecosystems are composed of multiple species and often multi-gear fisheries are used and effects at the ecosystem level are too often unknown (Coll et al. 2006, Griffiths et al. 2010). Ecosystem models have been developed that integrate fisheries, whole biological communities and the interactions between them to study ecosystem-wide fisheries impacts (Diaz-Uribe et al. 2007). These models have revealed how internal and external factors could affect marine ecosystem functioning (Brigolin et al. 2011).

Landings of Vietnam's marine fisheries have increased rapidly since the 1990s. Total estimated fisheries catches of Vietnam were 700,000 tons in 1990 but reached more than 2 million tons in 2012 (Anh et al. 2014a). During the past years, concern has been raised about the sustainability of these intensive practices. According to some recent assessments, catches have far exceeded the maximum sustainable yield in the coastal waters of Vietnam (Pomeroy et al. 2009, Anh et al. 2014a). The average catch per horse power (HP) estimated in the 1990s was around 0.6 tons·HP^{-1} and this number was reduced to about 0.35 tons·HP^{-1} in recent years (Anh et al. 2014a). Increasing activity of small fishing boats in the Vietnamese coastal areas has been suggested as a possible cause for the observed depletion of the coastal marine resources (Pomeroy et al. 2009, Armitage and Marschke 2013, Anh et al. 2014a). In the end of 1990s, the number of fishing vessel was only around 70,000 but it reached about 85,000 in 2005 (Anh et al. 2014a). In addition, the number of people depending on the Vietnamese marine resources also increased with 30% between the 1990s to the 2000s (Anh et al. 2014a). Increasingly anthropogenic activities exert great influence on the Vietnam's coastal marine ecosystem. Therefore, it is quite important to study the development level of this ecosystem and its state of maturity, which facilitates profound understanding of the structure and function of the whole ecosystem for analysing the impact of human influences.

In the present paper we studied the effect of fishing intensities on the functioning of Vietnamese coastal ecosystem where is defined from the shore to 24 nautical mile with total area of about 140,000 km^2. We hypothesized that different fishing intensities correspond to different rates of ecosystem functioning. We tested this hypothesis by applying inverse models on data collected in two different time periods (i.e. 1990s and 2000s) to reconstruct carbon flows between different functional groups. We calculated eight functional attributes, food web efficiencies of six functional groups, and eight functional food web indices in the coastal ecosystem of Vietnam for two periods with contrasting fishing intensities (high vs. low). Functional attributes were: total consumption, egestion, excretion, respiration, gross and net primary production, secondary production, and natural mortality (e.g. mortality not included in predation or fishing mortality). We also tested if food web efficiencies and functional food web indices were different between the two periods. Biomass data are considered as one of the most unreliable data sources in the present study. This is due to lack of time series data from fisheries dependent and independent research or unstandardized data collection methods. Therefore, we used sensitivity analysis to assess the solution's robustness to variations in the input data (biomass), and a perturbation analysis of input data was carried out once the initial balanced solution was obtained.

5.2. Methods

5.2.1. The data

In the present study, we only focus on the coastal ecosystem of Vietnam. Studied periods are the end of 1990s (1995-1999) and the early 2000s (2000-2005) when fish stock data and landings were available.

In the 1990s' model, biomasses of all demersal fish groups, cephalopods, crustaceans, and shrimps were estimated from fish and shrimp trawl surveys conducted between 1996 and 1999. In these trawl surveys, biomass of species were estimated based on the swept-area method (Gunderson 1993) and assumed catchabilities of 0.5 and 0.64 for fish and shrimp trawl fishery, respectively. The swept-area method assumes that biomass/density of species distributing in the swept area is proportional with biomass/density of species in entire the studied area (Gunderson 1993).

The biomasses of small pelagic groups were estimated from acoustic surveys conducted from 29 April to 29 May 1999 in a collaboration between the Research Institute for Marine Fisheries of Vietnam and the Southeast Asian Fisheries Development Center (Hassan et al. 1999). In the acoustic method, an echosounder transducer was mounted in the research vessel to vertically project sound beam into water. This sound beam can then detect the suspending objects under water such as fish, zooplankton, phytoplankton, etc. and reflect signal called "acoustic backscattering" to a receiver on board (Foote 1980). Then biomass of species was estimated by assuming that there was a proportional relationship between acoustic backscattering and fish biomass/density. A total of 43 acoustic transects (33 transects of 60nm and 10 transects of 30nm) were conducted within the acoustic survey of Hassan et al. (1999). Detailed methodologies were described in Hassan et al. (1999).

Biomass data for tuna, large predators, zoobenthos, zooplankton and phytoplankton of the 1990s' model were obtained from the literature (Christensen et al. 2003, Duana et al. 2009).

In the 2000s, biomass data of small pelagic groups were obtained from acoustic surveys conducted between 2003 and 2005 using 84 transects (RIMF 2005c). The methods for acoustic surveys of the 2000s' model were the same as for the model of the 1990s. The biomasses of tuna and large predators were estimated by a stock assessment program between 2003 and 2005 using gillnet fishery (RIMF 2005b).

The biomasses of all demersal fish groups, cephalopods, crustaceans, and shrimps were estimated from fish and shrimp trawl surveys conducted between 2000 and 2005 (RIMF 2005a) using the same method for trawl surveys as described above. Details are indicated in the Table 5.1. Biomass data of mammals, sea turtle, zoobenthos, zooplankton and phytoplankton were obtained from the literature (Chen et al. 2008a, Chen et al. 2008b, Van et al. 2010).

Details on the assembled biomass data sources are available in Appendix 5.1. Biomass data, reported as $t \cdot km^{-2}$, were converted to $t \cdot C \cdot km^{-2}$ using the following conversion factors: 0.05, 0.1, 0.13 and 0.15 ton carbon/ton wet weight for invertebrates (Hendriks 1999), phytoplankton (Lignell 1990), fish (Sakshaug et al. 1994) and mammals (Pinkerton and Bradford-Grieve, unpublished data), respectively (Table 5.2). For zooplankton, the equation of Wiebe (1988) was used: $\log(W) = -1.537 + 0.852 \cdot \log(C)$, where W and C are wet and carbon weight, respectively.

Table 5.1. Number of survey trips used to estimate biomass data for demersal fish and shrimp groups. Data from these surveys were analysed and assembled for one ecosystem model for all these regions.

Region	Name of survey	Number of survey trips	Studied period
Tonkin Gulf	Assessment of coastal marine resources using fish otter trawl	5	2001-2005
	Assessment of coastal marine resources using shrimp trawl	4	2002-2003
Central region	Assessment of coastal marine resources using fish otter trawl	2	2004-2005
South-eastern and south-western region	Assessment of coastal marine resources using fish otter trawl	7	2000-2005
	Assessment of coastal marine resources using shrimp trawl	4	2001-2002

Total catch reconstructed in Chapter 3 were partly used for input data of the present study. In this chapter, only Vietnamese coastal fisheries were taken into consideration for modeling. Annual total mean catch by the coastal fisheries was then calculated per ecological group and were expressed as $t{\cdot}C{\cdot}km^{-2}{\cdot}year^{-1}$ using the conversion factors as with biomass data (Table 5.3). Since the inverse models do not allow unidentified groups and thus catch data of "other groups" as described in the Chapter 3 were separated using the same catch rate as in Table 3.2. Because there is no information on discard on the Vietnamese coastal fisheries, we assumed that all catches were landed equal to total landings.

Table 5.2. Minimum (Min), maximum (Max), mean, standard deviation (SD) (unit of $t{\cdot}C.km^{-2}$) and coefficient variation (CV) of standing biomass used for the inverse models in 1990s and 2000s of the Vietnamese coastal ecosystem. CVs used for sensitivity analysis, (-) mean not available. Abbreviations of the functional group names are: TUN = Tuna, LAR = Large predators, LAD = Large demersal fish, OTD = Other demersal fish, LAP = Large pelagic fish, MEP = Medium pelagic fish, SMP = Small pelagic fish, CEP = Cephalopods, SHR = Shrimp and CRU = Crustaceans.

Group	1990s				2000s			
	Min	Max	Mean ± SD	CV %	Min	Max	Mean ± SD	CV %
TUN	-	-	-	56[a]	0.026	0.091	0.051 ± 0.021	35
LAR	-	-	-	56[a]	0.040	0.077	0.061 ± 0.014	22
LAD	0.005	0.162	0.118 ± 0.061	55	0.050	0.201	0.095 ± 0.045	44
OTD	0.025	0.197	0.080 ± 0.054	63	0.048	0.169	0.060 ± 0.008	56
LAP	0.010	0.100	0.050 ± 0.035	69	0.015	0.200	0.040 ± 0.016	52
MEP	0.020	0.140	0.071 ± 0.048	67	0.077	0.131	0.092 ± 0.021	23
SMP	0.010	0.110	0.055 ± 0.035	64	0.013	0.044	0.028 ± 0.010	37
CEP	0.020	0.054	0.033 ± 0.011	33	0.024	0.083	0.043 ± 0.019	44
SHR	0.020	0.103	0.055 ± 0.025	42	0.010	0.078	0.036 ± 0.020	42
CRU	0.100	0.560	0.246 ± 0.160	51	0.097	0.230	0.124 ± 0.045	25

Note: [a] calculated as average of all known coefficients of variation for biomass

Table 5.3. Minimum (Min), maximum (Max), mean and standard deviations (SD) of total catch by ecological groups ($t \cdot C \cdot km^{-2} \cdot year^{-1}$) in the Vietnamese coastal ecosystem for the period of 1990s to 2000s. The values in bold were used as the constraints in the inverse models. (-) mean not available. SEA = Sea turtle and other abbreviations of the ecological group names are indicated in the Table 5.2.

Group	1990s				2000s			
	Min	Max	Mean	SD	Min	Max	Mean	SD
SEA	-	-	-	-	**0.006**	**0.015**	0.010	0.004
TUN	**0.021**	**0.067**	0.044	0.019	**0.032**	**0.078**	0.052	0.017
LAR	**0.066**	**0.098**	0.082	0.012	**0.034**	**0.157**	0.095	0.051
LAD	**0.054**	**0.105**	0.077	0.018	**0.025**	**0.210**	0.071	0.056
OTD	**0.028**	**0.079**	0.058	0.021	**0.027**	**0.166**	0.059	0.047
LAP	**0.011**	**0.039**	0.025	0.009	**0.012**	**0.048**	0.030	0.013
MEP	**0.063**	**0.105**	0.090	0.011	**0.053**	**0.136**	0.097	0.031
SMP	**0.010**	**0.041**	0.027	0.013	**0.012**	**0.068**	0.031	0.021
CEP	**0.042**	**0.083**	0.066	0.013	**0.026**	**0.136**	0.083	0.030
SHR	**0.008**	**0.028**	0.019	0.007	**0.002**	**0.041**	0.022	0.021
CRU	**0.061**	**0.100**	0.082	0.012	**0.014**	**0.138**	0.091	0.042

5.2.2. The models

In these models, we assumed that there was no change in biomass ($\Delta B_i = 0$) during each studied period and that net migration was zero (migration out of, or into the study area, food intake of predators that are not part of the system). Also, production was equal to the biomass lost to fishing, predation, and natural mortality other than predation (e.g. disease, other natural causes of death and unexplained mortality (unsuspected processes occurring in the ecosystem)). The general structure of an inverse model includes (i) compartment mass-balance equations, (ii) data equations, and (iii) constraints (Savenkoff et al. 2007, De Laender et al. 2010). The mass-balance equations specify that, for each compartment, the sum of inflows (consumption) is balanced by the sum of outflows (production, respiration, and egestion). The data equations are used to fix the value of certain flows (or combination of flows) from observations or field experiments, whereas the constraints incorporate general knowledge about ecology and physiology. One of the main advantages of inverse modeling is its potential in under-sampled environments (Kones et al. 2006). Overall, inverse modeling can be used to reconstruct a large number of unknown flows or interactions from a relatively small number of observations and enables to quantify mass or energy exchange between food web compartments (Savenkoff et al. 2004, Vezina et al. 2004, Savenkoff et al. 2007, De

Laender et al. 2010). Furthermore, inverse modeling allows combining data with eco-physiological constraints on energy flows (inequalities) obtained from the literature (Vezina and Platt 1988). A flow chart of linear inverse model (LIM) process is indicated in the Figure 5.1.

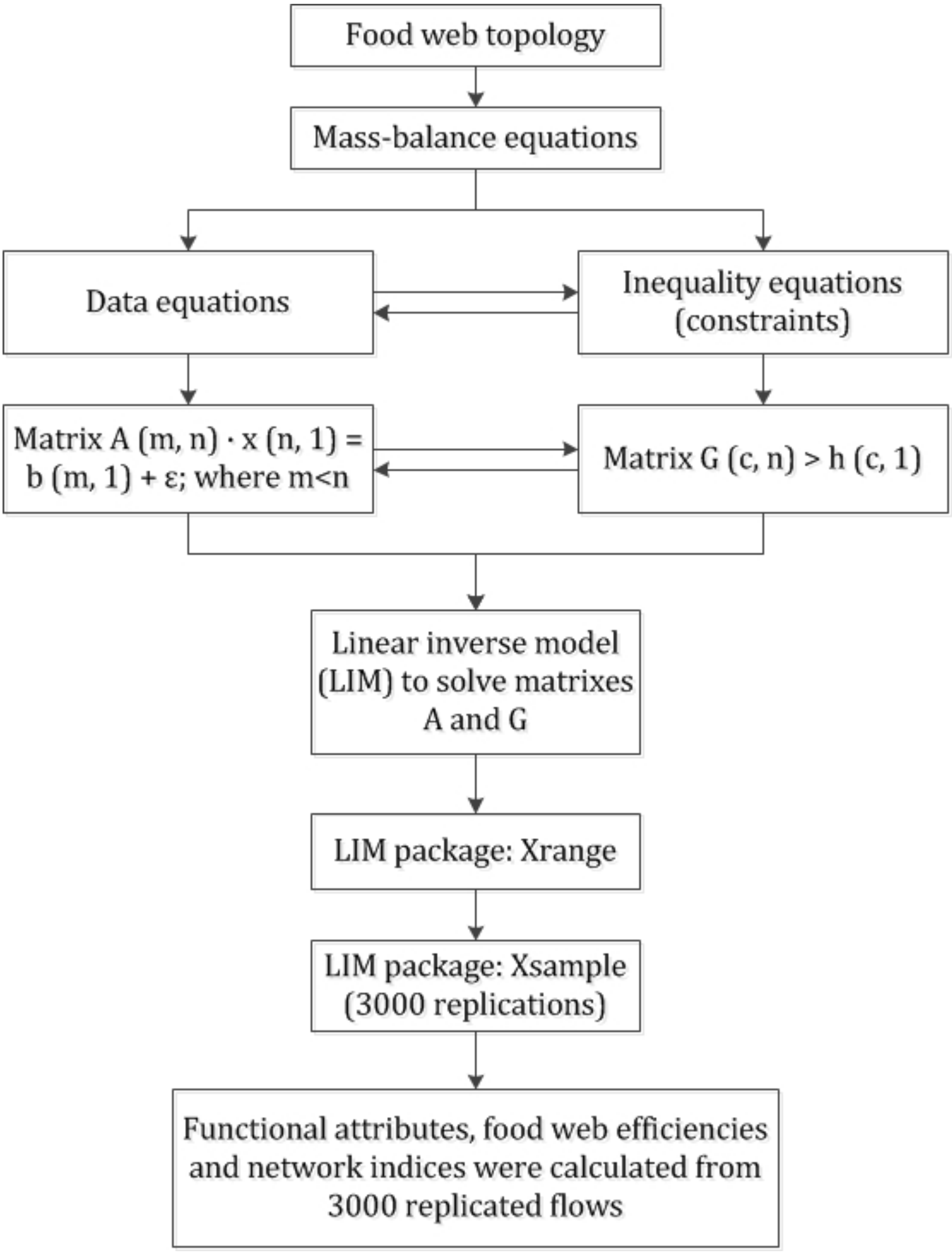

Figure 5.1. Basic steps of linear inverse model performed in the present study.

The first step to solve the inverse model was to write out the mass-balance and data equations in matrix form and set up a rectangular matrix *(A)* to represent the connections in the food web. The rows of the matrix are the mass-balance relations and the columns are flows in the food web. Once the matrix *A (m, n)* is constructed, we specify a vector *b* that has as many rows as *A* and gives the expected values of the mass-balance relations and values of the flows entered as data equations. Vector *b (m x 1)* thus contains the right-hand sides of the mass-balance and data equations. After that inverse methodology calculate a vector *x (n x 1)* that has as many elements as there are columns in *A*. Vector x represents the flows that once multiplied by *A*, approximate vector *b*. Then the matrix form of the data equations and mass-balance equations is written as following:

$A.x = b + \varepsilon$ (Eq. 5.1)

The constraints (inequality relations) can also be written in matrix form (matrix *G*) by setting up a matrix with as many rows as there are inequality relations (constraints, *c*) and the same number of columns (representing the flows) as *A*. We used a matrix *G (c, n)* and a vector *h (c x 1)* with as many elements as there are rows in *G* and corresponding to the right-hand sides of the inequality relations. Then the matrix form of the constraints is written as following:

$G.x > h$ (Eq. 5.2).

Two inverse models were constructed, one representing carbon flows in the ecosystem in the 1990s and one for the 2000s. Both models included 18 functional groups (including living and non-living compartments) interconnected by carbon flows. The models included mammals, sea turtles, seven groups of fish, five groups of invertebrates, primary producers, bacteria, detritus, and dissolved organic carbon (Appendix 5.1).

The food web topology (Appendix 5.2) of the inverse models in the 1990s and 2000s was the same. Phytoplankton assimilated dissolved inorganic carbon (DIC), an external food web model input, transforming it into particulate and dissolved organic carbon (DOC). Respiration, egestion and excretion of all populations were introduced by flows to DIC, detritus (DET) and DOC, respectively. For each living compartment, natural mortality (MOR) was introduced representing non-predatory mortality. MOR was then transferred to DET, representing decomposition of dead individuals. DET was partly consumed by detritus feeders and the rest was transferred to DOC. DOC was taken up by bacteria which were on their turn consumed by zooplankton. The fisheries catches were represented as external flows (Appendix 5.4).

In this study, minimum and maximum values of the total catch, as calculated from the landings data, were used as the constraints in the inverse models (Table 5.3). For other constraints such as consumption, respiration, excretion and ingestion, literature data were used (Table 5.4). In summary, 148 and 154 constraints (Appendix 5.4) were included in the models for the 1990s and 2000s, respectively.

The inverse models were solved using the R package LIM. The inverse food web models had fewer equality equations than unknown flows and thus represented under-determined systems (van Oevelen et al. 2010). Therefore, the equality matrix had an infinite number of solutions. A Monte Carlo procedure was used to obtain a range of possible solutions (van den Meerssche et al. 2009; De Laender et al. 2011) using the function Xranges (Appendix 5.3). In this study, we calculated 3000 possible solutions (realizations), a value considered large enough to adequately sample the entire solution space (Kones et al. 2009) using the function Xsample.

Table 5.4. Constraint values except the constraints of total catch used in the inverse models of the coastal ecosystem in Vietnam.

Constraints	Characteristic	Min	Max	Unit	Source
Consumption:					
Mammal (only used for the 2000s inverse model)	Annual ration[a]	2044	9125	%	Olson and Galvan-Magana (2002)
Large predators	Annual ration[a]	164	3387	%	Young et al. (2010)
Tuna	Annual ration[a]	712	1496	%	Griffiths et al. (2009)
Small pelagic fish	Annual ration[a]	219	6789	%	Maes et al. (2005)
Medium pelagic fish	Annual ration[a]	489	1405	%	Pakhomov et al. (1996)
Other pelagic fish	Annual ration[a]	1314	8395	%	Pakhomov et al. (1996)

Cephalopods	Annual ration[a]	730	3285	%	Quintela and Andrade (2002)
Large demersal fish	Annual ration[a]	332	419	%	Bulman and Koslow (1992)
Other demersal fish	Annual ration[a]	1668	4781	%	Pedersen (2000)
Shrimp	Annual ration[a]	172	9965	%	- Minimum value cited in Maynou and Cartes (1997) - Maximum value cited in Norte-Campos and Temming (1994)
Crustaceans	Annual ration[a]	986	17155	%	Wolff and Cerda (1992)
Zoobenthos	Annual ration[a]	2920	10990	%	Francesc and Joan (1998)
Zooplankton	Annual ration[a]	3395	26426	%	Froneman et al. (1996)
Growth efficiency:					
Mammal	Production/ Consumption	0.01	0.10		Christensen and Pauly (1992)
All fish groups, shrimp and crustaceans	Production/ Consumption	0.10	0.30		Christensen and Pauly (1992)
Zooplankton	Production/ Consumption	0.25	0.50		Christensen and Pauly (1992)

Assimilation efficiency:					
Shrimp	Assimilation efficiency	70	80	%	Franco et al. (2006)
Zooplankton	Assimilation efficiency	50	90	%	Sengul and Hans (2002)
Other groups except phytoplankton	Assimilation efficiency	70	90	%	Christensen and Pauly (1992)
Excretion rate:					
Phytoplankton	Excretion rate	0.05	0.6	Fraction of NPP	Vezina and Platt (1988)
Zooplankton	Excretion rate	0.33	1	Fraction of respiration rate	Vezina and Platt (1988)
Small pelagic fish	Excretion rate	0.05	0.15	Fraction of ingestion	Klumpp and Westernhagen (1986)
Bacteria	Excretion rate	0.33	1	Fraction of respiration rate	Vezina and Platt (1988)
Respiration rate:					
Phytoplankton	Respiration rate	0.05	0.3	Fraction of GPP	Vezina and Platt (1988)
Zooplankton	Respiration rate	25.34	41.30	$year^{-1}$	Ikeda (2012)
Cephalopods	Respiration rate	38.48	71.71	$year^{-1}$	Boucher-Rodoni and Mangold (1989)
Small pelagic fish	Growth	31.94	96.00	$year^{-1}$	Rudstam et al.

	respiration rate				(1988)
Small pelagic fish	Maintain respiration rate	6.94	8.76	$year^{-1}$	Megrey et al. (2007)
Large demersal fish[b]	Respiration rate	3.25	14.44	$year^{-1}$	Drazen and Yeh (2012)
Medium pelagic fish[c]	Respiration rate	1.91	11.55	$year^{-1}$	van der Lingen (1995)
Tuna[d]	Respiration rate	9.65	16.63	$year^{-1}$	Gooding et al. (1981)
Shrimp[e]	Respiration rate	0.25	1.404	$year^{-1}$	Franco et al. (2006)
Large predators[f]	Respiration rate	2.65	5.79	$year^{-1}$	Scharold et al. (1989)
Natural mortality:					
Large predator		0.001	0.05	$year^{-1}$	Knip et al. (2012)
Medium pelagic fish		0.14	0.51	$year^{-1}$	Ghosh et al. (2009)
Small pelagic fish		2.19	2.95	$year^{-1}$	Newberger and Houde (1995)
Shrimp		2.11	2.41	$year^{-1}$	Niamaimandi et al. (2007)
Crustaceans		0.42	0.87	$year^{-1}$	Hewit et al. (2007)
Other demersal fish		0.2	0.7	$year^{-1}$	Norman et al. (2004)

Large demersal fish		0.12	0.29	$year^{-1}$	Newman (2002)
Other constraints:					
Viral lysis of bacteria		10	40	Percentage of bacterial production	Fuhrman (2000)
GPP of phytoplankton	Standing stock-specific GPP	182.5	547.5	$year^{-1}$	MacIntyre et al. (2002)

[a] from daily ration were multiplied to 365 to convert into annual ration

[b] Calculated from 0.09–0.40 micro moles O_2 g^{-1} h^{-1}

[c] Calculated from 0.053 to 0.32 ml O_2 g^{-1} h^{-1}

[d] Calculated from 0.382 to 0.658 mg O_2 g^{-1} h^{-1}

[e] Calculated from 0.09 – 0.5 ml O_2 g^{-1} h^{-1}

[f] Calculated from 105.3 to 229.3 mg O_2 kg^{-1} h^{-1} .

5.2.3. Functional attributes and food web efficiency

Eight functional attributes and food web efficiencies were calculated from the flows generated by the Monte Carlo approach (3000 realizations). The functional attributes calculated in this study were total consumption, egestion, excretion, respiration, primary production (gross and net primary production), secondary production and other mortality (mortality other than predation or fishing). Total consumption of a consumer was calculated as the sum of all flows arriving in a consumer group including respiration, egestion, and all mortality (i.e. fishing and natural mortality). Egestion, excretion and respiration were calculated as the flows to detritus, dissolved organic carbon, and dissolved inorganic carbon, respectively. All these attributes have the same unit: $t \cdot C \cdot km^{-2} \cdot year^{-1}$.

The food web efficiency (unitless) of mammals, demersal fishes, large predators, pelagic fishes, large invertebrates and zooplankton were calculated using the equation of Rand and Stewart (1998):

$FWE_{group_A} = (\sum_{i=1}^{n} group_A \rightarrow i)/NPP$ (Eq. 5.3) where $group_A \rightarrow i$ are flows leaving the group *A*, representing export to higher trophic levels including fisheries ($t \cdot C \cdot km^{-2} \cdot year^{-1}$); NPP is the total net primary production (gross primary production minus phytoplankton respiration; $t \cdot C \cdot km^{-2} \cdot year^{-1}$).

Significant differences of the functional attributes and food web efficiency between the 1990s and the 2000s were tested by Wilcoxon Rank Sum tests (with $\alpha = 0.05$).

5.2.4. Network analysis

We used eight functional food web indices to evaluate differences between the two time periods. A detailed description of these functional food web indices can be found in Latham (2006). We calculated four general functional food web indices: total system throughput (TSTP), ascendency, development capacity and overhead. TSTP is obtained by summing all flows in and out the system and is used to measure the size and activity of the system (Ulanowicz 2004, Goerner et al. 2009). The more material/energy flowing through the system are, the larger the value of TSTP.

Ascendency, development capacity and overhead are measures of development and growth of ecosystem (Latham 2006). Ascendency is calculated by multiplying the average mutual information (an organization component) and TSTP (a size component) (Latham 2006). Overhead is calculated by difference between development capacity and ascendency. Development capacity is the upper bound on ascendency (Ulanowicz 2004) and was calculated by making the sum between ascendency and overhead. These indices were calculated for all 3000 replications of the Monte Carlo solution. Note that while mean ascendency + mean overhead = mean development capacity, this is not the case for the minimum and maximum values.

In addition, four other functional food web indices were calculated: (1) the ratio of ascendency and development capacity, a measure for the system's development (Latham 2006), (2) constraint efficiency index, a measure of the degree of inherent network constraints to maximum network uncertainty (Latham 2006), (3) Finn's cycling index, describing the degree of cycling in a system (Patten and Higashi 1984), and (4) the average mutual information, a measure of the average amount of constraint placed upon an arbitrary unit of flow anywhere in the network (Ulanowicz 2004). Details of code, formula and reference sources of the functional food web indices are indicated in the Appendix 5.5. All functional food web indices were calculated using the R package "NetIndices" (Soetaert and Kones 2008).

Significant differences between the 1990s and the 2000s of functional food web indices were tested as for the functional attributes and food web efficiency.

5.2.5. Sensitivity analysis

The robustness of the estimated carbon flows to calculate the functional attributes, food web efficiency and functional food web indices to changes in the used biomass data was tested using a sensitivity analysis. The sensitivity analysis was carried out by randomly sampling (100 iterations) biomass data from uniform distributions between the mean biomass plus and minus one standard deviation (±SD) (Savenkoff et al. 2004). We obtained 54 and 65 successful solutions of the 100 iterations for the models for the 1990s and 2000s, respectively. The successful solutions correspond to biomass values that can be used to solve the models. Values of the functional attributes, food web efficiencies and functional food web indices were re-calculated using the randomly sampled biomass data and compared to the values calculated from the original model using a variation index:

$$Variation = \frac{\left\| X_O - X_p \right\|}{X_O} \cdot 100\% \qquad \text{(Eq. 5.4)}$$

where X_o and X_p are the original values and the values based on randomized biomass data, respectively.

5.3. Results

5.3.1. Functional attributes

In general, ecosystem consumption, egestion, excretion, respiration changed over the two periods contributing total outflows of the 1990s higher than of those in the 2000s (858 and 816 t·C·km^{-2}·year^{-1}, respectively, Table 5.5). The other functional attributes to ecosystem also showed changes between two periods (Table 5.5).

Consumption, production, respiration and egestion of almost all groups decreased between the 1990s and the 2000s, except for large invertebrates (i.e. shrimp, cephalopods, other crustaceans) and zooplankton (Figure 5.2).

Table 5.5. Minimum (Min), maximum (Max) and mean ± standard deviations (SD) ($t \cdot C \cdot km^{-2} \cdot year^{-1}$) of some functional attributes of the Vietnamese coastal ecosystem in each time period. Numbers in brackets are proportions of the corresponding functional attributes with total system throughput. An asterisk (*) indicates that a significant difference was estimated between the two periods based on a Wilcoxon Rank Sum test (with α = 0.05).

Functional attributes	Model in 1990s				Model in 2000s				Significance
	Min	Max	Mean	SD	Min	Max	Mean	SD	
Total consumption	329.2	813.3	541.2	70.2	310.7	723.1	494.7	64.7	*
Total egestion	47.2	393.3	149.8	51.5	40.9	330.6	132.2	44.5	*
Total excretion	126.3	504.8	293.7	59.8	114.2	470.8	257.9	57.2	*
Total respiration	72.6	270.1	155.4	28.9	69.2	239.2	140.2	27.4	
Total outflows	575.3	1681.5	858.1	115.0	535	1763.7	816.0	129.7	*
Total prod. (except phytoplankton production)	80.9	425.7	235.8	59.4	70.8	392.8	222.4	53.7	*
Gross primary production	195.7	576.4	467.3	99.7	197.3	580.9	441.9	106.7	
Phytoplankton respiration	11.6	149.1	66.6	30.8	11.5	153.3	64.3	30.2	
Net primary production	184.1	427.3	400.8	103.8	185.8	427.6	377.5	108.8	*
Secondary production	35.1	117.5	84.3	15.1	33.6	145.4	101.1	20.5	
Total other mortality	79.0	620.0	339.1	10.5	47.2	565.3	314.9	19.7	*

The largest reductions on functional attributes (i.e. consumption, production, respiration and egestion) detected were for pelagic fish (i.e. small, medium and large pelagic fish) with an overall reduction rate of 28% (Figure 5.2). These reductions were lower for top trophic level groups (mammal and sea turtle), predator groups (tuna and large predators) and demersal fish (large and other demersal fish) with overall reduction rates of 15, 22 and 19%, respectively (Figure 5.2). The average rates of consumption, production, respiration and egestion increased from the 1990s to the 2000s by 34% for large invertebrates and by 22% for zooplankton (Figure 5.2).

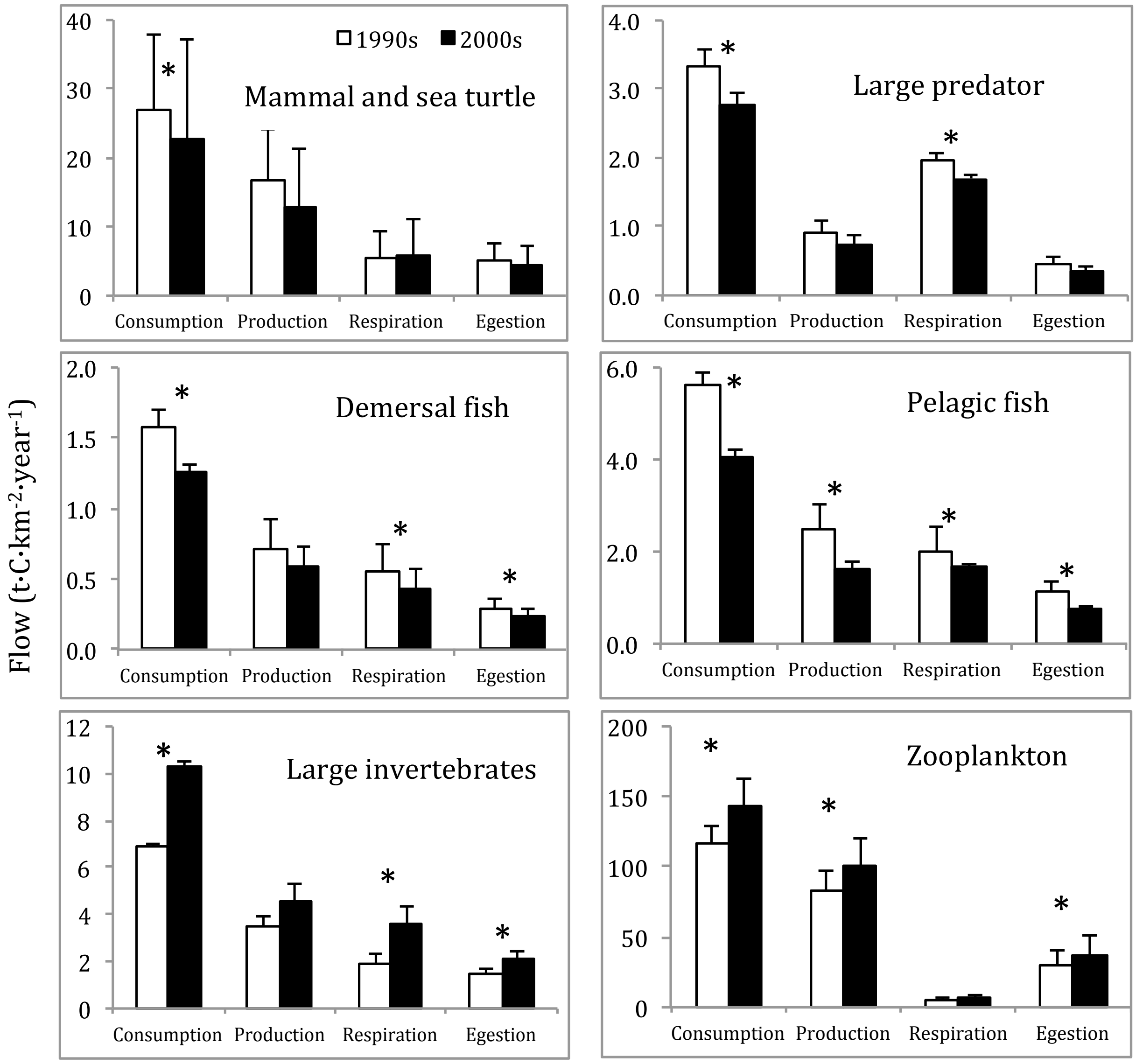

Figure 5.2. Consumption, production, respiration and egestion patterns in 1990s (white bars) and 2000s (black bars) for six functional groups. Standard deviations are shown. An asterisk (*) indicates that a significant difference was estimated between the two periods based on a Wilcoxon Rank Sum test (with α = 0.05).

5.3.2. Food web efficiency

There was a decrease (up to 40%) of the food web efficiency calculated based on the net production of mammals and sea turtle, demersal fish and large predators, and pelagic fish. However, the food web was about 10% and 41% more efficient in producing large zooplankton and invertebrates, respectively, in the 2000s than in the 1990s (Figure 5.3).

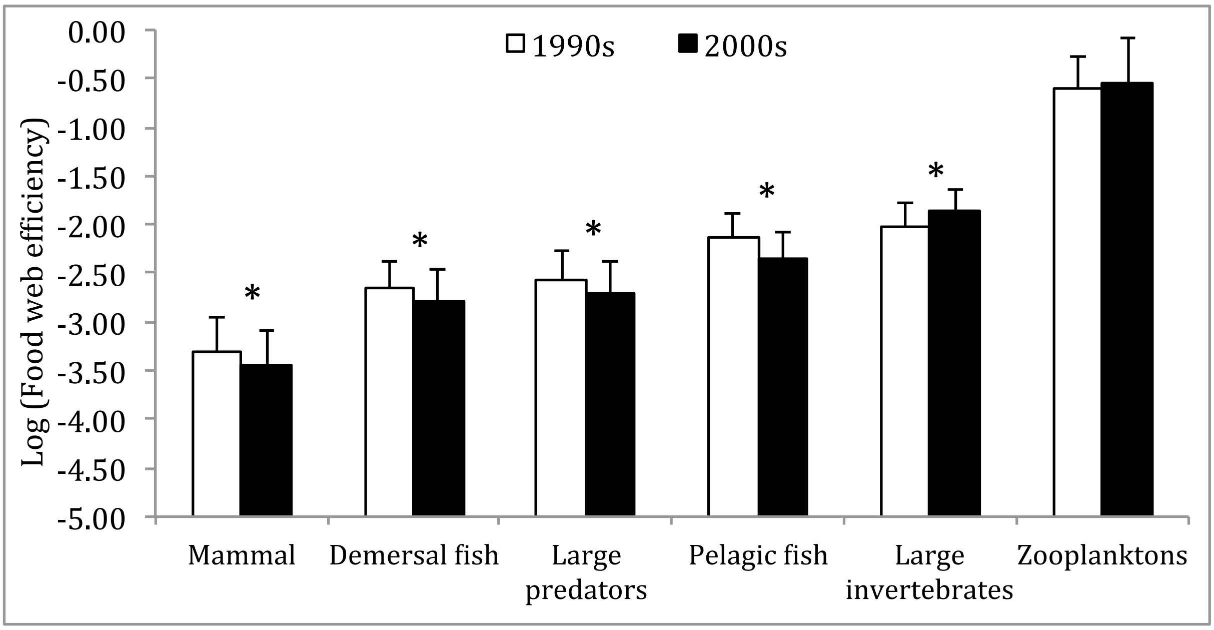

Figure 5.3. Food web efficiency expressed as ratios between net productivities of groups and net primary production corresponding to time periods. Means and standard deviations were calculated by analyzing all 3000 Monte Carlo solutions of food web models. An asterisk (*) indicates that a significant difference was estimated between the two periods based on a Wilcoxon Rank Sum test (with $\alpha = 0.05$).

5.3.3. Network analysis

In general, total system throughput (TSTP), ascendency, overhead and capacity were significantly higher in the 1990s than in the 2000s (Wilcoxon Rank Sum test; $p < 0.05$) (Table 5.6).

When considering the ratio of ascendency and development capacity, the constraint efficiency, and Finn's cycling index, a decreasing trend was found from the 1990s to the 2000s (Figure 5.4A, B, C). The average mutual information index did not differ between the two time periods (Figure 5.4D).

Table 5.6. Minimum (Min), maximum (Max), mean and standard deviation (SD) ($t \cdot C \cdot km^{-2} \cdot year^{-1}$) of some total system indices of the Vietnamese coastal ecosystem in 1990s and 2000s. TSTP is total system throughput. An asterisk (*) indicates that a significant difference was estimated between the two periods based on a Wilcoxon Rank Sum test (with α = 0.05).

Network index	Model in 1990s				Model in 2000s				Sign.
	Mean	Min	Max	SD	Mean	Min	Max	SD	
TSTP	1998.2	1122.0	2935.8	324.7	1841.0	1001.7	2547.3	326.8	*
Ascendency	3613.6	1637.1	6512.8	736.1	2804.8	1436.2	4892.9	759.8	*
Overhead	3931.2	2375.1	5458.2	511.2	3722.9	2233.6	5038.0	506.3	*
Dev. cap.	7544.8	4657.3	11126.7	977.4	6527.7	4037.0	9856.3	1022.5	*

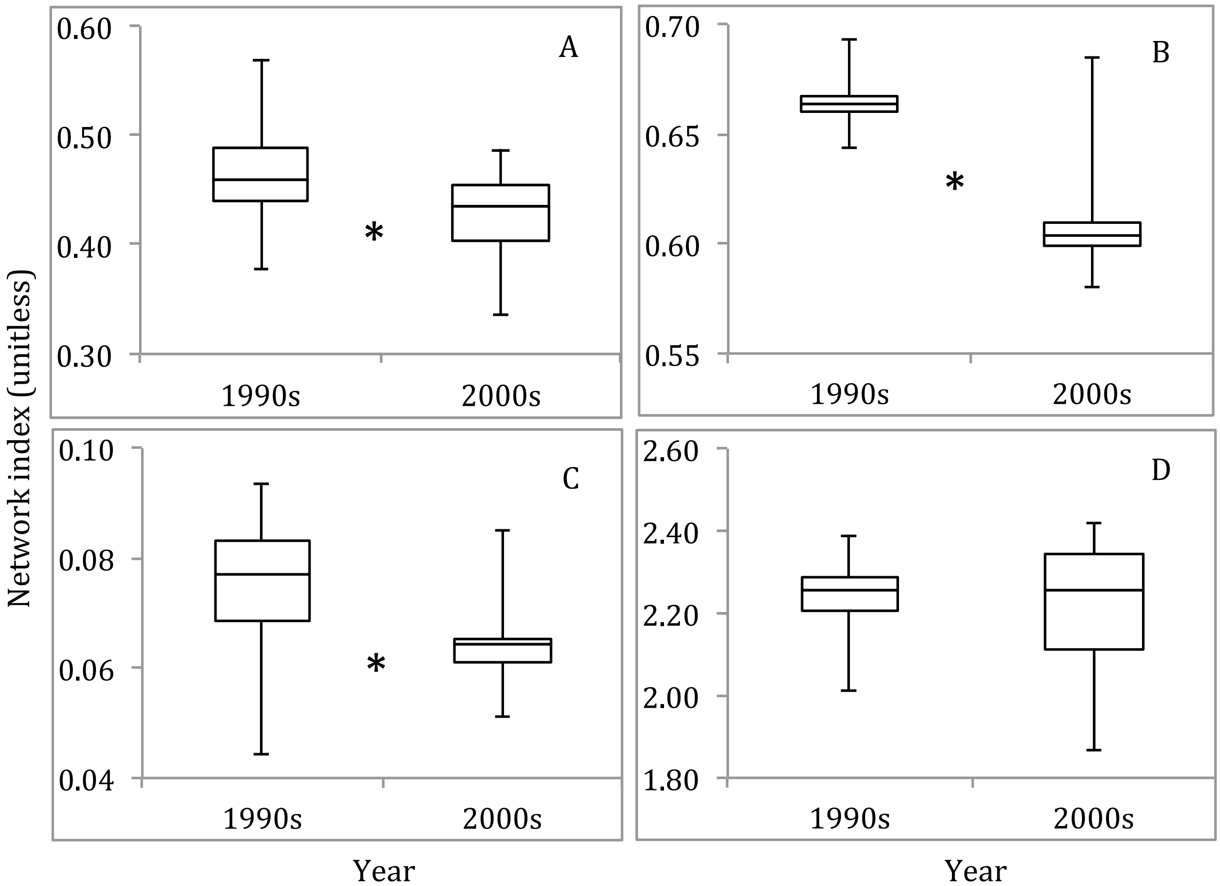

Figure 5.4. Comparison of network indices (unitless) between two time periods of the Vietnamese coastal ecosystems. Values of minimum, first quartile, mean, third quartile and maximum are shown in the boxplot. A: ratio of ascendency and development capacity of the system; B: constraint efficiency index; C: Finn's cycling index and D: average mutual information of system. An asterisk (*) indicates that a significant difference was estimated between the two periods based on a Wilcoxon Rank Sum test (with α = 0.05).

5.3.4. Sensitivity analysis

The functional attributes, food web efficiencies and the functional food web indices responded differently to the perturbations of the biomass data. In general, the variation between the initial and perturbed solutions of the functional food web indices was higher in the 1990s than in the 2000s (Table 5.7). The food web efficiencies were more sensitive to changes in the biomass data (an averaged variation of nearly 64%) than the functional attributes and functional food web indices (averaged variations of 48 and 31%, respectively) (Table 5.7). The food web efficiency of large predators was most sensitive to changes in biomass data, with a maximum variation of 83.3% for the 2000s model (Table 5.7).

However, varying the biomass data did not change the general trends. We still found reductions of the functional attributes and the food web efficiency for top trophic level groups and increases of those for lower trophic levels groups over time. In addition, the functional food web indices for the 1990s were still higher than of those for the 2000s ($p < 0.05$).

5.4. Discussion

5.4.1. Changes on the functional attributes and food web efficiency

Using the functional attributes such as production and related measures (e.g. food web efficiency) as ecosystem indicators is not new in ecological assessments. Production integrates a set of fundamental ecological processes and is an indicator of ecosystem functioning (Tilman et al. 1997). The presented results indicate that there is a general decline of production for almost all ecological groups at high trophic levels, except for zooplankton and large invertebrates (Figure 5.2). Total production of large invertebrates increased by up to 41% from the 1990s to 2000s (Figure 5.2), although averaged biomasses of these groups did not considerably increase from the 1990s to 2000s. This can be explained by the increase in secondary production that can be consumed by large invertebrates during studied periods. Nevertheless, changes in the lower trophic level could be more subtle and may be more related to environmental factors such as climate change (Crain et al. 2009) that were not accounted for by our models.

Table 5.7. Variation (%) from values estimated by initial and perturbation solution of the Vietnamese coastal ecosystem in 1990s and 2000s after performing sensitivity analysis (- or + indicates the direction of the variation).

	Model in 1990s		Model in 2000s	
	-	+	-	+
Functional attributes				
Total consumption	41.38	45.36	43.74	46.45
Total egestion (detrital flows)	48.49	54.84	51.83	48.19
Total excretion	46.85	48.12	43.92	47.54
Total respiration	39.12	44.41	41.40	45.77
Total production	42.60	42.68	42.99	45.87
Gross primary production	45.88	43.91	49.41	46.70
Secondary production	56.04	83.49	71.40	80.00
Total other mortality	45.44	43.06	48.73	44.92
Food web efficiency of				
Mammal	67.75	51.22	68.30	64.39
Demersal fish	74.22	76.98	71.54	60.35
Large predators	74.33	59.99	68.91	83.26
Pelagic fish	78.41	58.03	76.54	57.36
Large invertebrates	66.80	55.52	63.58	63.23
Zooplanktons	50.05	45.39	64.72	57.20
Network index				
Total system throughput	44.66	39.01	46.58	39.87
Ascendency	16.58	50.75	17.35	51.44
Overhead	63.34	26.20	76.27	24.92
Capacity	36.87	40.45	40.68	40.60
Development capacity of the system	13.71	19.30	15.37	21.19
Constraint efficiency index	4.51	8.92	1.62	5.94
Finn's cycling index	48.51	9.49	37.77	30.10
Average mutual information of system	12.52	20.40	13.03	18.22

In the coastal system of Vietnam, the ecosystem structure exhibits a shift from the 1990s to the 2000s however this shift was in a very short term and it is unclear that ecosystem structure can be changed in a longer term. Therefore, it is necessary to have long-term studies to evaluate changes of the Vietnamese coastal ecosystem structure. In fact, it is at present unclear to what extent the changes between two studied periods were only caused by fishing, or if other environmental perturbations contributed to such changes. In fact, climate change can cause a shift of marine biodiversity locally, regionally and globally (Vitousek et al. 1997). Also, Halpern et al. (2008) concluded that anthropogenic drivers associated with global climate change are an important factor impacting global ecosystems but also that a greater impact is expected in offshore ecosystems. Crain et al. (2009) also indicated that polar regions are more vulnerable to climate change than coastal ecosystems. Thus, we believe that fishing pressure has probably a more important impact on the ecosystem structure and functioning than climate change. Nevertheless, future studies should integrate a variety of disturbance sources into more representative models to quantify the relative contribution of several potential drivers to changes in the structure and functioning of the Vietnamese coastal ecosystem.

Results of the present studies indicated that there were overall reductions of consumption, respiration and egestion of pelagic fish, top trophic level (mammal and sea turtle), predators (tuna and large predators) and demersal fish (15 - 28%). Consumption, respiration and egestion of invertebrates and zooplankton increased by 34% and 22%, respectively (Figure 5.2). Although the carbon flows between trophic groups differed between the two time periods (Figure 5.2), the total functional attributes of entire system are relatively unchanged (Table 5.5). The differences in carbon flow is likely to be due to changes in the biomasses of these trophic groups between studied periods. The latter is on his turn probably related to increased fisheries landings of pelagic fishes, large tuna and other predatory fishes (Table 5.3), hence reducing the consumption of invertebrates and zooplankton. This appears to be evidence of a fishery-induced trophic cascade (Ferretti et al. 2010), but not a system-wide reduction of flow for the Vietnamese coastal ecosystem. This phenomenon was also found in the northern Gulf of St. Lawrence ecosystem by (Savenkoff et al. 2007). They indicated that total consumption of entire ecosystem were not significantly differences between studied periods. However, when total consumption was distributed among key ecological groups, then there were clear differences on total consumption of these ecological groups.

The decrease of the food web efficiency based on the production of mammals, demersal fishes and large predators reflected the observed standing stock reduction for these groups (Figure 5.3). The lower carbon requirements of these stocks of a smaller size led to increasing carbon transfers to lower trophic levels such as large invertebrates and zooplankton. Although it has been established that total production at the base of the food web will to a large extent determine the productivity at the top (Ware and Thomson 2005), the pathways and efficiency of the transfer between primary producers and top consumers can indicate how much energy is available for biomass production of top consumers like fish (Sommer et al. 2002). Therefore, investigations on the factors impacting on the transfer efficiency of carbon in food webs are needed in the future.

5.4.2. Coastal ecosystem changes based on network analysis

The energy flow across the trophic network (total system throughput, TSTP; 1998 and 1841 $t{\cdot}C{\cdot}km^{-2}{\cdot}year^{-1}$ in the 1990s and 2000s, respectively) we found here is higher than what Baird and Ulanowicz (1993) and Wilson and Parkes (1998) found for the Ems River in Germany (474 $t{\cdot}C{\cdot}km^{-2}{\cdot}year^{-1}$) and Bay of Dublin in Ireland (724 $t{\cdot}C{\cdot}km^{-2}{\cdot}year^{-1}$), respectively. However, the TSTP we inferred is much lower than the TSTP found for Narragansett Bay (5147.6 $t{\cdot}C{\cdot}km^{-2}{\cdot}year^{-1}$) and Chesapeake Bay (4541.5 $t{\cdot}C{\cdot}km^{-2}{\cdot}year^{-1}$) (Monaco and Ulanowicz 1997), but comparable to TSTP of the Bay of Somme in the northwest of France (2312 $t{\cdot}C{\cdot}km^{-2}{\cdot}year^{-1}$) (Rybarczyk et al. 2003). However, the differences on TSTP we found in the present study with other studies can be due to different climates (i.e. tropical vs. temperate) and ecosystem types.

In a study of coastal ecosystem of the Pearl River Estuary the TSTP declined by more than 70% from 1981 to 1998 (Duana et al. 2009). The decline found in the present study was only 8% between the studied periods. This indicated that energy flows of the entire Vietnamese coastal ecosystem changed less between the studied periods than the coastal ecosystem of the Pearl River Estuary. However, it is also noted that our studied duration was less than of that by Duana et al. (2009).

Ascendency, overhead, development capacity, the ratio of ascendancy to development capacity, Finn's cycling index, and the constraint efficiency index were higher in the 1990s than in the 2000s (Table 5.6 and Figure 5.4A, B and C). The ratio of ascendancy to development capacity of the 1990s was higher than of those of the 2000s in this study (46 and 42%, respectively, Figure 5.4A), which can be interpreted as a higher level of ecosystem development in the 1990s than in the 2000s (Ulanowicz and Norden 1990). The presented ecosystem is situated around the same level of ecological maturity as the Bay of Dublin (42%) (Wilson and Parkes 1998) but more mature than the Somme Bay (25%) (Rybarczyk et al. 2003) and Chesapeake Bay (30%) (Monaco and Ulanowicz 1997).

The higher value of the constraint efficiency in the 1990s (66%) than of those of the 2000s (60%) indicates that there is more room for the ecosystem's development in the 1990s than in the 2000s. Kones et al. (2009) compared the constraint efficiency revealed from four different food webs and found that the constraint efficiency of these four food webs varied from 52% to 68%. The constraint efficiency values found in our studies (66 and 60% in the 1990s and 2000s, respectively, Figure 5.4B) are also in the range reported by Kones et al. (2009).

The Finn's cycling index is a measure of ecosystem maturity (Allesina and Ulanowicz 2004) and recovery time (Vasconcellos et al. 1997). Duana et al. (2009) found Finn's cycling index to decrease from 9.2 to 2.7% of the total system throughflow between 1981 and 1998. Finn's cycling index was found by Chen et al. (2008) of 9.73% in the Tonkin Gulf (an ecosystem is a part of the present study) using Ecopath with Ecosim model to estimate. Kones et al. (2009) found Finn's cycling index to vary among food webs, ranging from 5 to 20%. The values of the Finn's cycling index that were found in the present study (7.8 and 6.5% for the 1990s and 2000s, respectively, Figure 5.4C) indicate a medium value compared to the previous studies. In summary, results from the network analysis can initially indicate that the Vietnamese coastal ecosystem network indices in the 1990s were higher than of those of 2000s and there may be possibilities that the coastal ecosystem in 1990s was more developed, stable and mature than in the 2000s. However, our studied period was too short and there were several assumptions and constraints behind the present results. In fact, input data of the inverse models were much referred from different sources. It is therefore important to acknowledge that our results and interpretations will be more reasonable when local independent and large-scale monitoring is conducted for better validation.

5.4.3. Data reliability and performance of inverse modelling methods

Obtaining sufficient data to develop and apply ecosystem models is challenging (Christensen et al. 2009). This challenge applies not only to Vietnam but also to almost all developing countries where fisheries monitoring systems are scarce (Pomeroy et al. 2009). In this study, we developed the inverse models using data aggregated by 18 compartments because detail knowledge on complex food web topology is currently insufficient. The selection of the a topology of food web model had substantial effects on the outcome of both the inverse method and for analysis of a system using network analysis (Johnson et al. 2009). However, Opitz (1996) examined the effects of aggregating a coral reef food-web and concluded that aggregation had no obvious effect on the information content (ascendency) of the network. Other authors have demonstrated that food web properties may be affected by aggregation (Sugihara et al. 1989, Martinez 1991), but they do not seem to have agreed on the extent of the effect. Investigation of aggregation of food web topology is therefore needed in the Vietnamese coastal ecosystem studies in the future to define if the aggregation can affect model outcomes.

Inverse modelling can estimate unknown food web flows in the ecosystem, which makes it very useful in such data limitation conditions. In this study, we used input data that were collected in the studied area and were referred from many different sources. The reference of data from outside can cause potential biases of model outcomes. However, the present inverse model using Monte Carlo approach allows to randomly selecting any values from minimum to maximum range that were referred from literature (Vezina and

Platt 1988) and this can reduce uncertainties of the model.

The inverse model results represent extrapolations of food web structure from a small set of measured flows and a larger set of constraints and mass balance considerations (Kones et al. 2009). Some criterion is needed to select the best solution. Standard criteria based on the minimization of the sum of the squares of the estimated components have been applied in geophysical and ecological inversions (Vezina and Platt 1988). However, this method can only provide a single solution and it has no robust ecological underpinning (Kones et al. 2006). In this study, we used alternative approach applying the Monte Carlo procedure to obtain a range of possible solutions (3000 solutions). As indicated in Kones et al. (2006) that food web flows obtained by the Monte Carlo simulation may be a better approach to determine the most likely flow estimate and its associated uncertainty.

This study is one of the first attempts to develop an ecosystem tool for EAFM in Vietnam. Previously, only single-species management was applied for Vietnamese fisheries management. Single-species management was based on the assumption that stocks can be viewed out of the context of their role in the ecosystem, that density dependence is the main regulating factor in population dynamics (Magnusson 1995). Therefore, single-species management tools take no account of the role of the stock as it interacts with other species or the population dynamical processes (FAO 2008). Nevertheless, single-species models are still the dominant tools in the world to provide timeous and scientific advice regarding the management of commercially valuable stocks (Plaganyi 2007). Therefore, it is important that modellers have a good understanding of both single-species and ecosystem approaches.

Chapter 6: An integrated food web model to analyse the impact of fisheries management scenarios on the coastal ecosystem of Vietnam

Redrafted from

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6. Chapter 6: An integrated food web model to analyse the impact of fisheries management scenarios on the coastal ecosystem of Vietnam

Abstract

In this chapter a model for the coastal marine ecosystem of Vietnam was developed to evaluate interactions between fisheries and the food web, using fisheries data from 2000 to 2005. We selected this period due to data availability of the studied area. Comparing the maximum trophic level of fish estimated by the model (4.195) and the mean trophic level of the catch (3.712) indicates that fisheries have been harvesting high trophic level species. Using the model, the present study found that maintaining the fishing effort at the 2000-2005 level puts the coastal marine resources at risk as the biomasses of ten out of twelve stocks decline by 5 to 20% in a 15 years period. A 20% fishing effort reduction of fish and shrimp trawling or gillnet and purse seine fishing still resulted in 10% biomass reductions of several key functional groups. Reducing fishing effort for all fisheries by 10% increased the biomass of almost all groups in the ecosystem up to 14% (large demersal fish). Meeting social and economic, but not ecological constraints required an increase from 4 to 8.5-fold in fishing effort and resulted in the collapses of sea turtle, tuna, small pelagic fish and cephalopods. When only meeting ecological constraints, fishing efforts reduced for four out of the eight fisheries, e.g. a 95% reduction was recommended for the gillnet fishery. A trade-off scenario indicated that achieving economic, social and ecological goals was possible by four-fold increase of traditional small-scale fisheries (e.g. handline), combined with 40 and 45% reduction of purse seine and fish trawl fisheries, respectively.

6.1. Introduction

At the end of last century, fisheries management used single-species stock assessment methods to quantify fish stocks (Caddy and Cochrane 2001). Unfortunately, this approach had – from an ecological perspective – strong limitations and shortcomings as ecosystems are composed of multiple species and often multi-gear fisheries are used (Coll et al. 2006, Griffiths et al. 2010). Recently, ecosystem approaches have received considerable attention by fishery managers and scientists due to its potential to support comprehensive management decisions (Murawski 2007, Griffiths et al. 2010). To implement such approaches, ecosystem models need to be developed that integrate harvesting by fisheries and community dynamics (Christensen and Walters 2004, Diaz-Uribe et al. 2007). Three different multi-species models can be distinguished: (1) dynamic biomass models (Spencer and Collie 1995); (2) age-structured models such as Multi-Species Virtual Population Analysis (MSVPA) (Magnusson 1995); and (3) trophic mass-balance models such as Ecopath with Ecosim (EwE) (Christensen and Pauly 1992, Walters et al. 1997, Christensen and Walters 2004). The dynamic biomass models acknowledge the time lags between biomass removal by fishing and biomass growth (Hilborn and Walters 1992). These models try to explain changes in an abundance index (normally catch per unit effort) as a function of biomass removal by fishing, the biomass in the previous time period and growth. MSVPA is a technique that uses commercial fisheries catch-at-age and fish stomach-content data to estimate both the past fishing mortalities and the predation mortalities on some of the major fish species of interest (Sparre 1991). However, disadvantages of the dynamic biomass and MSVPA models are that they provide an incomplete picture of ecosystem processes and dynamics (Garrison and Link 2005), and they are not suitable for data-poor areas (Diaz-Uribe et al. 2007).

The EwE approach, which is based on trophic mass-balance modelling, was designed for straightforward construction, parameterization and analysis of ecosystem models (Polovina 1984, Christensen and Pauly 1992). In addition, the capacity of the approach to quantify structural and functional aspects of ecosystems has been evaluated and applied in data-poor areas (Whippe et al. 2000). The EwE model can also address uncertainty using a resampling routine included to accept input probability distributions for the biomasses, consumption and production rates, catch rates, and diet compositions (Christensen and Walters 2004). The EwE can be used to explore optimal harvesting strategies by examining the ecosystem effects of fishing together with other fisheries management aspects such as economic, social constraints (Christensen and Walters 2004). These evaluations of fisheries management decisions from a community or ecological viewpoint are also considered to be essential for the long-term success of fisheries (Christensen and Pauly 2004).

Landings of Vietnam's marine fisheries have increased rapidly since the 1990s. The total estimated fisheries catch was only about 700,000 tons in 1990 but reached more than 2 million tons in 2012 (DECAFIREP 2013). The fisheries sector is significantly contributing to total Gross Domestic Product and national economic growth and is considered as a source of poverty alleviation and food security in Vietnam (Han 2007, Armitage and Marschke 2013). However, during the past years, fisheries scientists and managers in Vietnam have questioned the sustainability of the intensive fishing activities. According to recent assessments, catches have by far exceeded the maximum sustainable yield in the coastal waters of Vietnam and many marine fish stocks have been seriously reduced (Pomeroy et al. 2009), as demonstrated by declining catches per unit effort from the 1980s onwards. The average catch per horsepower (HP) estimated in the 1980s was around 1.1 tons·HP^{-1} and this number was reduced to about 0.35 tons·HP^{-1} in recent years (Anh et al. 2014b).

Increasing activity of small fishing boats has been suggested as a possible cause for the observed depletions of the coastal marine resources in Vietnam (Pomeroy et al. 2009). The total number of small fishing boats with the capacity below 50 HP along the Vietnamese coast was estimated at approximately 90,000 in 2010, while in the 1990s this was only around 50,000 (DECAFIREP 2013). Increased fishing intensity at high tropic levels can cause the food web to collapse (Pauly et al. 1998, Bhathal and Pauly 2008, Freire and Pauly 2010).

In 2010, a Vietnamese fisheries development strategy has been approved to safeguard Vietnam's marine resources (VFDS 2010). The objective of this strategy was to reduce fishing pressure in the coastal areas while intensifying offshore activities. The coastal marine ecosystem is defined from the shore to 24 nautical mile with total area of about 140,000 km^2 which is less than 30 m deep in the Northern and the Southern region and less than 50 m deep in the Central region of Vietnam. Coastal fishing activities, such as fish trawling, were considered most harmful for the coastal ecosystems and hence their reduction was identified as a priority. Decision-making tools for Vietnam's coastal management were based on single stock assessments that were inherently unable to understand effects at the ecosystem level. In this study, an ecosystem model was developed to simulate the coastal ecosystem in Vietnam using field data gathered between 2000 and 2005. A first objective of this study was to use this newly developed model to increase understanding of the coastal ecosystem's trophic structure and status, and the relationship with fishing activities. A second objective was to use this model to evaluate the impact of alternative management scenarios on the Vietnamese coastal ecosystem. To this end, the study used eight management scenarios evaluated by three different predator-prey controls. First, we simulated four fishing scenarios representing current management strategies. Next, we tested four other management scenarios that combined ecological, economic and social constraints to derive the desired fishing effort reduction. Relative biomass changes (end vs. start biomass) were calculated from these management scenarios and used to evaluate the Vietnamese coastal ecosystem status after 15 years. Finally, suitable management scenarios were proposed for the Vietnamese coastal ecosystem management.

6.2. Methods

6.2.1. Modelling approach and structure

We constructed a mass-balance model of the coastal ecosystem in Vietnam using the Ecopath with Ecosim (EwE) software (www.Ecopath.org). EwE was designed for construction, parameterization and analysis of aquatic and terrestrial ecosystems (Christensen and Walters 2004). We grouped species into ecological groups (Table 6.1), based on similarities in their ecological and biological features (e.g., feeding, habitat, mortality), and the importance of the species for fisheries (Coll et al. 2006). For each group, model parameters (e.g. mass-specific rates of production (P/B), relative food consumption (Q/B) and biomass (B)) were set to the average of the parameter values across all species within the groups. When there were few values to calculate the mean values, a species with sufficient data was selected to calculate parameters.

6.2.2. Parameter estimation

Model's parameters were based on local data. If such data were not available, we searched for data from the same region. If no local or regional data were available, we considered data from similar ecosystems or from Fishbase (Froese and Pauly 2006) (Appendix 6.1). We intended to develop the model to represent the food web from the lowest up to the highest trophic level using a functional group approach. However, due to data availability, we could not develop a model with a highly detailed representation of all taxa in the food web. The basic parameters of 18 ecological groups derived from local data, literature or estimated by Ecopath were indicated in the Table 6.4 and Appendix 6.1.

Table 6.1. List of species included in the coastal ecosystem model in Vietnam.

No.	Ecological Groups	Included species/taxa
1	Mammal	*Sousa chinensis*
2	Sea turtle	*Deemochelys coriacea, Caretta olivacea, Chelonia mydas, Chelonia caretta*
3	Large predators	Carcharinidae, Scombridae (*Scomberomorus commerson* and *Scomberomorus guttatus*)
4	Tuna	*Katsuwonus pelamis*, *Euthynnus affinis*
5	Medium pelagic fish	Trichiuridae, Stromateidae
6	Small pelagic fish	Clupeidae
7	Other pelagic fish	Carangidae (*Atule mate, Alepes kalla, Alepes djedaba, Megalaspis cordyla, Scomberoides spp., Selaroides leptolepis, Seriolina nigrofasciata*, Theraponidae, *Lactarius lactarius* and *Selar crumenopthalmus*, Caesionidae, Scombridae (*Rastrelliger spp.*), *Decapterus maruadsi*, *D. russelli*, *D. kurroides*
8	Anchovy	*Stolephorus commersoni, Encrasicholina heteroloba, E. punctifer, Stolephorusindicus, E. devisi*
9	Cephalopods	Includes squids (*Loligo spp.*), cuttlefish (*Sepia spp.*) and octopus (*Octopus spp.*)
10	Large demersal fish	Ariidae, Cepolidae, Cynoglossidae, Drepannidae, Fistularidae, Gobiidae, Holocentridae, Meneidae, Monocanthidae, Nemipteridae, Muraenidae, Ostraciidae, Paralichthidae, Pegasidae, Platycephalidae, Plotosidae, Polynemidae, Priacanthidae, Rhinobathidae, Sciaenidae, Syngnathidae, Synodontidae, Tetraodontidae, Lethrinidae, Serranidae, Scorpaeinidae
11	Reef fish	Chaetodontidae, Labridae, Pomacentidae
12	Other demersal fish	Bothidae, Cynoglossidae, Gerreidae, Haemulidae, Mullidae, Nemipteridae, Presttodidae, Siganidae, Sillaginidae, Soleidae, Sparidae, Teraponidae, Sciaenidae
13	Shrimp	Penaeidae, Palaemonidae, Scyllaridae, Soleidae, Solenoceridae, Squillidae
14	Crustaceans	Portunidae, Palinuridae
15	Zoobenthos	Polychaeta, Coelenterata, Echinodermata, Porifera
16	Zooplankton	Copepoda, Chaetognatha
17	Phytoplankton	Macroalgae; Algae; Phytoplankton
18	Detritus	Particulate and dissolved organic matter

6.2.2.1. Biomass data

Biomasses of all demersal fish groups, shrimps, cephalopods and other crustaceans were estimated from fish and shrimp trawl surveys conducted between 2000 and 2005 as described in the Chapter 5 (RIMF 2005a). Biomasses were estimated based on the swept-area method (Gunderson 1993) and assumed catchabilities of 0.5 and 0.64 for fish and shrimp trawl fishery, respectively. Biomasses of large, medium, and small pelagic fish and anchovy species were obtained from acoustic surveys conducted from 2003 to 2005 (RIMF 2005c). The method assumed that there was a proportional relationship between acoustic backscattering from an echosounder and fish biomass (Foote 1983, 1987). A total of 84 acoustic transects (43 transects of 60nm and 41 transects of 30nm) were conducted within this survey (RIMF 2005c). Biomasses of mammals, sea turtles, zoobenthos, zooplankton and phytoplankton were missing in these surveys and thus they were estimated using the Ecopath model.

6.2.2.2. Diet matrix

Diet compositions were poorly investigated in Vietnam's coastal ecosystem. Only a few studies on feeding behaviour of fish were conducted in this area, most of which are unpublished. Therefore, for some groups, we relied on regional data from the South China Sea (i.e. large predators, large demersal fish, other demersal fish, reef fish, shrimp and crustaceans) (Chen et al. 2008a, Wang et al. 2012), and the Western and Central Pacific Ocean (i.e. mammals, sea turtle and tuna group) (Griffiths et al. 2010) to estimate diet compositions. For large, medium and small pelagic fish, and anchovy, we used diet data from Fishbase (Froese and Pauly 2006) (Appendix 6.2).

6.2.2.3. Catch data

Total catch used in this chapter was extracted from reconstructed catch as described in the Chapter 3, limited on inshore and coastal fisheries only. Since the EwE model does not allow to put unidentified ecological groups as input data into the model, catch data of "other groups" in Chapter 3 were separated using the same catch rates as in Table 3.2. Average catch by the fisheries and by ecological groups was calculated by averaging of every year (Table 6.2).

Table 6.2. Average catch (t·km^{-2}·year^{-1}) for the different fisheries considered in the coastal region for the period 2000 to 2005 (landings to be considered as catches since there are no discards in the Vietnam's fisheries). Other (mixed gear) includes lift net, stick net and traditional fisheries as defined in the Chapter 3 but currently is only focussing on the coastal fisheries.

Group name	Gillnet	Fish trawl	Shrimp trawl	Purse seine	Traps	Fish handline	Squid handline	Other
Sea turtle	0.30							
Tuna	0.55			0.01				
Large predators	1.34	0.01		0.32				
Large demersal fish		1.38	0.20		0.02			
Other demersal fish		0.90	0.32		0.02	0.02		
Reef fish			0.29					0.30
Large pelagic fish	0.20	0.05		0.03		0.01	0.01	
Medium pelagic fish	0.30	0.02		0.01			0.02	
Small pelagic fish	0.50			0.40		0.01		0.04
Anchovy				0.05				
Cephalopods		0.03	0.02	0.02			1.11	0.09
Shrimp		0.01	1.25					
Crustaceans		0.03	0.05		0.19			

6.2.2.4. Other parameters

6.2.2.4.1. The production/biomass ratio (P/B)

For reef fish, medium pelagic fish and anchovy, the production/biomass ratio (P/B) was calculated by assuming the P/B to be equal to the total instantaneous mortality (Z), where Z = M + F (Allen 1971) with M and F the natural and fishing mortality of the exploited species, respectively. M was calculated using the empirical formula of Pauly (1980):

$$M = K^{0.65} \cdot L_{\infty}^{-0.29} \cdot T^{0.463} \qquad \text{(Eq. 6.2)}$$

where L_{∞} is the asymptotic length, K is the growth coefficient and T is the average sea surface temperature during the period studied. The average temperature in this study was observed by simultaneous investigations in the studied period and was 24.5ºC

(RIMF 2005a). For other groups (14 groups), P/B was collected from the literature (see Appendix 6.1).

6.2.2.4.2. The consumption/biomass (Q/B)

For tuna, reef fish and anchovy groups, the consumption/biomass (Q/B) ratios (i.e. relative food consumption) were computed using the predictive model of Palomares and Pauly (1989):

$$\log Q/B = 7.946 - 0.204 \cdot \log W_\infty - 1.967/1000 \cdot T + 0.083 \cdot A + 0.532 \cdot h + 0.398 \cdot d \quad \text{(Eq. 6.3)}$$

where W_∞ (gram) is the weight at infinite length (L_∞), calculated as $W_\infty = a \cdot (L_\infty)^b$, a and b are two constants, T is temperature (Kelvin), A is the aspect ratio of the caudal fin indicating metabolic activity and expressed as the ratio of the square of the height of the caudal fin and its surface area, as obtained from Fishbase (Froese and Pauly 2006), and h (1 for herbivores, and 0 for detritivores and carnivores) and d (1 for detritivores, and 0 for herbivores and carnivores) express food type. Data to estimate the length-weight (L-W) relationship were available from previous studies in Vietnam (RIMF 2005a, RIMF 2005b). We gathered the Q/B data for other groups (13 groups) from the literature (see Appendix 6.1).

6.2.3. Model balancing and network analysis

Diet composition and biomass were considered as the most uncertain parameters in the present model and thus their values were adjusted by 10% until the model was balanced. After the model balanced, we used the Ecoranger routine (Christensen and Walters 2004) to allow the probability distribution to be specified for each input variable, and used a Monte Carlo simulation of samples from the input distributions to generate the probability distributions of the output variables (Christensen and Walters 2004). The production/consumption ratio is one of the outputs of Ecopath model and is called the gross food conversion efficiency (GE). The model was considered as balanced when: 1) estimates of ecotrophic efficiency (EE) were < 1; 2) values of P/Q (the GE) were between 0.1 and 0.3 (Christensen and Walters 2004); and 3) values of the output (i.e. biomass and ecotrophic efficiency in this study) were consistent with literature data in related coastal waters (e.g. of Chen et al. 2008a).

The uncertainty on the input parameters was accounted for using a pedigree routine (Christensen and Walters 2004). For each input value, a description of the data was made and confidence was evaluated based on the data's origin, e.g., sampling, approximate or indirect methods, other models or the literature. The uncertainty on B, P/B, Q/B, Y, and diet composition was quantified using a pedigree index between from 0 (lowest confidence) and 1 (highest confidence). Then, based on these pedigree index values, an overall pedigree index (P) was calculated as: $P = \sum_{i=1}^{n} \frac{I_{ij}}{n}$ (Eq. 4)

where I_{ij} is the pedigree index for model group i and parameter j, and n is the total number of modelled groups (Christensen and Walters 2004).

Direct and indirect trophic interactions within the ecosystem were analysed by means of the mixed trophic impact (MTI) routine (Ulanowicz and Puccia 1990), allowing the quantification of direct and indirect trophic impacts among groups. Such impacts are quantified using the MTI, which ranges from −1 to 1 and can be compared among ecological groups (Christensen and Walters 2004). The MTI values indicate how a small increase in the biomass of one group (the impacting group) affects the biomass of other groups (the impacted groups), including the fisheries. The MTI has a positive value if the impact is beneficial to the impacted group and is negative if impact is negative.

Trophic levels in Ecopath and in other modelling approaches as well (De Laender et al. 2010a) are not necessarily integers as proposed by Lindeman (1942), but can be fractional as suggested by Ulanowicz (1995). A routine assigns trophic levels of 1 to producers and detritus and to consumers it assigns a trophic level of 1 plus the weighted average of the preys' trophic level. Here, detritus represents non-assimilated food (egestion), decaying phytoplankton matter that is not consumed by grazing herbivores, and dead animals due to non-predatory mortality. The fishery is assigned a trophic level corresponding to the average trophic level of the catch, i.e. without adding 1 as is done for ‘ordinary’ predators (Christensen and Walters 2004).

Overall properties of the system such as the total system throughput and gross efficiency were calculated. The total system throughput was estimated as the sum of four components of the flows (i.e., total consumption + total export + total respiration + total flows to detritus) (Hirata and Ulanowicz 1984) and was calculated by the Ecopath model (Christensen and Walters 2004). The gross efficiency was calculated as the ratio of the catch to the primary production. Other network indices such as the system omnivory index (SOI) and the connectance index (CI) were also calculated. The SOI is a measure of how the feeding interactions are distributed between trophic levels (Odum 1969, Christensen and Pauly 1992) and the CI indicates the ratio of actual links between ecological groups to the number of theoretically possible links (Christensen and Walters 2004). The CI and the SOI are two indices used to describe the maturity and complexity of an ecosystem and are expected to be higher in a more mature and complex system.

6.2.4. Ecosim model scenarios

The balanced Ecopath model was used to explore the consequences of different fisheries management scenarios on the coastal ecosystem of Vietnam using Ecosim.

First, we simulated four fishing scenarios that are partly based on fisheries management strategies approved by the Vietnamese Government (VFDS 2010). These scenarios were performed by varying the fishing effort of preselected fisheries. Fishing effort was quantified based on the amount of vessels implementing a specific fishery technique. These scenarios were: no change in the fishing effort (Scenario 1), a 10% decrease of fishing effort for all fishing fleet, representing the precautionary principle for fisheries management (Williams 2010) (Scenario 2), a 20% decrease of the fishing effort for fish and shrimp trawl fisheries only (Scenario 3), and a 20% decrease of fishing effort for gillnet and purse seine fisheries only (Scenario 4). Scenarios 1, 3 and 4 are based on the Vietnamese fisheries development strategy (VFDS 2010).

Next, we tested four other scenarios that accounted for economic (Scenario 5), social (Scenario 6), and ecological criteria (Scenario 7), and one scenario that accounted simultaneously for all three criteria (Scenario 8). These scenarios were analysed using a weight value of 1 for the criterion to be optimized and a low non-zero value (0.0001) for the criterion not to be optimized (Table 6.3) (Christensen and Walters 2004). The Davidson-Fletcher-Powell nonlinear estimation method was used to change the fishing effort until the criteria were fulfilled. The economic criterion was defined as the net economic value of the fisheries and was calculated as the economic value of the total landed catch minus the total operating cost over the simulation period (MacKenzie 1983), with a discount rate (reduction rate of catch price) of 4% as a default value (Christensen and Walters 2004). The fixed and effort-related costs of all fleets (the operating costs) were equal to 35 and 20% of the total value, respectively (Chen et al. 2008b). The social criterion is defined by the ratio of jobs (employment) to the landed catch value (catch) for each fishery and equal to the number of fishers per the catch value (Christensen and Walters 2004). We assumed that lift net, stick net and traditional fisheries (defined as “other” fisheries in the Table 6.2) and traps were less efficient and thus needed more people per unit of catch. Therefore, we set the job/catch ratio of the “other” fisheries and the traps to 5 and 4, respectively. For fish handline and squid handline this value was set to 2; for gillnet, shrimp and fish trawl, and purse seine this was 1 job/catch. The ecological criterion used the inverse of the P/B ratio for each group, and represented size and life spans (Zetina-Rejon et al. 2004) (Table 6.3). This means that species or groups with longer life spans and larger body sizes are assumed to have more ecological important roles in the ecosystem.

Table 6.3. Relative weight assigned to each value component under four policy scenarios with three independent scenarios of three criteria Economic, Social and Ecosystem and a combined scenario.

Value component	Relation weight values for scenarios			
	Economic	Social	Ecological	Combination of three criteria
Net economic value	1	0.0001	0.0001	1
Social value (employment)	0.0001	1	0.0001	1
Ecosystem structure	0.0001	0.0001	1	1

To account for the uncertainty on the predator-prey interactions, every scenario was performed three times, once assuming bottom-up control, once assuming wasp-waist control, and one assuming mixed control. These three types of predator-prey interactions were introduced by adjusting the vulnerability parameters according to Coll et al. (2006) as following:

1. Setting values of v = 2, the default value of Ecopath model for all groups to represent the mixed flow control (neither top-down nor bottom-up control) (Shannon et al. 2000).

2. Setting values of v = 1 of phytoplankton, zooplankton and zoobenthos groups to their predators to describe the bottom-up flow control.

3. Setting values of $v \gg 1$ of the prey to intermediate trophic level groups (pelagic fishes and anchovy) (top-down control of these groups on their prey) and v = 1 of pelagic fishes to their predators (bottom-up control of pelagic fishes on their predators) to represent warp-waist control within ecological groups.

Dynamic simulations were run for five years without a change from the baseline level of the Ecopath model in order to assure stable initial conditions. After that, a 15-year simulation was applied for every scenario. We made eight scenarios for two management strategies multiplied to three predator-prey control scenarios. So there were total 24 different simulations performed (Figure 6.1). The results from the simulations were then used to compare relative biomass and fishing effort changes for the scenarios without consideration and consideration on socioeconomic and ecological aspects, respectively.

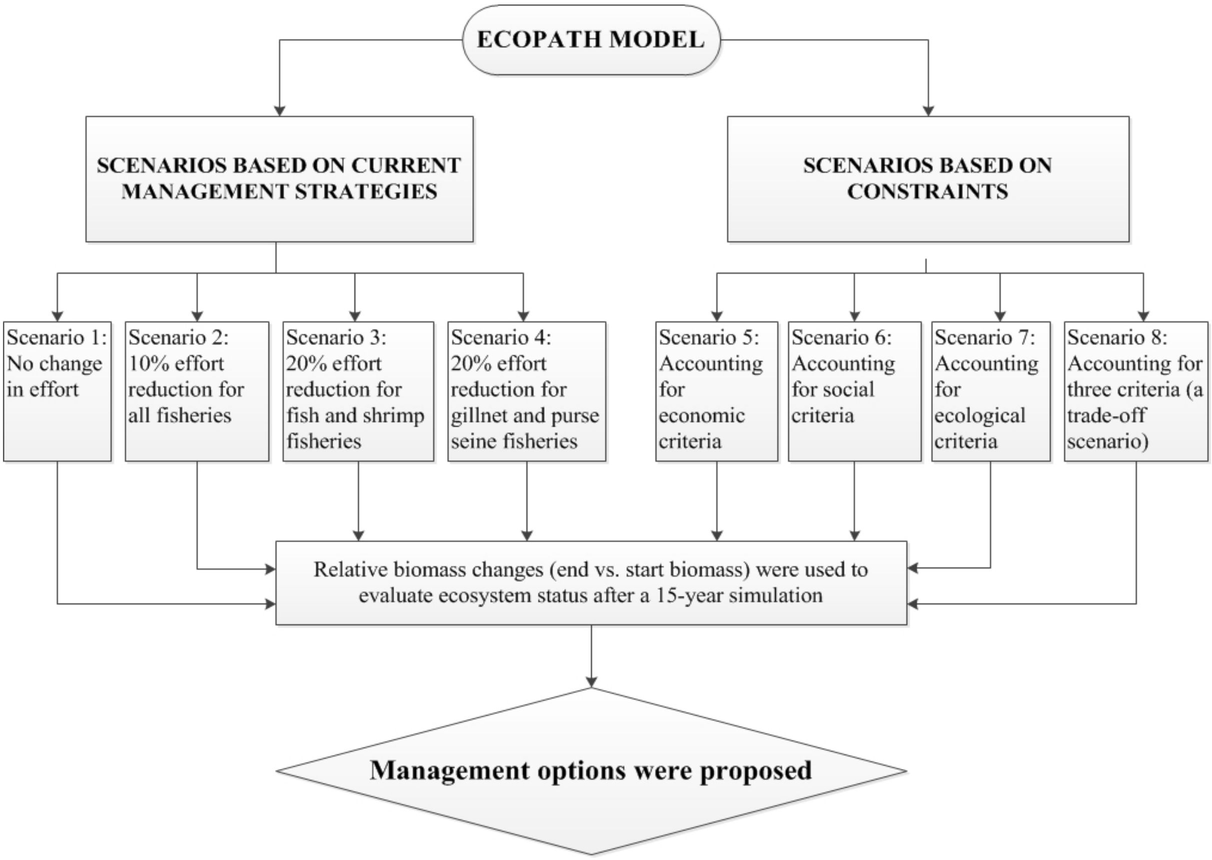

Figure 6.1. Schematic diagram indicating the scenarios considered. Every scenario was performed three times: once assuming bottom-up control, once using wasp-waist control, and once assuming mixed control.

6.3. Results

6.3.1. Ecopath model

Ecotrophic efficiencies (EE) were relatively high for most ecological groups except for detritus (EE=0.01). The production/consumption ratios (P/Q) varied from 0.020 to 0.293 and were within the range expected from thermodynamic limits (Christensen and Walters 2004). Trophic levels estimated by the model varied from 1.00 for primary producers and detritus to 4.503 and 4.195 for sea mammals and large predators, respectively (Table 6.4). A graphical representation of the final balanced model indicating prey-predator relationships and the trophic levels of the ecological groups is shown in Figure 6.2.

Table 6.4. Basic input and output (in bold) parameter values of the Ecopath model for the coastal ecosystem in Vietnam. TL is the trophic level; B is Biomass, P/B is production rate, Q/B is the consumption rate, EE is the ecotrophic efficiency, P/Q is production per consumption ratio.

No.	Ecological groups	TL	B ($t \cdot km^{-2}$)	P/B ($year^{-1}$)	Q/B ($year^{-1}$)	EE	P/Q	(1/P/B)
1	Mammal	**4.503**	**0.073**	0.050	2.560	0.100	**0.020**	**20.000**
2	Sea turtle	**3.532**	**0.197**	0.190	3.500	0.114	**0.054**	**5.000**
3	Tuna	**3.938**	0.693	1.200	4.100	**0.920**	**0.293**	**2.041**
4	Large predators	**4.195**	1.850	0.300	4.120	**0.733**	**0.073**	**3.333**
5	Large demersal fish	**3.326**	1.520	0.900	5.110	**0.848**	**0.176**	**1.111**
6	Other demersals fish	**2.669**	1.284	2.200	8.600	**0.638**	**0.256**	**0.455**
7	Reef fish	**3.247**	0.833	1.556	14.968	**0.572**	**0.104**	**0.643**
8	Large pelagic fish	**3.414**	1.560	1.450	6.300	**0.760**	**0.230**	**0.690**
9	Medium pelagic fish	**2.907**	0.948	2.800	8.560	**0.896**	**0.270**	**0.357**
10	Small pelagic fish	**2.910**	2.404	3.350	17.600	**0.895**	**0.190**	**0.299**
11	Anchovy	**3.191**	0.590	3.380	15.820	**0.722**	**0.214**	**0.296**
12	Cephalopods	**3.212**	1.335	3.100	16.640	**0.910**	**0.186**	**0.323**
13	Shrimp	**2.579**	1.565	3.800	16.380	**0.616**	**0.280**	**0.186**
14	Crustaceans	**2.421**	3.529	5.900	26.900	**0.913**	**0.219**	**0.169**
15	Zoobenthos	**2.132**	**8.769**	6.570	27.400	0.740	**0.240**	**0.152**
16	Zooplankton	**2.053**	**8.356**	36.000	186.000	0.650	**0.194**	**0.028**
17	Phytoplankton	**1.000**	**157.501**	368.000		0.624		**0.003**
18	Detritus	**1.000**	163.000			**0.010**		

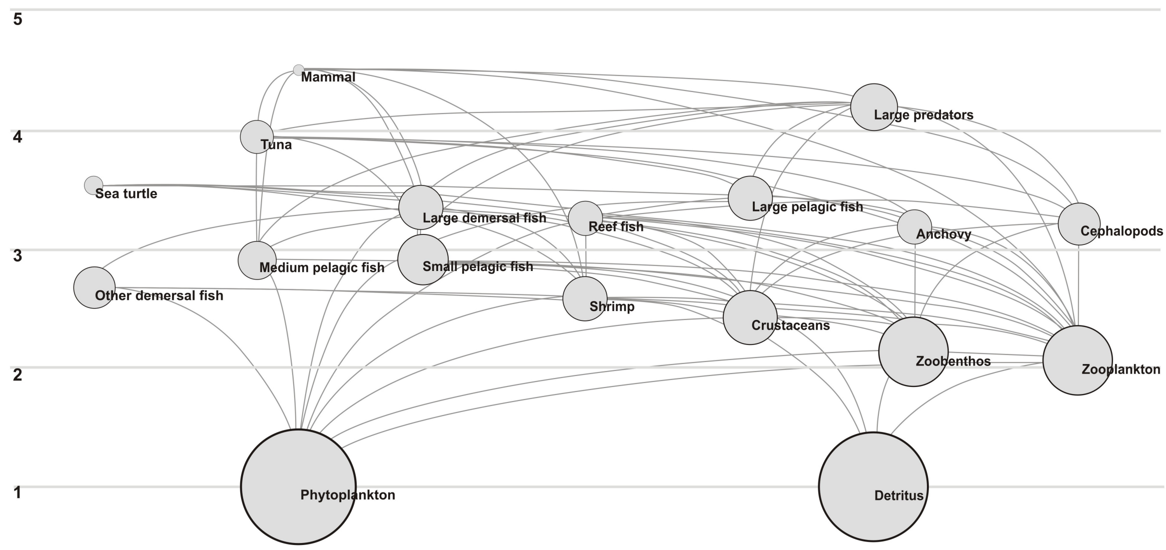

Figure 6.2. A food web diagram of the coastal ecosystem in Vietnam. Flows are in t·km^{-2}·year^{-1}. The surface area of the circles is proportional to the biomass of groups. The components of the system are structured along the vertical axis according to their trophic level defined as 1 for primary producers and detritus and as 1 plus the weighted average of the prey's trophic level for consumers.

6.3.2. Summary statistics

The total system throughput (i.e. total consumption + total export + total respiration + total flows to detritus) was 17,027 t·km^{-2}·year^{-1} (Table 6.5). Of this, 29.7% represented consumption by predators, 18.9% was exported outside the system, 24.8% was lost via respiration, and 26.6% flowed to detritus. The gross efficiency and the mean trophic level of the catch of the coastal system in the present study was 0.0018 and 3.712, respectively (Table 6.5). The system omnivory index and the connectance index were calculated to be 0.146 and 0.298, respectively. The pedigree index of the present model estimated was 0.32 (Table 6.5).

Table 6.5. System statistics of the coastal ecosystem in Vietnam.

Parameters	Value	Unit
Total consumption	5,053	$t \cdot km^{-2} \cdot year^{-1}$
Total exports	3,212	$t \cdot km^{-2} \cdot year^{-1}$
Total respiratory flows	4,229	$t \cdot km^{-2} \cdot year^{-1}$
Total flows into detritus	4,533	$t \cdot km^{-2} \cdot year^{-1}$
Total system throughput	17,027	$t \cdot km^{-2} \cdot year^{-1}$
Total production	5,373	$t \cdot km^{-2} \cdot year^{-1}$
Calculated total net primary production	5,796	$t \cdot km^{-2} \cdot year^{-1}$
Net system production	5,673	$t \cdot km^{-2} \cdot year^{-1}$
Total catches	10.43	$t \cdot km^{-2} \cdot year^{-1}$
Total biomass (excluding detritus)	193	$t \cdot km^{-2}$
Gross efficiency (catch/net primary production)	0.0018	
Mean trophic level of the catch	3.712	
Connectance Index	0.298	
System Omnivory Index	0.146	
Ecopath Pedigree Index	0.32	

6.3.3. Mixed trophic impacts

Negative trophic impacts were found within groups, due to within-group competition for resources. This observation was most apparent for the large predator and zooplankton groups (Figure 6.3). In addition, among all groups, the large predators and large demersal fish had the largest negative impact on other groups, presumably due to their broad diet. However, this was not found for other high trophic level groups such as sea mammals and turtles, which may be due to their low biomass. Increasing fishery activities would directly and negatively impact on the target groups of the fisheries in the case of gillnet fishery on sea turtle, large predators, and pelagic fish groups. For the sea turtle group, the impact of gillnets can be interpreted as both normal fishing activities and “ghost fishing” by nets lost at sea. However, increasing fishing efforts can also positively affect low trophic level groups because fishing can reduce the biomass of higher trophic level groups and hence predation pressure (Figure 6.3).

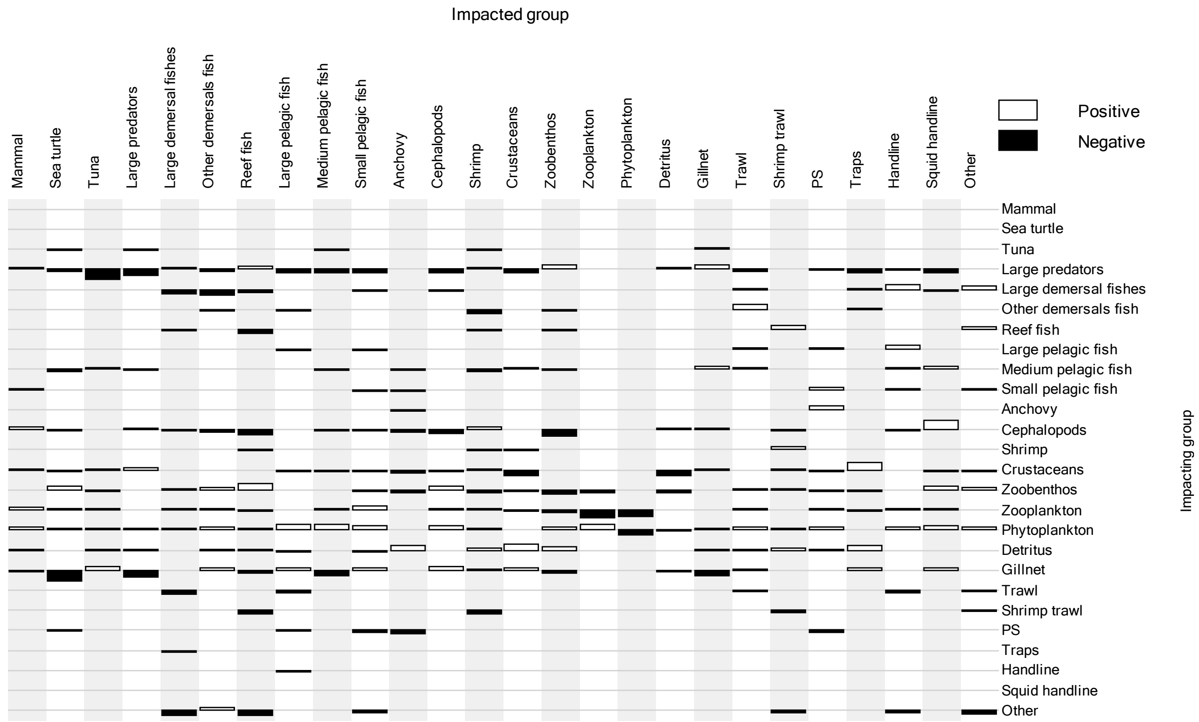

Figure 6.3. Mixed trophic impact analysis from the model. Impacted groups are placed along the horizontal axis and impacting groups are shown on the vertical axis. The bars indicate relative impacts between -1 (highest negative impact) and 1 (highest positive impact); zero means no impact. Gillnet, fish trawl (trawl), shrimp trawl, PS (purse seine), traps, fish handline, squid handline and 'other' including lift net, stick net and traditional fisheries are also considered as components of the studied ecosystem.

6.3.4. Management scenarios

6.3.4.1. Scenario 1: No change in the fishing effort

Scenario 1 resulted in 5 to 20% biomass reductions for most groups, except for the anchovy and crustacean groups (Figure 6.4A). The largest biomass reduction was noted for large demersal fish (13 to 17%, depending on the predator-prey control selected; Figure 6.4A), while pelagic fish and the top predators decreased to a lower extent (6 to 12%; Figure 6.4A). In contrast, for anchovy and crustaceans, the initial/final biomass ratio increased by 17 to 23% and 4 to 10%, respectively (depending on the predator-prey control selected; Figure 6.4A).

6.3.4.2. Scenario 2: 10% effort reduction for fishing effort

In contrast to the first scenario, the results of the second scenario indicated that the biomasses of most groups increased by 2 to 14% (Figure 6.4B). For example, under the wasp-waist control, the large demersal fish group increased by 14%, and large predator and reef fish groups increased by 9 and 10%, respectively. In case of mixed and bottom-up control, the final and initial biomass ratios of groups also increased (Figure 6.4B). Interestingly, although fishing pressure was reduced by 10% for all fisheries, the estimated anchovy biomass was lower than in scenario 1 (without fishing effort reductions), regardless of the control.

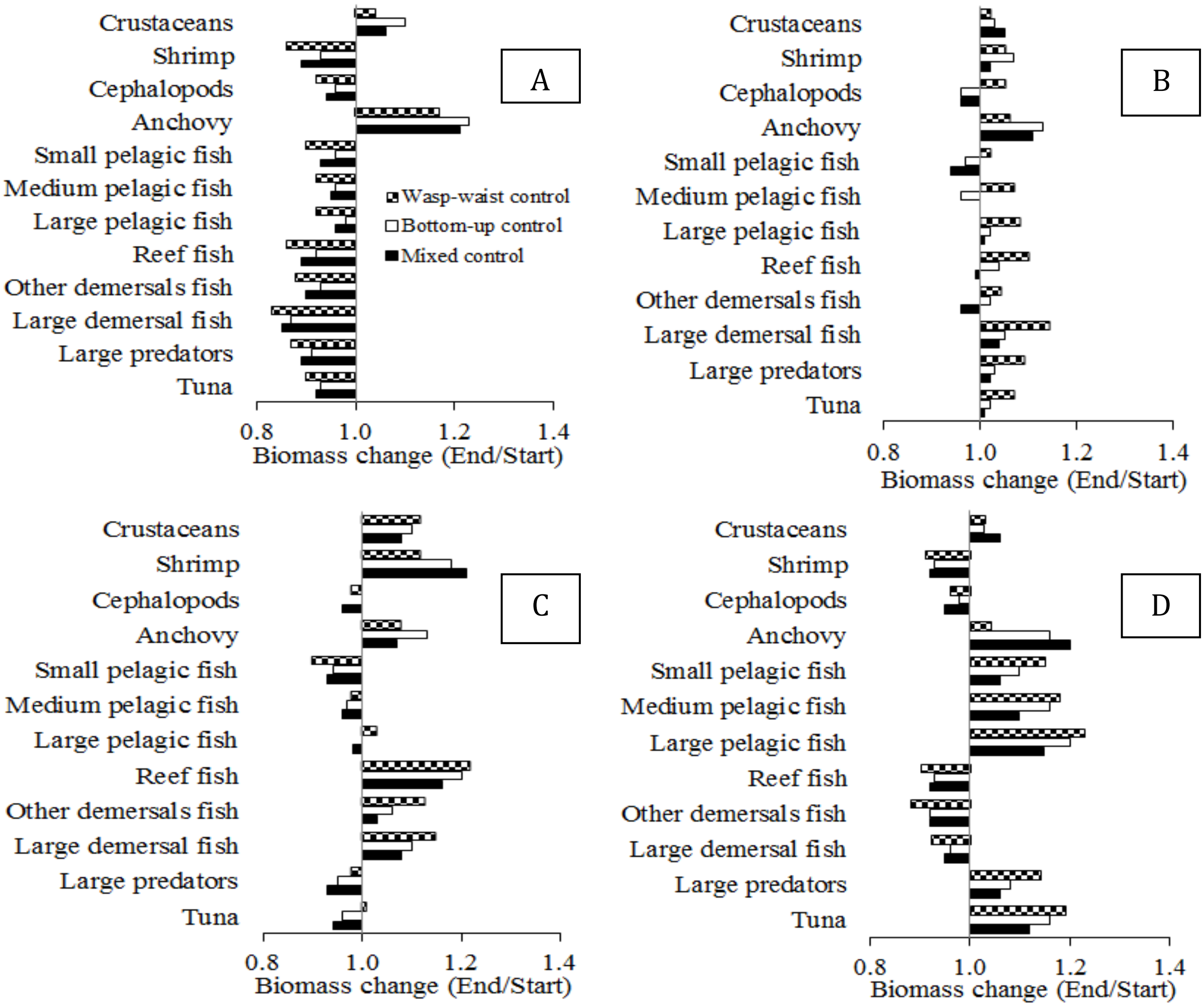

Figure 6.4. End biomass/start biomass ratio from Ecosim simulations under mixed flow control, bottom-up control and wasp-waist control under baseline fishing effort (A), a 10% decrease in fishing effort for all fisheries (B), a 20% decrease of fishing effort for fish and shrimp trawl fisheries only (C) and a 20% decrease of fishing effort for gillnet and purse seine only (D).

6.3.4.3. Scenario 3: 20% effort reduction for fish and shrimp trawl fisheries

A 20% reduction of the fish and shrimp trawl fisheries efforts increased biomasses of ecological groups that were directly impacted by these fisheries. The ecological groups that benefited most from this scenario were reef fish, shrimp, large demersal fish and crustaceans. Biomass increases of 16 to 22% (reef fish) and of 12 to 21% (shrimp) were noted (Figure 6.4C). For large demersal fish and crustaceans, biomass increases were smaller (8 to 15%).

6.3.4.4. Scenario 4: 20% effort reduction for gillnet and purse seine fisheries

Similar to scenario 3, a 20% reduction of the gillnet and purse seine fisheries efforts only caused biomass increases of ecological groups that were directly impacted by these fisheries. Tuna, large predators and pelagic fish - the most vulnerable species to gillnet and purse seine fisheries - benefited most from this scenario. Biomass of tuna increased between 12 and 19% (equivalent to 0.78 and 0.82 $t{\cdot}km^{-2}$) and that of large predators between 6 and 14% (equivalent to 1.96 and 2.11 $t{\cdot}km^{-2}$). The biomass ratio of large, medium and small pelagic fish also increased (15 to 23%; Figure 6.4D).

6.3.4.5. Scenario 5: Accounting for economic criteria

To account for economic criteria, increasing fishing effort for all fisheries was required, with the highest increase noted for gillnet (4.6 to 5.6-fold), regardless of the chosen predator-prey control (Figure 6.5A). The fishing efforts of purse seine and shrimp trawl fisheries needed to be increased to a lesser extent (1.7 to 3.7-fold, depending on the predator-prey control selected; Figure 6.5A). However, these fishing effort increases caused large changes in ecosystem structure, especially at high trophic levels. Biomass of sea turtle, tuna, and large predators declined by up to 90% or even went extinct (Figure 6.6A). Biomass of large and medium pelagic fish also declined but with a lower extent (20 to 60% depending on the predator-prey control selected).

6.3.4.6. Scenario 6: Accounting for social criteria

A substantial increase in fishing effort was needed to meet social requirements (Figure 6.5B). For the “other fisheries” (i.e. lift nets, stick nets and traditional fisheries), 7 to 8.5-fold increases were needed, while 3.7 to 5.3-fold increases for trap fishery were needed. Under this scenario, especially the biomass of the lower trophic levels was reduced. The groups exploited by small scale/artisanal fisheries such as small pelagic fish, anchovy, cephalopods, and shrimp were seriously threatened and almost depleted under warp-waist control (Figure 6.6B).

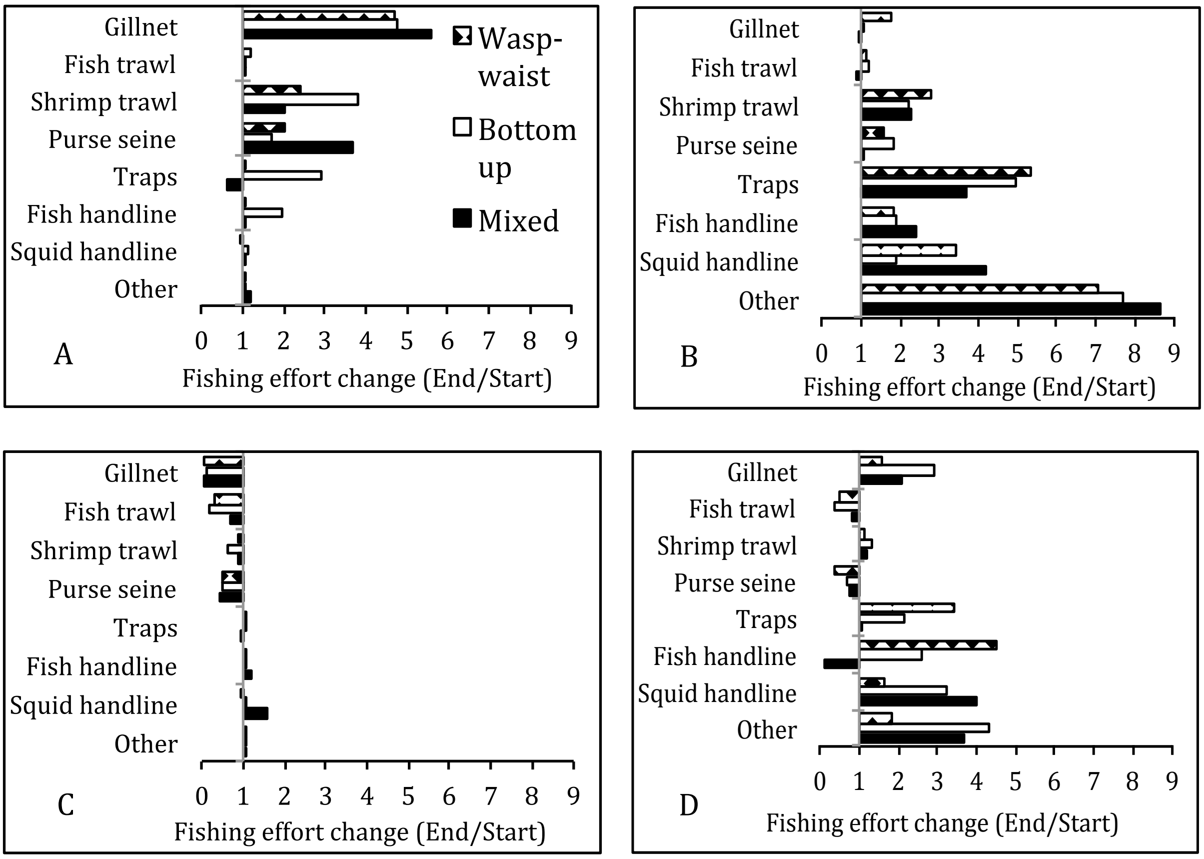

Figure 6.5. Suggested fishing effort change when meeting economic criteria only (A), social criteria only (B), ecological criteria only (C), and for a trade-off scenario (D). 'Other' denotes lift net, stick net and traditional fisheries.

6.3.4.7. Scenario 7: Accounting for ecological criteria

To safeguard the ecological structure, fishing effort reductions were required for all fishing gears that targeted high trophic levels and impacted to the seabed habitat such as gillnet and fish trawling. For instance, a reduction of 90 to 95% was recommended for the gillnet fishery (Figure 6.5C). Under this scenario, the biomass of large predators, tuna and large demersal fish increased between 1.3 to 2 times (Figure 6.6C). Biomass of large demersal fish and tuna reached to 2.93 - 3.82 and 1.04 - 1.22 t·km^{-2}, respectively, under different predator-prey controls after a 15-year simulation.

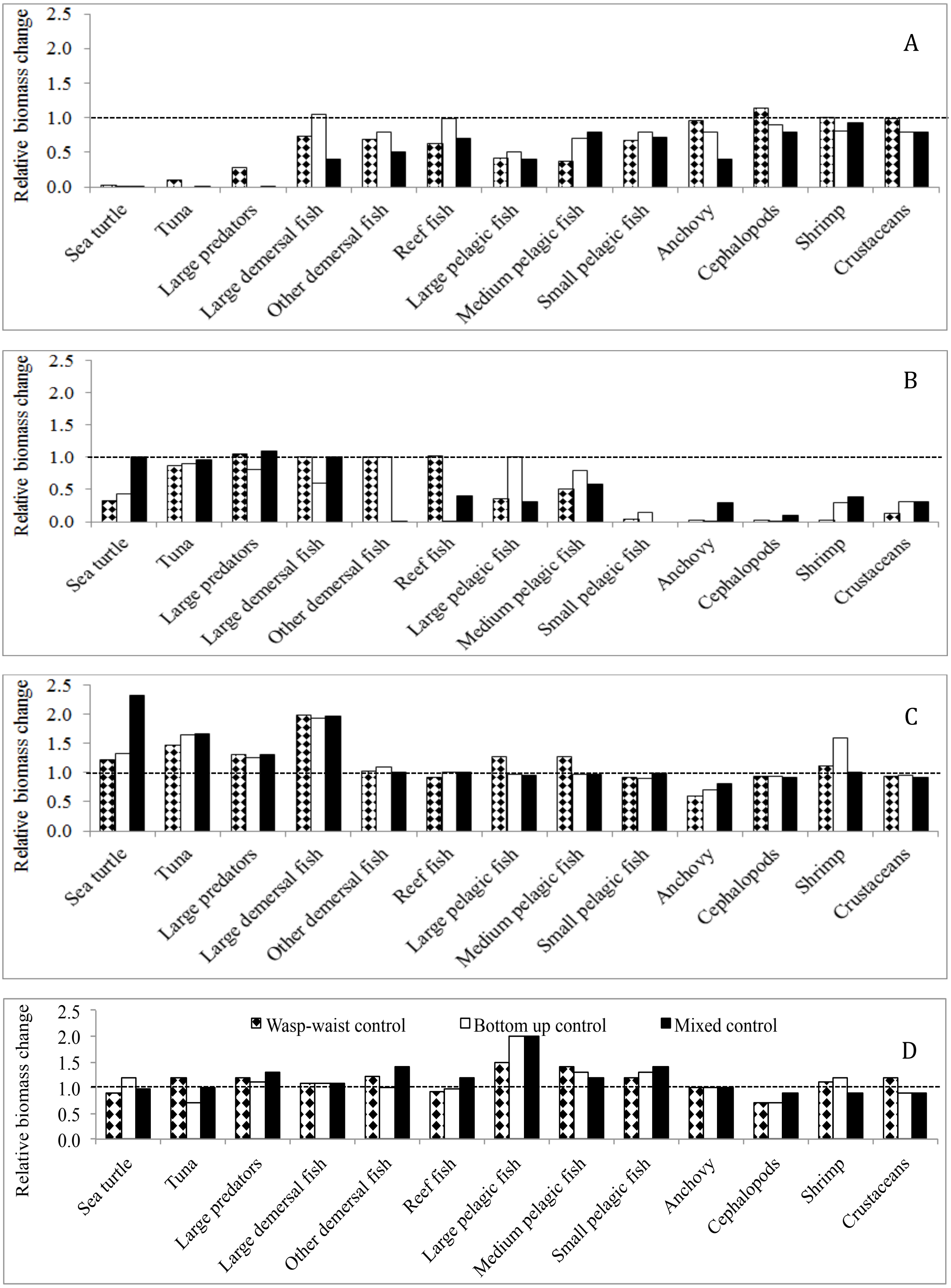

Figure 6.6. End biomass/start biomass ratio from Ecosim simulations under three predator-prey controls. A, B, C, D, and 'Other' as in Figure 6.5.

6.3.4.8. Scenario 8: Accounting for economic, social and ecological criteria

In the trade-off scenario, accounting simultaneously for economic, social, and ecological constraints, up to 4-fold increases for gillnet, fish handline, squid handline and ‘other’ fisheries was found. In contrast, a fishing effort reduction of 40 and 45% was suggested for the purse seine and fish trawl fisheries (Figure 6.5D). Under the trade-off scenario, biomass changes were less severe than for scenarios 5 and 6 (Figure 6.6D). This scenario was predicted to result in a biomass increase (1.1 to 1.5-fold) for most ecological groups (Figure 6.6D).

6.4. Discussion

EAFM has widely been recognized and considered as a fisheries management approach at the national (Canadian Government 1997, Fluharty 1999, OECD 2010), regional (Europen Commission 2004a, Europen Commission 2004b, Europen Commission 2005, Europen Commission 2008) and international (FAO 2003, FAO 2008) levels. At national level, EAFM has been implemented in United States of America (Fluharty 1999), Australia (OECD 2010), Canada (Canadian Government 1997) to manage the marine ecosystem in a broader and more holistic manner. Although there have been significantly achieved on application of EAFM in these countries, it is also noted that there are many uncertainties for example difficulties on defining disturbing sources (natural or anthropogenic sources) impacting the ecosystem to better manage these marine resources and ecosystem.

Some fisheries management measures such as fishing closed seasons and protected areas, and mesh size restrictions have been implemented in Vietnam (MOFI 2006, MARD 2011). However, these management policies have been established using only information from single-species assessments that is based on the assumption that stocks can be viewed out of the context of their role in the ecosystem. Consequently, interactions among species are not assessed and this remains a limitation for the accurate improvement of fisheries management in Vietnam. The presented model provides valuable insight into the interactions within the ecosystem and between the ecosystem and fishing activities. The total system throughput estimated in this study was 17,027 $t \cdot km^{-2} \cdot year^{-1}$ and was relatively high compared to the value for Pearl river estuary (4,799 $t \cdot km^{-2} \cdot year^{-1}$) (Duana et al. 2009) and for Beibu Gulf (8,520 $t \cdot km^{-2} \cdot year^{-1}$) (Chen et al. 2008a). The low value of ecotrophic efficiency obtained for detritus (0.01) indicates that only a small fraction of detritus biomass was consumed, whereas the rest was buried in the sediment or exported out of the system. Large predators also played an important role for energy export out of the ecosystem, because they fed on many groups in the ecosystem and were eventually removed by fishing.

The mean trophic level of the catch in the model was 3.712. The mean trophic level of the catch in the coastal ecosystem of Vietnam is higher than that of the catch in the coastal ecosystem of the Pearl river estuary in 1981 (2.85) and 1998 (2.3) (Duana et al. 2009). This may be explained by different structures of fisheries in the ecosystems. Ideally, in this study, fisheries catches were mainly caught from gillnet (31% of total catches, Table 6.2) which is targeting on species at high trophic levels such as large predators and tuna. Focusing on high trophic levels can cause the food web being fished down which might lead declining catches afterward (Pauly et al. 1998).

The gross efficiency of the catch in the present model (0.0018), calculated as the ratio of the catch to the primary production, is almost ten times higher than the values estimated by Christensen et al (2005) using global data (0.0002). However, this value is lower than those obtained by Duana (2009) for the coastal ecosystem of the Pearl river estuary in 1998 (0.0047) and by Wolff (1994) for the Tongoy Bay in Northern Chile (0.018). The low values we found in the present study again are symptomatic of fisheries harvesting species at high trophic levels (Duana et al. 2009).

The ‘system omnivory index’ (0.146) of the coastal ecosystem of Vietnam is lower than of that of Tonkin Gulf (0.198, (Chen et al. 2008a)) but higher than that of the coastal ecosystem in the Pearl river estuary (0.127, (Duana et al. 2009)) and of the Tamiahua system (0.13, (Cruz-Escalona et al. 2007)). The ‘connectance index’ in the present study (0.298) was lower than that found by Chen et al. (2008a) for the Tonkin Gulf ecosystem (0.33), higher than that for the Pearl river estuary (0.27, (Duana et al. 2009)), but comparable to that found for Laguna Alvarado, western Gulf of Mexico (0.3) (Cruz-Escalona et al. (2007)). This index indicated trophic links of prey on the ecosystem. The higher the index is, the more complexity consumers use their preys and vice verse (Odum 1969).

The coastal ecosystem of Vietnam has been experiencing intensive overexploitation by fisheries. Scenario A (no changes in fishing effort) caused declines in the biomasses of almost all species groups except for anchovy and crustaceans. The biomass increases of anchovy and crustaceans can be explained by the reduced density of their predators. Another explanation for these increases was that losses due to fishing were lowest for these two groups (only 0.05 and 0.27 $t{\cdot}km^{-2}{\cdot}year^{-1}$ for anchovy and crustaceans, respectively; Table 6.2). The development of the fishing capacity following the increasing demand for marine resources is placing intensive pressure on marine resources in the coastal ecosystem of Vietnam (Pomeroy et al. 2009). Simulations for scenario A show that keeping fishing effort at the 2000-2005 level puts the coastal marine resources at risk as further stock declines or collapses are possible. Reductions of 20% for trawl fisheries or gillnet and purse seine fisheries (Scenario C or D) on the other hand did also not certify a benefit for the entire coastal ecosystem. Scenario B with a 10% reduction of fishing effort for all fisheries is likely to stop fishery resource decline for all groups at least under the wasp-waist flow control situation. However, the prospect of the application of such a management action is difficult. Indeed, the total number of small fishing boats with a capacity below 90 HP operating in the coastal areas contributed more than 80% of total vessels in 2012 (DECAFIREP 2013). In addition, communities fishing in the coastal areas are very poor with very low net incomes and their lives are based to a large extent on fishing (Pomeroy et al. 2009). The implementation costs and social benefits also need to be considered and issues such as fishermen community resistance and/or cooperation need to be taken into account if scenario B is to be put into practice. Alternatively, a management paradigm shirt from coastal into offshore fisheries will be a possible solution. According to stock assessment of offshore fisheries of Vietnam, continuing to enhance fishing capacity in the offshore areas targeting on oceanic tuna fisheries is possible (RIMF 2005b). Therefore, a possible management strategy is to reduce fishing effort in the coastal areas to move into offshore areas. Certainly, there is a need to upgrade or change current small fishing vessels into larger ones.

A scenario that maximizes all benefits in the ecosystem approach to fisheries management is a great challenge and it is not easily achieved. There are usually trade-offs between conservation and socio-economic objectives in fisheries management of tropical marine ecosystems with multi-species and multi-gear perspectives (Cheung and Sumaila 2008). Results of this study revealed inverse pictures in economic and social scenarios (Figure 6.5A and 5B). This is demonstrated that when a high weight is given to economic gain then priority would be given to the fisheries targeting the high valuable species (e.g. gillnet, shrimp trawl and purse seine). In contrast, when a high weight is given to social aspects, a fishing effort increase is suggested for the artisanal fisheries (e.g. other, squid handline, fish handline and trap fisheries) to provide more livelihoods for coastal fishing communities. However, these can cause the ecosystem to be vulnerable and thus a suitable and wise management policy strategy in ecosystem approach to fisheries management is to balance socio-economic and ecological goals. In this study, we found a compromise by giving equal weights to economic, social and ecological criteria. In this scenario, there was no considerable loss in socioeconomic and ecological aspects, except a reduction of the effort for purse seine and fish trawl fisheries. However, it should be noted that fish trawl fisheries in Vietnam are already considered unsustainable and are no longer the most profitable and important fishery due to high increase of fuel price at the moment (DECAFIREP 2013).

As indicated already by Christensen et al. (2009), obtaining sufficient data to develop and apply ecosystem models is challenging. This not only applies to Vietnam but to almost all developing countries where fisheries monitoring systems are insufficient (Pomeroy et al. 2009). In the present model, we used data from different sources - locally and regionally. Data uncertainty was evaluated by using the pedigree index as described in the methodological section. The pedigree index we found for our model was 0.32, which was located in the lower range of what is typically found for Ecopath models by Morissette et al. (2006), indicating that more investigations and local data are needed, especially data on the feeding ecology of the species groups in the model. In the present study, producing the feeding matrix was hampered by a lack of studies on the feeding ecology of the species present.

The lack of human and financial resources usually leads to low quality data in developing countries. In some cases, we estimated catch data from logbooks provided by fishing communities and these data are usually considered underestimations. The assessment of non-reported and/or under-reported landings should be considered for future studies. In addition, in the future, it is critical that model estimates are subjected to detailed evaluations, including thorough sensitivity analyses of uncertain parameters, comparisons with independent field estimates for a variety of taxa, and comparisons with outputs of other models. Once those validations have been conducted and the model has been further refined, the next logical step will be to use how the Vietnamese coastal ecosystem might respond to alternative fisheries management strategies and changes in environmental variables.

Chapter 7: General discussion

Redrafted from

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7. Chapter 7: General discussion

Since the enactment of the Fisheries law in 2003 Vietnamese fisheries management agencies have been mandated to ensure that fisheries activities are carried out in a sustainable manner (Vietnamese Government 2003). However, under the Fisheries law, management measures have tended to focus on single stocks or species, with catch limits determined by estimates of the harvest rate. Following the recent shifts towards ecosystem approach to fisheries management (EAFM) at global and regional level, Vietnam needs to redefine the sustainable fisheries management objectives. EAFM is a tool to manage many components in the ecosystem, consulting different stakeholders (FAO 2008). The data and information requirements are larger than those for single-species approaches (FAO 2008). Therefore, it is better to realize what is needed, what is available and how to best use the data for analysis (Christensen et al. 2009). Data-limitation can apply even when there are large datasets but these are not useful for assessing impacts of fishing or policy outcomes. This can be due to data quality issues, lack of information in the data with respect to key variables, or because the data are not relevant to management questions or are not available at appropriate scales for aiding management decisions. It is also necessary to select suitable tools that input data and information can also easily be obtained and referred from the literature in other related regions. In this thesis, three different tools were used to evaluate impacts of fishing on the ecosystem types of Vietnam. Although each has its strengths and its weaknesses, they can be considered as the suitable options and 'best available tools'.

Firstly, I used fishery-based indicators, which are easily interpretable and cost effective tools for assessing ecosystem impacts. Estimations of these indicators do not require intensive data collection and complex modelling for describing the state of fishery resources and fishery activity. The use of this approach would provide information benefits by improving the assessment findings, and the applicability of fishery management measures, particularly in data limited fisheries of Vietnam. However, the environmental, economic and social factors that influence the ecosystems cannot be introduced in this approach.

Next, I made use of linear inverse models (LIM). The structure of the model provides an overall view of the ecosystem and underlines the uncertainties that could be filled with future studies. The use of upper and lower limits to constrain the majority of input values (production, consumption, export, and diet composition) and the choice of row and column weights make inverse modelling as a flexible tool to quantify mass-balanced flow diagrams and trophic transfer efficiencies that are internally consistent. Under insufficient data situations, the inverse model enable referring different information used to quantify the flow diagram and that are sometimes inaccessible for measurements (Gaedke 1995). For example, collecting information in the arctic sea, the deep sea and seamounts where normal fishing gear types may not sometimes be used to access. This Chapter aimed to answer the question if there are differences on ecosystem function and to study the development level of this ecosystem and its state of maturity, which facilitates profound understanding of the function of the whole ecosystem for analysing the impact of human influences. An inverse modelling was developed to reconstruct carbon flows between different functional groups of the local food web for the 1990s (lower fishing intensity) and for the 2000s (higher fishing intensity), using local fish stock data and ecological and physiological constraints obtained from the literature. Overall, results of this study can initially indicate changes between the 1990s and 2000s. However, there were several assumptions and constraints behind the present results. In fact, input data of the inverse models were much referred from different sources. It is therefore important to acknowledge that our results and interpretations will be more reasonable when local independent and large-scale monitoring is conducted for better validation. In addition, future studies should integrate a variety of disturbance sources including environment and human factors into ecosystem models to quantify the relative contribution of these drivers to changes in the structure and functioning of the Vietnamese coastal ecosystem.

Finally, I implemented an Ecopath with Ecosim model (EwE) extrapolating the missing data from the literature or from existing available sources (e.g. diet composition referred from Fishbase). This will be necessary and useful for fisheries monitoring and management activities in Vietnam where fisheries data are usually limited due to lack of financial and human resources. This Chapter consists of an assessment of the Vietnamese fisheries in an ecosystem-based framework, which is the latest approach in fisheries assessment and covered more broad issues including simulations of different management perspectives that have not been implemented in the previous chapters. As in the other chapters, the main focus of the assessment was the fisheries. This chapter has provided the most detailed assessment on other aspects in the fisheries such as social and economic constraints. In addition, it quantifies the interactions among the organisms and the fisheries. As far as the fisheries are concerned, it presents quantitatively the actual values of the level of exploitation in relation to the potential of the resources. And the most important for management is that it can predict what are the consequences for the fisheries and the ecosystem under different management scenarios. It is also the most important section to be considered in any decision-making process. All the previous chapters assess what has happened to the system up to the present. Such type of information is important for knowing where we are and how far we have exploited the resources. However, they are not applied and tested to quantify the future possible scenarios. This chapter has provided the ecosystem-based assessment that is emphasizing fish communities and considering ecological, social and economic aspects and thus it benefits and practices for the Vietnamese fisheries assessment and management. However, it is critical that future studies should integrate detailed evaluations including thorough sensitivity analyses of uncertain parameters and comparisons with independent field estimates to reduce potential uncertainties caused by input data.

7.1. Strengths and weaknesses of used tools

7.1.1. Ecological indicators

Ecosystem approach to fisheries management (EAFM) requires that managers integrate a wide range of fisheries impacts when setting management objectives. Attempts to meet these objectives will need to be supported by reliable scientific advice and effective management decision-making (Murawski 2000). Fishery-based indicators such as marine trophic indicators have been developed at global (Pauly et al. 1998), regional (Pauly and Palomares 2005) and local level in many countries such as Thailand (Christensen 1998), Canada (Pauly et al. 2001), China (Pang and Pauly 2001), Portugal (Baeta et al. 2009), Senegal (Laurans et al. 2004), the United States of America (Steneck et al. 2002), Mexico (Sala et al. 2004), Chile (Arancibia and Neira 2005), Greece (Stergiou and Karpouzi 2001), India (Bhathal and Pauly 2008), and Brazil (Freire and Pauly 2010). Details are indicated in the Table 7.1. Of those, many studies have been considered for management. Fishery-based indicators usually do not give absolute answers but they often suggest the next most essential question. There are several advantages to use fishery-based indicators. First, the data are more likely to be readily available. Second, the indicators can help to define problem areas and can be combined with other indicators to compensate each other to reduce weaknesses. Therefore, in this study I used a number of fishery-based indicators such as MTL, FiB and P/D to compare the ecosystem status over time. The use of fishery-based indicators in the evaluation of ecosystem can be suitable for data-poor areas as in the Vietnamese fisheries. However, some problems such as the aggregation and disaggregation of landing data by species and areas (Moutopoulos et al. 2014), bycatch issue (Catchpole et al. 2008), etc. and their influence on indicator calculations should be considered to reduce future uncertainties.

7.1.2. Inverse modelling

Investigation of a food web structure is needed to determine how energy from primary production channels between compartments in a food chain. However, one major problem in food web structure studies is to quantify the exchange of mass or energy between all the food web components. To do so, complex interactions and a lot of data are required. To overcome this data deficiency, various mathematical models that estimate the unmeasured quantities in a food web have been developed. Inverse analysis is increasingly used in ecosystem modelling to objectively reconstruct a large number of unknown flows or interactions from a small number of observations (Vezina and Platt 1988). This research has illustrated that this approach is suitable in the data-poor situations like in Vietnam. So far there is not an application of inverse model at global level. However, many of studies on inverse models have been performed at regional and national levels worldwide (Table 7.1). Especially, these studies could be used in conjunction with other approaches to provide practical management advices such as the case of fisheries management in Canada (Savenkoff et al. 2007). However, because the inverse models can refer different information as the constraints and this may also cause shortcomings on the inverse modelling. Therefore, it is necessary to validate the outcomes using data collected from local independent and large-scale monitoring.

7.1.3. Ecopath with Ecosim model

This modelling approach is based on the assumption of mass-balance to describe the structure and functioning of ecosystems, and allows temporal and spatial simulations that can be robust even in data-deficient environments (Christensen and Walters 2004). Ecopath with Ecosim model (EwE) is widely used to explore optimal harvesting strategies by examining the ecosystem effects of fishing (Christensen and Walters 2004). This approach does not need to explicitly address the full complexity of ecosystems but it can still provide an ecological perspective for the assessment and management that is suitable for multi-species and multi-gear fisheries (Pauly et al. 2000).

The principal advantage with EwE is that the input values (mainly total mortality, consumption and diet composition) are often already available for several species or groups in the studied ecosystem or they can easily be referred from other ecological models (Christensen and Pauly 1992). Therefore, the model makes it possible to ensure that available data for an ecosystem will be referred from the other ecosystem (Christensen and Walters 2004). Another advantage of the EwE is ease of use due to its well-structured parameterization framework and its publicity worldwide.

EwE models have been used to examine the trophic structure and functioning of a host of aquatic ecosystems, including lakes, aquaculture systems, estuaries, small bays, coastal systems and coral reefs, shelf systems, upwelling systems, and open seas (Morissette 2007). It was applied at global, regional and local scale (Table 7.1).

In this thesis, EwE was used to gain better insights into the responses to exploitation of target and non-target species in the relatively poorly studied ecosystem.

Ecosim simulations were used to verify if the fishing effort changes could affect the components of the ecosystem after a 15-year simulation. In addition, scenarios that optimized for economic, social, and ecological criteria were also implemented to select suitable fisheries management strategies in Vietnam. The first of these, optimizing economic (profits), was based on calculating profits as the value of the catch (catch · price, by species) less the cost of fishing (fixed + variable costs). Giving a high weight to this objective often results in removing out most fleets except the most profitable ones.

The second criterion, optimizing social benefits, was expressed through the employment supported by each fleet. The benefits were calculated as number of jobs (fishers) relative to the catch value, and were fleet specific. Therefore, social benefits are largely proportional to fishing effort. This means that optimizing the social criterion will promote fisheries that produce more employment (e.g. manual collection or traditional fisheries). Optimizing efforts often leads to even more extreme (with regards to overfishing) fishing scenarios than optimizing for profit.

The last criterion included optimising ecosystem structure (or 'health'). It was based on the description of ecosystem maturity by Odum (1969), in which mature ecosystems are assumed to be characterized by the domination of large and long-living organisms (e.g. Christensen 1995). The default setting for ecosystem structure was therefore the group-specific biomass/production ratio as this measure is indicative of the longevity of the groups. The ecosystem structure optimization often implies reduction of fishing effort for all fleets except those targeting species with low weighting factors.

A main aim of fisheries management is to regulate fishing effort over time so as to achieve economic, social and ecological sustainability objectives. Ecosim simulations were thus to provide insight about how high these fishing effort should be, and how they should be varied over time. Therefore, we should at least be able to define reasonable and pragmatic ranges for fishing effort. In summary, the present model results have proven that EwE was a useful tool for policy exploration of harvesting strategies in multispecies management. However, exploration and interpretation of results must be addressed carefully, considering how input parameters can affect results. It must be stressed that the EwE did not intend to provide an actual and practical fishing policy evaluation for the Vietnamese fisheries. Rather, it is an exercise to explore and test the overall responses of the coastal ecosystem model to various multispecies management strategies. A more realistic approach is to include accurate data for prices per species, fleet operational costs, employment indicators for each fishery, and ecological parameters that better represent the current status for the ecosystem.

Table 7.1. Overview of used tools for management application.

	Examples where the tools used		
Scale	**Fishery-based indicators**	**Inverse model**	**Ecopath with Ecosim model**
At global scale	(Pauly et al. 1998)	N/A	Christensen et al. (2009)
At regional scale	Atlantic and Pacific (Pauly and Palomares 2005), West Africa (Laurans et al. 2004).	Northern east Pacific (Vezina and Savenkoff 1999), Southern Barents Sea (De Laender et al. 2010), North Atlantic (Daniels et al. 2006), Arctic Ocean (Forest et al. 2011).	Eastern Mediterranean (Gucu 2002), Scotian shelf (Bundy 2005), Baltic Sea (Harvey et al. 2003), Northwestern Mediterranean (Coll et al. 2006), central Pacific (Kitchell et al. 2002).
At national scale	Thailand (Christensen 1998), Canada (Pauly et al. 2001), China (Pang and Pauly 2001), Portugal (Baeta et al. 2009), The United States of America (Steneck et al. 2002), Mexico (Sala et al. 2004), Chile (Arancibia and Neira 2005), Greece (Stergiou and Karpouzi 2001), India (Bhathal and Pauly 2008), and Brazil (Freire and Pauly 2010), Vietnam (this study).	English coast (Vezina and Platt 1988), French (Marquis et al. 2007), Canada (Savenkoff et al. 2007; 2004) The United States of America (Stukel et al. 2012), Vietnam (this study).	Mexico (Diaz-Uribe et al. 2007), China (Duana et al. 2009), South Africa (Shannon et al. 2008), Australia (Griffiths et al. 2010; 2009), Sri Lanka (Haputhantria et al. 2008), Pacific Ocean (Cox et al. 2002), Canada (Morissette et al. 2009), Vietnam (this study).

7.2. Changes in structures and function of Vietnamese marine ecosystem

In this thesis, changes on structure and function of the Vietnamese marine ecosystem were investigated. It was found that there was a slight decline in mean trophic level of landing of about 0.01 trophic level per decade. However, because of small change in the trophic level and uncertainty of input data and the present results did not confirm the "fishing down the food web" phenomenon (Pauly et al. 1998). This can be due to trophic networks in the tropical ecosystems have a lower likelihood of being impacted by fishing effects than those in the temperate ecosystems (Navia et al. 2012).

Results presented in Chapter 5 and 6 demonstrate that the Vietnamese coastal ecosystem has experienced changes, and stress the need for a closer inspection of the ecological impact of fishing. In the Chapter 5, food web efficiency of the groups at high trophic levels was reduced and more carbon budget is transferred to the groups at the lower trophic levels such as large invertebrates and zooplankton over time. Results of the models also revealed that total system throughput, ascendency, overhead and capacity, ratio of ascendancy to development capacity, Finn's cycling index, and constraint efficiency index were higher than of those in the previous period. This indicates that the Vietnamese coastal ecosystem in the past was more developed, stable and mature than in its present status. However, it is not very clear what is a main factor causing these changes and thus, future studies should integrate a variety of disturbance sources into more representative models to quantify the relative contribution of several potential drivers to changes in the structure and functioning of the Vietnamese coastal ecosystem. In addition, the studied period was too short and there were several assumptions and constraints behind the present results. It is therefore important to acknowledge that the present results and interpretations will be more reasonable when local independent and large-scale monitoring is conducted for better validation.

Results of Chapter 6 demonstrate that if fishing effort is maintained at the 2000-2005 level, the biomasses of almost all stocks decreased after a simulation 15-year period. A simulation of changing fishing effort showed that reducing fishing effort for all fisheries by 10% increased the biomass of almost all groups in the ecosystem. The scenarios to select optimal fishing strategy indicated that achieving economic, social and ecological goals was possible by four-fold increase of traditional small-scale fisheries, combined with 40 and 45% reductions of purse seine and fish trawl fisheries, respectively. However, due to practical complexity of implementing such reductions in the specific fisheries, a more pragmatic measure would be an overall reduction by 10%. Nevertheless, further implementation of this result must be addressed carefully, considering how input parameters can affect results using sensitivity analysis to validate the present results.

Although it is possible to divide the effects of fishing impacting the ecosystem structure and functioning, they are all interrelated and may magnify if fishing pressure increases (Andres et al. 2012) or due to natural effects. In fact, a trophic network is a complex structure due to high levels of interaction among its elements to maintain dynamic processes that contribute to its stability. As indicated throughout the different Chapters of this thesis, the Vietnamese fisheries can affect trophic network by ecosystem structural and functional changes. Therefore, an effect generated by one of these changes will likely affect the others (Andres et al. 2012).

Table 7.2. Overview of the spatiotemporal scale of the studied data and the purpose of the analyses.

Studied properties	**Chapter 4: Fishery-based indicators**	**Chapter 5: Inverse model**	**Chapter 6: Ecopath with Ecosim model**
Studied temporal scale	1981-2012	1990-1995 and 2000-2005	2000-2005
Studied spatial scale	Macro scale (i.e. entire Vietnamese EEZ)	Micro scale (i.e. limited within the coastal ecosystem)	Micro scale (i.e. limited within the coastal ecosystem)
Ecosystem structure	Decline of trophic level of 0.01 per decade	N/A	Fishing at high trophic level
Ecosystem functioning	N/A	Changes on functional attributes, food web efficiencies and network indices were found	N/A
Management scenarios	N/A	N/A	A simultaneous reduction of 10% on fishing effort of all fisheries recommended.

In this thesis, I firstly used fishery-based indicators in Chapter 4 to analyse the Vietnamese marine ecosystem at a macro scale for entire Vietnamese EEZ and long time series fisheries data (Table 7.2). The outcomes are relevant to obtain an overview of the ecosystem as a whole under impacts of fishing. Later, I analysed impacts of fishing at a micro scale (i.e. only focussing on the Vietnamese coastal ecosystem) (Chapter 5 and 6). Therefore, any changes highlighted by the analysis at the macro scale at Chapter 4 were then verified at the smaller scale at Chapter 5 and 6 (Table 7.2). At a certain level, these analyses have indicated the different conditions of the Vietnamese ecosystem under a changing fishing intensity. However, it is necessary to have future studies using data locally collected to validate the present results and propose suitable management options in the light of EAFM.

7.3. Future improvements and applications

It is important that future ecosystem models can more accurately describe ecosystem processes and deal with as many changes as possible on the ecosystem (e.g. human-induced changes, climate and genetic changes, etc.). In fact, the marine ecosystem may be under enormous stress. There may be a variety of human-induced disturbances causing three main effects: 1) changes in nutrient cycles and climate which usually affect ecosystem structure from the bottom up, 2) fishing activity which could usually affect ecosystems from the top down, and 3) habitat change and contamination which affect ecosystems at all trophic levels (Navia et al. 2012). Climate change can cause biodiversity shifts in marine ecosystems locally, regionally and globally (Vitousek et al. 1997). As such, investigations on the combination of natural and human-induced changes taking place in the ecosystem (e.g. rising temperatures, ocean acidification, changes in biodiversity and species distribution, and depletion of fisheries stocks, etc.) are necessary. The identification of single pressures, and of their effects, may not be sufficient to account for possible cumulative effects. However, the tools supporting for ecosystem approach to fisheries management, until now, dealt mainly with ecological issues such as predator-prey relationships, fisheries management, biodiversity, etc. Now that ecosystem evaluation tools are becoming more and more popular, there is a need to focus on merging different fields to better understand the structure and function of ecosystems. Many studies have shown that fish stocks can adapt genetically to new life history traits by so-called fisheries induced-evolution with faster matured individuals and a smaller size at first maturity (Law 2000, Kuparinen and Merila 2007). The evolution of the preys can also modify considerably predatory-prey relationships (Yoshida et al. 2003). However, in this thesis, diet compositions were derived from literature and the evolutionary ability of the predator-prey relationship can be masked. This can cause higher uncertainty on estimated flows and hence limit the present results. This higher uncertainty is undoubtedly the result of limited data that are available to constrain these flows, a situation that is typical for food web reconstructions. This may be improved by more detailed information on diet composition using local data in future studies.

Some biotic compartments are missing from the food web topology of the present models. For instance, microfauna or nanobenthos are not included or represented less detail because of a lack of empirical data. In addition, the fish compartments were also aggregated in the present models. Therefore their role in the Vietnamese ecosystem structure and functioning could not sufficiently be assessed and this represents a shortcoming of the present models.

The assumptions and limitations of used tools were also discussed in the relevant chapters. Ideally, these results would be examined in more depth, for example, to examine ecosystem evolution and the impacts of model assumptions and input data. However, these are out of the present research scope and thus new approaches should integrate these different fields for even more representative models and analyses.

7.4. Recommendations for management

7.4.1. Development of regular data collection systems

Fisheries management in general needs good scientific data on the status of target species. Adequate data on bycatch, non-target or target species, and indicators of ecosystem changes are fundamental prerequisites for EAFM (Morishita 2008). In this study, because there is no information on discard and bycatch on the Vietnamese coastal fisheries, we assumed that all catches were landed equal to total landings. This can cause bias and uncertainties on the model outcomes and thus discard and bycatch information should be considered and collected in data collection programs such as observer on board to reduce potential biases for future studies.

Science, which will form the basis of management measures, should be transparent and reliable. Differing scientific views and uncertainties need to be clearly presented and, ideally, scientific findings should be peer-reviewed and confirmed by a third party. As controversy is often generated around scientific uncertainties and the validity of the claims such as the magnitude and impacts of bycatch, transparency, and validity are essential elements of good science for ecosystem management.

In data-poor fisheries such as in Vietnam, it is important to ensure an accurate and effective use of existing data for the monitoring of fisheries and fish stocks. In fact, in this thesis, we used approaches to reconstruct historical data for entire the Vietnamese EEZ (Chapter 3). However, this historical data reconstruction can cause unexpected results by uncertainties on assumptions. In fact, in this study we assumed that catch rate were the same in every year of the studied periods. In addition, because there were no available data to break down by gears of lift net, stick net and traditional fisheries and thus landing of these gear types were all denoted as the mixed gears (Appendix 3.10). These assumptions can affect the present results. Therefore, future data collection should be developed to collect detail information on different gear types and catch and can be used to evaluate variations on the catch rates by different fishing gears and then such validated input data will improve model outcomes in the future.

The most basic need for an ecosystem approach is the collection of comprehensive data on the stocks and habits of both target and non-target species within an ecosystem (Hilborn 2011). However, in Vietnamese fisheries, even a basic stock assessment is sometime missing. There is a need for the development of more efficient ways to collect better data and to use these data in management decisions. Good use of available data can significantly improve the chances of setting appropriate management measures. In view of these, it is important to develop and establish catch and effort recording systems from local to central level to have time series data for fisheries monitoring and assessment in Vietnam. These data collection systems must be considered in such a way as to account for as many of sources of bias as possible in the collection of data, particularly in the absence of sufficient resources (financial resources) for the collection of fishery independent statistics.

In general, data collection, data processing and the use of data in management decisions can be resource intensive. However, in some cases useful data will inevitably be available at fishing communities themselves. Therefore, enhancing better collaboration and strong partnerships between the fishing and scientific communities are crucial for acquiring locally held or from traditional knowledge.

In addition, enhancing and initial implementing market-based incentive tools is also necessary (Anh et al. 2014b). This management approach is to develop standards, ecolabels, and product traceability initiatives. The approach require fishing communities to more comply with legal frameworks and better cooperate with management agencies on providing fisheries data (their logbooks including fishing effort, catch, fishing locations, etc.) if they want to get their certified products.

By enhancing strong partnerships and implementation of market-based incentive tools, fishing communities and management agencies can work together on fisheries management in general and data provision in particular. This is because increasing knowledge and awareness of stakeholders on fisheries management could help to create the enthusiasm of all stakeholders to drive positive change.

Once time series fisheries data are sufficiently available, ecosystem tools can be further developed and validated. They could also be used to manage overexploitation of both target and non-target species that prey on or compete with target species. Such ecosystem data and tools could also be used to evaluate ecosystem-based reference points, and lead to modification of exploitation rates to achieve desired ecosystem states.

7.4.2. Reduction of fishing capacity

Coastal and near-shore areas are commonly considered to be fishing ground of many small scale fisheries (Vinh et al. 2006). However, due to lack of enforcement of policy regulations to remove larger vessels fishing in inshore waters, increased amount of conflict between small and large vessel has been reported (MARD 2013). In fact, as found in the Chapter 6 that maintaining the fishing effort at the 2000-2005 level puts the coastal marine resources at risk as the biomasses of ten out of twelve stocks decline by 5 to 20% in a 15-year period.

Various approaches to reduce over-exploitation have been tried out without success in the past in Vietnam. For example, buyback schemes for small-scale fishers (the Government paid money to buy old and small vessels) have been tried in several locations in Vietnam. Legal legislations to temporally or permanently prohibit fishing activities in some areas have been adopted (MOFI 2006, MARD 2011). However, the implementation of these legal legislations is either weak or lack of fully compliance of fishing communities (DECAFIREP 2013). The main reason for this is that legal frameworks were only based on top-down control management regimes with forcing from management agencies without involving relevant stakeholders on developing and implementing the legislations (Pomeroy et al. 2009). Given these realities, the only feasible solution may be the one based on a coordinated and integrated approach involving a mixed strategy of resources management (access control and property rights); resource restoration; economic and community development (linkages of coastal communities to regional and national economic development), including poverty reduction and livelihoods; and new governance arrangements. Thus, in practical terms, reduction of over-capacity implies an increased focus on people-related solution. This is due to policies that reduce the number of fishers in small-scale fisheries without creating non-fishery livelihood opportunities will inevitably fail. It is necessary to give fishers and their families a broader range of livelihood options and to reduce the household's economic dependence on the fishery. However, this approach requires strong inter-ministerial and national and provincial and district government linkages to ensure coordination and cooperation for planning and implementation.

7.4.3. Fisheries management toward ecosystem approach in Vietnam

The need to move away from the management of single species towards management of whole ecosystems, the so-called "ecosystem approach to fisheries management" (EAFM), is now accepted by many scientists and managers (FAO 2008). The EAFM should balance human and ecological well-beings and can fairly combine different aspects on fisheries management such as ecological, social and economic constraints (Anh et al. 2014a).

Fishing at high catch levels affects not only the most productive species in an ecosystem but also the less productive species taken coincidentally as part of a multispecies complex or as bycatch (FAO 2014). The latter can lead to serious population declines. In facts, the bycatch proportion annually estimated was nearly 38.5 million tons contributing about 40% in the total annual global marine catch (Davies et al. 2009). Bycatch is impacting on entire spectrum of marine fauna and fishing gear including turtles on hooks, juvenile fish in net, and benthic invertebrates in trawl and dredge gear

(Davies et al. 2009). According to recent assessment at Western and Central Pacific ocean, resource status of billfish stock that is the bycatch of tuna fisheries is overfished (WCPFC 2014). Addressing these management problems on ecosystem level requires an EAFM to integrate many components in management actions. EAFM requires gathering information using broad monitoring programs, ecosystem model development for long-term policy comparison, and field-scale adaptive management experiments to directly evaluate the benefits and pitfalls of particular policy measures. Adaptive management experiments may provide the best information to support EAFM. EAFM can consider marine ecosystem complexity on environmental/physical variability and the complex integrations between and among internal and external ecosystem components. To date, no attempts have been made to set up a management system taking marine ecosystems in Vietnam into account. It should be noted however that with limitations of capacity and financial constraints in a developing country such as Vietnam, and small-scale fisheries, a rapid change towards EAFM is difficult because of its long term benefits (Garcia and Cochrane 2005). Sometimes the benefits from using the EAFM do not immediately appear but in the long-term process. Therefore, social-economic aspects should be considered as high priorities. Participatory and adaptive approaches will also need to be developed, drawing on existing traditional rights and management systems whenever possible (Garcia and Cochrane 2005).

This thesis has developed and explored a number of tools to assess impacts of fishing on Vietnamese marine ecosystem. These designed tools were applied to help fishery managers and scientists think more clearly about issues relating to EAFM, particularly related to future policy making in order to balance ecological, social and economic aspects in the management perspective in context of multispecies fisheries of Vietnam. However, it is noted that the work presented here represents only a very small set of issues that will need to be considered when setting up ecosystem-based management plans in Vietnam.

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Thesis abstract

The Vietnamese marine ecosystem is under threat from a wide range of human activities including fisheries. These activities can change the structure and functioning of the marine ecosystem. To achieve more sustainable fisheries, an ecosystem approach to fisheries management (EAFM) is needed. Therefore, the overarching aim of this thesis is to develop scientific tools that can support EAFM in the Vietnamese marine ecosystem. To do so, assessing fishing impacts on ecosystem structure and functioning of the Vietnamese marine ecosystem were performed using different methodologies. First, a time-series assessment of the fisheries was carried out based on reconstructed catch data (landing) between 1981 and 2012. Then, based on the landing data, fishery-based indicators were used to assess the impacts of fishing on the structure of the entire Vietnamese marine ecosystem. Next, inverse models and Ecopath with Ecosim model (EwE) were used to evaluate fishing effects on fish communities. Of those, the inverse models were applied to investigate human-induced changes on the functioning of fish communities in the Vietnamese coastal ecosystem. The EwE modelling tool was used to predict ecosystem-level consequences of various fishing scenarios, taking into account the trophic interactions and ecological, social and economic constraints of the Vietnamese coastal fisheries.

The results provide knowledge about the Vietnamese marine ecosystem emphasizing on fish communities and its fisheries and show changes in the functioning and structure of the ecosystem within the investigated time frame.

Chapter 1 and 2 provide respectively a general introduction on impacts of fishing on ecosystem. Overview and selection of potential approaches for assessing fisheries management in Vietnam were also implemented in these chapters.

In chapter 3, different techniques were applied to reconstruct total numbers of fishing vessel and catches by fisheries from 1981 to 2012. The total fishing vessel of seven types of fisheries and related catch of 12 ecological groups were reconstructed and used as a basis input data for further investigations in the next chapters. Effort data reconstruction indicates that there were a stable trend on number of fishing vessels for all gear types during 1981 to 1985 and then a slight increase trend from 1990 to 1995. Total number of fishing vessels for all gear combined increases four times from 1981 to 2012 (i.e. from 30,000 to 120,000 vessels). Similar trends found for reconstructed data of fisheries catch as for effort data. Although there is a level of uncertainty associated with reconstructed data, outcomes from this study will provide the necessary input for further studies attempting to assess and manage the Vietnamese marine ecosystem.

In Chapter 4, I use a set of fishery-based indicators (i.e. mean trophic level of landing (MTL), the fishing in balance index (FiB), and the pelagic/demersal (P/D) ratio to assess

fishing impacts for entire Vietnamese EEZ from long-term data (1981-2012) of 12 ecological groups. The analyses in chapter 4 found a slight decline in mean trophic level of landing of 0.01 trophic level per decade. The FiB index increased in certain periods that are in line with fisheries development strategies of the Vietnamese Government to expand further fishing grounds at that time. The use of fishery-based indicators in the evaluation of ecosystem offers an alternative to complex models but it is useful in the Vietnamese fisheries under data-poor area. However, the artifact problems on indicator calculation of future studies should carefully be taken into account to reduce the unnecessary biases in the outcomes.

In Chapter 5, I perform inverse models that integrate fisheries and food webs to evaluate changes on functioning of the Vietnamese coastal ecosystem. Functional attributes, food web efficiencies, and functional food web indices were used to investigate their differences between the 1990s (low fishing intensity) and the 2000s (high fishing intensity). Overall, results from this chapter demonstrate that the Vietnamese coastal ecosystem has experienced changes between the 1990s and 2000s, and stress the need for a closer inspection of the ecological impact of fishing. However, it is not very clear to what extent the changes between two studied periods were only caused by fishing or other causes. Thus, future studies should integrate a variety of disturbance sources into more representative models to quantify the relative contribution of several potential drivers to changes in the structure and functioning of the Vietnamese coastal ecosystem. In addition, our studied period was too short and there were several assumptions and constraints behind the present results. It is therefore important to acknowledge that the present results and interpretations will be more reasonable when local independent and large-scale monitoring is conducted for better validation.

In Chapter 6, an Ecopath with Ecosim model was used to evaluate interactions between fisheries and the food web with fisheries data from 2000 to 2005 focusing only on the Vietnamese coastal ecosystem. The present results found that maintaining the fishing effort at the 2000-2005 level puts the coastal marine resources at risk as the biomasses of many stocks decline in a 15-year simulated period. An overall reduction of 10% of fishing effort for all fisheries will increase the biomass of almost all groups in the ecosystem. In addition, the scenarios to optimize fishing strategy indicated that achieving economic, social and ecological goals was possible by four-fold increase of traditional small-scale fisheries, combined with 40 and 45% reductions of purse seine and fish trawl fisheries, respectively. However, the overall reduction by 10% of all fisheries will be a more pragmatic option due to practical complexity of implementing the reductions in the specific fisheries as recommended in the optimal scenario of economic, social and ecological. It is also emphasized that exploration and interpretation of results must be addressed carefully because input parameters used in the present model can affect results. In conclusion, I did not intend to use the present results to recommend practical fishing policies rather than they are only an exercise to explore and test the overall responses of the ecosystem model to various multispecies management strategies. A more realistic approach is to include sensitivity analyses of uncertain parameters, comparisons with independent field data estimates to reduce potential uncertainties.

Samenvatting

Het vietnamese marien ecosysteem wordt bedreigd via een brede waaier aan menselijke activiteiten, waaronder de professionele visserij. Deze activiteiten kunnen zowel de structuur als het functioneren van het marien ecosysteem beïnvloeden. Om naar een meer duurzame visserij te streven is het daarom nodig om een ecosysteem-gebaseerd visserijbeheer (EGVB) te ontwikkelen en implementeren.

In die context is de belangrijkste doel van deze thesis om wetenschappelijke instrumenten te ontwikkelen en gebruiken om een EGVB op te stoelen. Hiervoor werden verschillende methoden gebruikt om de beoordeling van de structuur en het functioneren van het Vietnamees marien ecosysteem mee uit te voeren.

Bij het begin van dit doctoraat werd een overzicht gegeven van bestaande modellen die hiervoor zouden kunnen gebruikt worden, en werd een selectie doorgevoerd op basis van hun relevantie en haalbaarheid voor het Vietnamese marien ecosysteem te beschrijven en beoordelen inzake structuur en functioneren. Het praktisch onderzoek werd vervolgens gestart met de analyse van een tijdsreeks die uit gereconstrueerde landingsgegevens van de periode 1981-2012 bestond. Vervolgens werden verschillende indicatoren berekend op deze gegevens om de impact van de visserij op de structuur van de mariene visgemeenschap van het hele mariene ecosysteem van Vietnam weer te geven. In een volgende stap werd een invers model ontwikkeld om de impact van de visserij op het functioneren van het mariene ecosysteem te bepalen. Tenslotte werden Ecopath in combinatie met Ecosim toegepast om zowel het functioneren van het mariene systeem te bepalen als tevens verschillende visserijscenarios door te rekenen, en hierbij zowel de ecologisch als economische en sociale aspecten in kaart te brengen.

Op het einde van het doctoraat worden praktische adviezen gegeven om de mariene visserij in Vietnam op een meer duurzame wijze te ontwikkelen in de toekomst.

Curriculum vitae

Personal information:

Full name: Pham Viet Anh

Date and place of birth: 21 September 1979, Hanoi (Vietnam)

Marital status: Married

Address (office): Department of Capture Fisheries and Resources Protection, Ministry of Agriculture and Rural Development, No. 10 Nguyen Cong Hoan, Ba Dinh, Ha Noi, Vietnam

Telephone (office): +84 (0) 43 37245276

E-mail: VietAnh.Pham@ugent.be

Educational history:

2010 - 2015: PhD student at Aquatic Ecology Research Unit, Department Applied Ecology and Environmental Biology, Faculty of Bioscience and Engineering, Ghent University, Jozef Plateaustraat 22, 9000 Gent, Belgium.

2006 - 2008: Master Degree in Fisheries Biology and Management at University of Bergen, Norway.

1999 - 2003: Bachelor Degree in Aquaculture and Fisheries Biology at Ha Noi University of Agriculture No. 1, Hanoi, Vietnam.

Work experiences:

2008 - now: Fisheries officer at the Department of Capture Fisheries and Resources Protection, Ministry of Agricultural and Rural Development, Ha Noi, Vietnam.

2006 - 2008: Master Degree on Fisheries Biology and Management at the Faculty of Mathematics and Natural Sciences, University of Bergen, Norway.

2003 - 2006: Researcher at the Research Institute for Marine Fisheries (RIMF), Hai Phong, Vietnam.

Scientific awards:

VIFOTEC price 2006 for young researcher awarded by Vietnamese Government (2006).

Research Grants:

Scholarship of Asian Institute for Technology for Bachelor Degree (2001-2003).

Norwegian Government Scholarship (NORAD) for Master Degree (2006-2008).

Belgian Bilateral Scholarships for Master Degree (2006 – 2008).

Belgian Bilateral Scholarships for PhD research (2010 – 2014).

Training, Workshops and Conferences:

Fisheries data collection and analysis. Research Institute for Marine Fisheries, Vietnam (2004).

Life History in Fisheries Ecology & Management. Research Institute for Marine Fisheries, Vietnam, (2005).

Biological modelling and statistics. Institute for oceanography, Vietnam (2006).

Ecosystem approaches to fisheries management. Training course is organized by cooperation project between Vietnam and Denmark. Department of Capture Fisheries and Resources Protection, Vietnam (2008).

Monitoring biodiversity on coral reef ecosystems in Vietnam. Nha Trang, Vietnam (2008).

Marine biology and marine resources utilization in Vietnam, Vietnamese Academic Institute for Science and Technology, Vietnam (2009).

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Abstract of oral presentation:

1. Pham Viet Anh, Frederik De Laender, Gert Everaert, Chu Tien Vinh and Peter Goethals (2010). Towards an ecosystem approach for Vietnam's fisheries management: A review of existing tools. Seventh International Conference on Ecological Informatics: Unravelling Complexity and Supporting Sustainability. Pand Conference Building, Ghent, Belgium, 13-16 December 2010.

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1. Pham Viet Anh, Frederik De Laender, Gert Everaert, Chu Tien Vinh and Peter Goethals (2014). An integrated food web model to test the impact of fisheries management scenarios on the coastal ecosystem of Vietnam. Ocean & Coastal Management 92: 74-86.

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Appendix 3.1. List of fishing gear types in Vietnam recorded in national vessel registration system.

Order	Fishing gear types
1	Pair of trawl including fish and shrimp trawl
2	Single trawl including fish and shrimp trawl
3	Surface gillnet (nylon net)
4	Surface gillnet (normal net)
5	Bottom drift net
6	Trammel gillnet
7	Daily purse seine
8	Light purse seine
9	Anchovy purse seine
10	Tuna purse seine
11	Fish handline
12	Squid handline
13	Oceanic squid handline
14	Tuna longline
15	Bottom longline
16	Squid lift net
18	Dredges
19	Lift net
20	Crab trap
21	Oceanic squid trap
22	Other traps (including pots, stow or bag nets, fixed traps)
23	Fixed net
24	Diving
25	Falling gears
26	Others
27	Unidentified gears

Appendix 3.2. List of references used for reconstruction of fishing effort and landing data.

Data type	**Description**	**References**
Fishing effort	Total fishing vessel from 1981-2012	Anh et al. (2014a)
	Vessel by gear types from 1996-2005, 2007, 2009-2012	Annual reports of Sub-Department of Capture Fisheries and Resources Protection (Sub-DECAFIREP) from 1996-2012
Landing data	For demersal fishes and shrimp groups between 1996-2000	ALMRV (1996), ALMRV (1997a), ALMRV (1998), ALMRV (2000)
	For demersal fishes and shrimp groups between 2001-2005	ALMRV (2001), ALMRV (2003), RIMF (2005a)
	For pelagic fishes, anchovy between 1996-1999	ALMRV (1999)
	For pelagic fishes, anchovy between 2000-2005	RIMF (2005c)
	For cephalopods, crustaceans between 1996-2005	Extracted from logbook and observer data in the database called "Vietfishbase" at Research Institute for Marine Fisheries
	For tuna, large predators between 2003-2005	RIMF (2005b)
	For all other groups	Additional data were provided by individual landing data collected by local authorities (e.g. Binh Dinh, Khanh Hoa, Nghe An, Ben Tre, Da Nang)
Catch rate	For all gears from 1996-2005	Extracted from survey database, observer database and logbook database that are available at Research Institute for Marine Fisheries (RIMF)

Appendix 3.3. Number of vessel grouped by seven main fishing gears and registered to operate entire the Vietnamese EEZ.

Year	Fish trawl	Shrimp trawl	Purse seine	Gillnet	Fish handline	Squid handline	Mixed gears
1981	6684	712	2223	5610	4404	881	9070
1982	6693	727	2206	5575	4360	876	9020
1983	6593	714	2178	5535	4328	869	8949
1984	6645	753	2210	5594	4416	877	9053
1985	6627	688	2187	5577	4369	871	8986
1986	7168	786	2381	6012	4716	944	9708
1987	8024	870	2653	6728	5263	1048	10856
1988	8073	881	2672	6762	5329	1059	10948
1989	8360	932	2782	7043	5492	1104	11282
1990	9343	1007	3102	7826	6164	1230	12625
1991	9979	1105	3283	8323	6519	1306	13439
1992	12388	1272	4103	10356	8140	1626	16728
1993	13946	1506	4628	11713	9227	1841	19028
1994	15119	1591	5024	12745	9956	2009	20599
1995	15621	1694	5169	13037	10266	2354	21166
1996	11063	2797	4795	12370	7603	3295	19060
1997	11293	3234	4887	12623	7780	3876	19413
1998	11335	3396	4906	12708	7824	4395	19502
1999	11674	3397	5018	12966	7975	4512	19906
2000	12094	3926	5205	13505	8257	4645	20641
2001	12508	4151	5403	13961	8563	4815	21415
2002	12926	4553	5607	14531	8853	4993	22238
2003	13206	4647	5702	14754	9046	5093	22620
2004	12294	4942	5706	14750	9055	5096	22550
2005	13319	4765	5749	14855	9167	5153	22834
2006	13587	4939	5854	15156	9308	5245	23148
2007	13732	6798	6920	18327	9385	5797	23570
2008	19566	10108	8411	21810	13369	7518	33383

2009	19379	9824	8397	21672	13297	7492	33287
2010	20313	8673	8243	20787	13912	7881	34840
2011	18287	8880	8814	22712	13973	7843	35032
2012	23382	9655	9349	22164	14115	7895	32245

Appendix 3.4. Catch (in tons) of trawl fisheries presented as ecological groups.

Year	Large predators	Large demersal fish	Other demersal fish	Reef fish	Large pelagic fish	Medium pelagic fish	Anchovy	Cephalopods	Shrimp	Crustaceans	Unidentified
1981	590	101005	61847	7753	3876	1555	767	2322	767	2342	13951
1982	656	112251	68732	8616	4308	1728	853	2580	853	2602	15505
1983	737	126064	77190	9676	4838	1940	958	2898	958	2923	17413
1984	576	98571	60356	7566	3783	1517	749	2266	749	2285	13615
1985	534	91359	55940	7013	3506	1406	694	2100	694	2118	12619
1986	602	103029	63086	7908	3954	1586	783	2368	783	2389	14231
1987	851	145650	89183	11180	5590	2242	1107	3348	1107	3377	20118
1988	700	119792	73350	9195	4597	1844	910	2754	910	2777	16546
1989	946	161922	99147	12429	6214	2492	1230	3722	1230	3754	22366
1990	1077	184191	112783	14138	7069	2835	1399	4234	1399	4270	25442
1991	928	158864	97275	12194	6097	2445	1207	3652	1207	3683	21943
1992	914	156425	95781	12007	6003	2407	1188	3596	1188	3626	21606
1993	1144	195768	119871	15027	7513	3013	1487	4500	1487	4539	27041
1994	1426	243972	149387	18727	9363	3755	1854	5609	1854	5656	33699
1995	1735	296843	181761	22785	11393	4569	2255	6824	2255	6882	41002
1996	920	157470	96421	12087	6044	2424	1196	3620	1196	3651	21751
1997	1309	224003	137160	17194	8597	3448	1702	5149	1702	5193	30941
1998	856	146406	89646	11238	5619	2253	1112	3366	1112	3394	20222
1999	1085	185630	113663	14249	7124	2857	1410	4267	1410	4304	25640

2000	1312	224407	137407	17225	8613	3454	1705	5159	1705	5203	30996
2001	1434	245337	150223	18832	9416	3776	1864	5640	1864	5688	33887
2002	998	170833	104603	13113	6556	2629	1298	3927	1298	3960	23596
2003	988	168993	103477	12972	6486	2601	1284	3885	1284	3918	23342
2004	1123	192157	117660	14750	7375	2957	1460	4417	1460	4455	26542
2005	1477	252688	154724	19396	9698	3889	1920	5809	1920	5858	34903
2006	1117	191065	116992	14666	7333	2941	1452	4392	1452	4430	26391
2007	1634	279624	171217	21463	10732	4304	2125	6428	2125	6483	38623
2008	1727	295500	180939	22682	11341	4548	2245	6793	2245	6851	40816
2009	1727	295561	180976	22687	11343	4549	2246	6795	2246	6852	40825
2010	1993	341081	208848	26181	13090	5249	2591	7841	2591	7907	47112
2011	2137	365711	223930	28071	14036	5629	2779	8407	2779	8478	50514
2012	1505	257444	157636	19761	9880	3962	1956	5918	1956	5968	35560

Appendix 3.5. Catch (in tons) of shrimp trawl fisheries presented as ecological groups.

Year	Large demersal fish	Other demersal fish	Reef fish	Cephalopods	Shrimp	Crustaceans	Unidentified
1981	680	1108	986	68	3814	171	415
1982	1055	1718	1528	106	5915	265	644
1983	897	1462	1301	90	5033	226	548
1984	1054	1717	1527	105	5910	265	643
1985	1193	1943	1729	119	6690	300	728
1986	899	1465	1303	90	5043	226	549
1987	1372	2236	1989	137	7696	345	837
1988	1393	2269	2019	139	7812	350	850
1989	926	1509	1342	93	5195	233	565
1990	1516	2469	2197	152	8501	381	925
1991	1785	2909	2588	179	10014	449	1089

1992	1914	3118	2774	192	10734	481	1168
1993	2204	3591	3194	221	12362	554	1345
1994	1511	2463	2191	151	8478	380	922
1995	1710	2787	2479	171	9593	430	1044
1996	4031	6568	5842	404	22609	1013	2460
1997	4276	6967	6198	428	23984	1075	2609
1998	4827	7865	6997	483	27077	1213	2946
1999	3375	5500	4892	338	18932	848	2060
2000	6027	9821	8736	603	33808	1515	3678
2001	5265	8579	7631	527	29533	1323	3213
2002	5682	9258	8236	569	31872	1428	3467
2003	6862	11181	9946	687	38489	1725	4187
2004	5569	9074	8072	558	31239	1400	3398
2005	7774	12666	11267	778	43604	1954	4744
2006	5022	8182	7278	503	28167	1262	3064
2007	6994	11397	10138	700	39232	1758	4268
2008	13588	22140	19694	1360	76217	3415	8292
2009	12122	19752	17570	1213	67995	3047	7397
2010	12180	19847	17654	1219	68321	3061	7433
2011	14243	23207	20644	1426	79891	3580	8691
2012	14022	22847	20323	1404	78651	3524	8556

Appendix 3.6. Catch (in tons) of purse seine fisheries presented as ecological groups.

Year	Tuna	Large predators	Large pelagic fish	Medium pelagic fish	Small pelagic fish	Anchovy	Cephalopods	Unidentified
1981	327	3928	9812	13042	24849	1640	654	2108
1982	259	3112	7772	10330	19683	1299	518	1670
1983	204	2448	6116	8129	15488	1022	407	1314
1984	206	2475	6183	8218	15658	1033	412	1328
1985	225	2700	6744	8964	17079	1127	449	1449
1986	331	3981	9944	13217	25183	1662	663	2136
1987	296	3562	8898	11827	22534	1487	593	1911
1988	319	3835	9580	12733	24262	1601	638	2058
1989	381	4577	11433	15196	28955	1911	762	2456
1990	443	5325	13302	17680	33687	2223	886	2858
1991	375	4508	11259	14965	28514	1882	750	2419
1992	487	5846	14603	19410	36982	2441	973	3137
1993	468	5625	14051	18676	35584	2349	936	3018
1994	672	8080	20183	26826	51114	3374	1345	4336
1995	737	8857	22124	29405	56028	3698	1474	4753
1996	418	5019	12538	16664	31752	2096	835	2693
1997	423	5089	12711	16895	32190	2125	847	2731
1998	617	7411	18511	24604	46879	3094	1233	3977
1999	553	6650	16612	22079	42068	2777	1107	3568
2000	666	8009	20004	26588	50659	3344	1333	4297
2001	546	6560	16387	21780	41499	2739	1092	3520
2002	743	8926	22296	29634	56463	3727	1486	4790
2003	845	10158	25374	33724	64257	4241	1691	5451
2004	483	5807	14505	19279	36734	2424	966	3116
2005	594	7142	17840	23711	45178	2982	1189	3832

2006	707	8492	21212	28193	53718	3545	1413	4557
2007	823	9894	24714	32848	62587	4131	1647	5309
2008	979	11771	29401	39078	74457	4914	1959	6316
2009	1114	13384	33432	44435	84665	5588	2228	7182
2010	693	8323	20789	27631	52646	3475	1385	4466
2011	712	8557	21374	28408	54128	3572	1424	4591
2012	708	8507	21250	28244	53816	3552	1416	4565

Appendix 3.7. Catch (in tons) of gillnet fisheries presented as ecological groups.

Year	Tuna	Large predators	Large pelagic fish	Medium pelagic fish	Small pelagic fish	Unidentified
1981	21541	29316	66964	56988	90595	13265
1982	18299	24904	56885	48410	76960	11268
1983	18402	25044	57205	48683	77392	11332
1984	22265	30301	69215	58903	93640	13710
1985	20114	27374	62529	53214	84595	12386
1986	19216	26152	59737	50837	80818	11833
1987	21227	28889	65988	56157	89275	13071
1988	24735	33663	76893	65438	104028	15231
1989	19578	26644	60861	51794	82338	12056
1990	26781	36447	83254	70851	112633	16491
1991	25417	34590	79012	67240	106894	15651
1992	32271	43918	100318	85373	135720	19872
1993	37667	51262	117094	99649	158416	23195
1994	29158	39682	90643	77139	122630	17955
1995	31698	43138	98537	83857	133310	19519
1996	50052	68117	155595	132414	210503	30821
1997	30476	41475	94738	80624	128171	18766

1998	32959	44856	102460	87196	138617	20296
1999	43000	58520	133672	113757	180843	26478
2000	40737	55440	126637	107771	171327	25085
2001	33021	44939	102651	87358	138875	20334
2002	35172	47867	109338	93049	147923	21658
2003	33896	46131	105373	89674	142558	20873
2004	59790	81371	185868	158178	251460	36818
2005	53788	73202	167210	142299	226217	33122
2006	45182	61490	140456	119531	190022	27822
2007	52710	71735	163859	139447	221684	32458
2008	84965	115631	264127	224777	357335	52320
2009	68929	93808	214277	182354	289894	42445
2010	56534	76939	175746	149563	237765	34813
2011	88506	120451	275136	234146	372230	54501
2012	114075	155249	354622	301791	479766	70246

Appendix 3.8. Catch (in tons) of fish handline fisheries presented as ecological groups.

Year	Other demersal fish	Large pelagic fish	Small pelagic fish	Unidentified
1981	10603	5289	5828	730
1982	14846	7405	8160	1022
1983	12645	6308	6950	870
1984	16847	8404	9260	1159
1985	17225	8593	9468	1185
1986	11551	5762	6349	795
1987	18297	9127	10057	1259
1988	14300	7133	7860	984
1989	13322	6645	7322	917

1990	19686	9820	10820	1355
1991	24712	12327	13583	1700
1992	24237	12090	13322	1668
1993	29629	14780	16286	2039
1994	38419	19165	21117	2644
1995	28870	14402	15869	1987
1996	18800	9378	10333	1294
1997	27493	13714	15111	1892
1998	25509	12725	14021	1755
1999	19115	9535	10507	1315
2000	18563	9260	10203	1277
2001	31664	15795	17404	2179
2002	21598	10774	11871	1486
2003	24441	12192	13434	1682
2004	20638	10295	11344	1420
2005	33375	16649	18345	2297
2006	30634	15281	16838	2108
2007	36761	18338	20206	2530
2008	49625	24755	27276	3415
2009	36902	18408	20283	2539
2010	32824	16374	18042	2259
2011	46902	23397	25780	3227
2012	46083	22988	25329	3171

Appendix 3.9. Catch (in tons) of squid handline fisheries presented as ecological groups.

Year	Large pelagic fish	Medium pelagic fish	Cephalopods	Unidentified
1981	128	184	2801	93

1982	117	169	2568	86
1983	196	284	4307	143
1984	137	199	3018	101
1985	201	291	4422	147
1986	227	328	4978	166
1987	249	360	5470	182
1988	156	226	3432	114
1989	208	301	4576	152
1990	270	389	5916	197
1991	301	435	6615	220
1992	315	455	6909	230
1993	274	395	6005	200
1994	464	671	10191	339
1995	412	595	9047	301
1996	638	922	14010	467
1997	856	1236	18785	626
1998	965	1394	21181	706
1999	633	914	13892	463
2000	923	1333	20249	675
2001	834	1204	18298	610
2002	1059	1530	23247	774
2003	1029	1486	22579	752
2004	973	1406	21361	712
2005	675	976	14822	494
2006	1182	1708	25952	864
2007	710	1026	15594	519
2008	1316	1901	28888	962
2009	1363	1969	29918	997
2010	1277	1845	28024	933

2011	1065	1538	23373	779
2012	1157	1671	25385	846

Appendix 3.10. Catch (in tons) of mixed gears presented as ecological groups.

Year	Reef fish	Small pelagic fish	Cephalopods	Unidentified
1981	31487	38933	11039	8145
1982	24684	30521	8654	6385
1983	32227	39848	11299	8336
1984	28513	35256	9997	7376
1985	26753	33079	9379	6920
1986	25005	30918	8767	6468
1987	33975	42009	11911	8789
1988	30954	38274	10852	8007
1989	44333	54817	15543	11468
1990	32980	40779	11563	8531
1991	57078	70576	20012	14765
1992	55509	68636	19461	14359
1993	57069	70564	20008	14763
1994	90154	111473	31608	23321
1995	68581	84799	24044	17740
1996	53644	66330	18808	13877
1997	76856	95031	26945	19881
1998	69409	85823	24335	17955
1999	75504	93360	26472	19531
2000	65607	81122	23002	16971
2001	92001	113758	32255	23799
2002	97254	120253	34097	25158

2003	68390	84563	23977	17691
2004	73384	90738	25728	18983
2005	62855	77719	22037	16259
2006	62406	77164	21879	16143
2007	89035	110090	31215	23031
2008	131908	163102	46247	34122
2009	126027	155831	44185	32601
2010	103829	128383	36402	26858
2011	106738	131980	37422	27611
2012	97068	120023	34032	25110

Appendix 4.1. Catch (tons) presented as ecological groups and used to calculate trophic indices. TUN = Tuna, LAR = Large predators, LAD = Large demersal fish, OTD = Other demersal fish, REF = Reef fish, LAP = Other pelagic fish, MEP = Medium pelagic fish, SMP = Small pelagic fish, ANC = Anchovy, CEP = Cephalopods, SHR = Shrimp and CRU = Crustaceans.

Year	TUN	LAR	LAD	OTD	REF	LAP	MEP	SMP	ANC	CEP	SHR	CRU
1981	20945	32619	103460	79079	49809	84571	79672	145099	2357	18496	4705	2397
1982	17637	27636	115680	91702	42049	75159	67284	123612	2108	15574	7007	2742
1983	17800	27242	129568	98073	53012	73056	65570	124451	1930	20652	6172	3004
1984	21507	31982	101802	84565	46414	86550	76788	138796	1742	17216	6892	2429
1985	19403	29394	94268	80011	43483	80270	71190	131274	1778	18006	7698	2300
1986	18569	29583	105964	81582	41946	78082	72993	131530	2384	18226	6018	2510
1987	20549	32078	149143	116994	57400	88055	78132	147343	2528	23230	9086	3559
1988	23857	36630	123296	96050	51685	96623	89286	158293	2461	19366	9076	2994
1989	19067	31038	166883	122790	71252	83559	77182	154309	3067	26891	6618	3807
1990	25950	41304	189140	145596	59173	112096	101549	180536	3528	24488	10191	4428
1991	24604	38627	163287	134016	89048	107027	94669	194214	3022	34148	11652	3936
1992	31361	48901	161530	131132	87016	130864	118888	228635	3546	34007	12400	3922
1993	36425	55626	201886	164036	92864	151111	135659	253187	3734	34412	14340	4867
1994	28567	47418	251210	204010	138561	137608	118184	268480	5093	53311	10609	5768
1995	30905	51862	304549	228967	114424	144492	129942	260136	5794	45169	12141	6987
1996	47884	70770	164917	130005	87353	180429	169717	289435	3209	41127	24745	4439
1997	29468	46342	231711	183512	123340	127755	112520	238197	3727	56832	26689	5978
1998	32003	51342	154627	130736	108314	137733	126509	255100	4108	55248	29432	4371
1999	41378	63776	193199	147956	117411	164340	154679	293124	4086	50437	21137	4931
2000	39518	62562	234807	176979	111338	162170	153083	282852	4922	54556	36944	6387
2001	31981	51021	255681	205061	145784	142383	124938	272776	4483	62943	32703	6684
2002	34144	55570	179484	144421	148021	146853	138740	296599	4915	69449	34626	5119
2003	33189	55332	179557	147897	111503	147114	139469	275581	5378	57456	41732	5340
2004	57523	84496	201105	156927	118050	214606	203680	352972	3783	57647	34023	5569
2005	51819	78864	264394	213194	111941	207439	189435	335143	4785	48077	47271	7393

2006	43861	68279	199987	166618	102561	181857	168713	309783	4862	58742	30774	5415
2007	51086	80076	291740	235375	147256	214508	196343	374787	6110	60565	42691	7837
2008	81617	123897	313913	268813	214965	324086	301697	560556	6988	93328	81545	9740
2009	66587	104843	312876	253882	203874	273194	257167	495112	7636	91877	73161	9396
2010	54626	83904	360674	278707	177446	222868	203389	389704	5897	81465	73930	10384
2011	55073	86075	386196	313243	186820	327550	300991	531551	6219	78337	86296	11421
2012	49579	82709	275778	240047	165249	400010	375993	617638	5341	74001	84049	8972

Appendix 4.2. Some properties of ecological indicators.

Indicator type	**Data demand**	**Properties**	**Advantages**	**Disadvantages**	**References**
Mean trophic level	- Landing data of species/groups to be caught in entire ecosystem, - Trophic level of species/groups to be caught	- Increasing trend indicating fisheries focusing on high trophic levels - Decreasing trend indicating a decline of high trophic levels and an increase of low trophic levels and can be a signal of fishing down the food web	- Easy to calculate - Easy to interpret	Do not consider fisheries expansion and eutrophication	Christensen (1998) Pang and Pauly (2001) Pauly and Palomares (2005) Bhathal and Pauly (2008) Babouri et al. (2014)
Fishing in Balance	- Landing data of species/groups to be caught in entire ecosystem, - Trophic level of species/groups to be caught - Transfer efficiency for each trophic level	- Increasing if catches increase faster than would be predicted by TL declines - Decreasing if an increase in catches fails to compensate for a decrease in TL	Can consider fisheries expansion and eutrophication	Can only be proof when ecosystems are being degraded by fishing	Pennino et al. (2011) Babouri et al. (2014)
Pelagic/demersal (P/D) ratio	- Landing data of pelagic species/groups - Landing data of demersal species/groups	Increasing trend of P/D indicating fishing down the food web	Can be used to partly evaluate changes on structure and functioning of ecosystem	Too subjective to interpret fisheries' general status	Caddy et al. (1998) Pennino and Bellido (2012) Babouri et al. (2014)

Appendix 4.3. Testing of the correlation significance between landings and fishery-based indicators. MTI: Marine trophic index, FiB: Fishing in Balance and P/D: Pelagic/Demersal ratio.

	Landings and MTI	Landings and FIB	Landings and P/D ratio
Correlation coefficient (r-absolute)	0.610	0.950	0.080
t-value	4.216	16.664	0.440
p-value	0.000	0.000	0.663

Appendix 5.1. Overview of data sources of ecological groups used in the inverse models for the 1990s and 2000s period in the Vietnamese coastal ecosystem. N/A = not available (these are estimated from the models).

No.	Ecological Groups	Code	Biomass data sources		Diet matrix
			Model in the 1990s	Model in the 2000s	All models
1	Mammal	MAM	N/A	Chen et al. (2008a)	Chen et al. (2008a)
2	Sea turtle	SEA	N/A	Chen et al. (2008b)	Griffiths et al. (2010)
3	Large predators	LAR	Christensen et al. (2003)	Large pelagic surveys from 2003-2005	Chen et al. (2008a)
4	Tuna	TUN	Christensen et al. (2003)	Large pelagic surveys from 2003-2005	Griffiths et al. (2010)
5	Medium pelagic fishes	MEP	Acoustic survey in 1999	Acoustic survey in 2003-2005	Froese and Pauly (2006)
6	Small pelagic fishes	SMP	Acoustic survey in 1999	Acoustic survey in 2003-2005	Froese and Pauly (2006)
7	Other pelagic fishes	LAP	Acoustic survey in 1999	Acoustic survey in 2003-2005	Froese and Pauly (2006)
8	Cephalopods	CEP	Trawl surveys from 1996-1999	Trawl surveys from 2003-2005	Chen et al. (2008a)
9	Large demersal fish	LAD	Trawl surveys from 1996-1999	Trawl surveys from 2003-2005	Chen et al. (2008a)
10	Other demersal fish	OTD	Trawl surveys from 1996-1999	Trawl surveys from 2003-2005	Chen et al. (2008a)
11	Shrimp	SHR	Trawl surveys from 1996-1999	Trawl surveys from 2003-2005	Wang et al. (2012)
12	Crustaceans	CRU	Trawl surveys from 1996-1999	Trawl surveys from 2003-2005	Chen et al. (2008a)
13	Zoobenthos	ZOB	Duana et al. (2009)	Chen et al. (2008a)	Griffiths et al. (2010)
14	Zooplankton	ZOP	Duana et al. (2009)	Chen et al. (2008b)	Griffiths et al. (2010)
15	Phytoplankton	PHY	Duana et al. (2009)	Van et al. (2010)	N/A
16	Bacteria	BAC	N/A	N/A	N/A
17	Detritus	DET	N/A	N/A	N/A
18	Dissolved organic carbon	DOC	N/A	N/A	N/A

Appendix 5.2. Food web matrix indicating the links between food web compartments where rows represent recipients and columns represent the senders. DIC: dissolve inorganic carbon, CATCH: export by fisheries, and MOR: natural mortality other than predation mortality is considered as an external compartment. Reference sources and other abbreviations (belonging to internal compartments) are shown in the Appendix 5.1.

		to																				
		MAM	SEA	LAR	TUN	MEP	SMP	LAP	CEP	LAD	OTD	SHR	CRU	ZOB	ZOP	PHY	DET	DOC	BAC	DIC	CATCH	MOR
from	MAM	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	1	1	1
	SEA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	1	1	1
	LAR	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	1	1	1
	TUN	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	1	1	1
	MEP	1	0	1	1	1	0	1	0	0	0	0	0	0	0	0	1	1	0	1	1	1
	SMP	1	0	1	1	1	0	1	0	1	1	0	0	0	0	0	1	1	0	1	1	1
	LAP	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	1	1	0	1	1	1
	CEP	0	0	0	1	0	0	1	0	0	0	1	0	0	0	0	1	1	0	1	1	1
	LAD	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	1	1	1
	OTD	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	1	1	0	1	1	1
	SHR	1	1	0	1	1	1	1	0	1	1	0	0	0	0	0	1	1	0	1	1	1
	CRU	1	1	1	1	0	0	0	0	1	1	0	0	0	0	0	1	1	0	1	1	1
	ZOB	0	0	0	0	0	0	0	0	1	1	0	1	1	0	0	1	1	0	1	0	1
	ZOP	1	1	1	1	1	1	1	1	1	1	1	0	1	1	0	1	1	0	1	0	1
	PHY	0	0	1	0	1	1	1	1	1	1	0	1	0	1	0	0	1	0	1	0	1
	DET	0	0	0	0	0	1	1	0	1	0	1	1	1	0	0	0	1	0	0	0	0
	DOC	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
	BAC	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	1	0	1
	DIC	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
	CATCH	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	MOR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0

Appendix 5.3. Ranges (minimum and maximum, t·C·km^{-2}·year^{-1}) for all flows of the coastal marine food web models. DIC: dissolved inorganic carbon, CATCH: export by fisheries, and MOR: natural mortality other than predation mortality is considered as an external compartment. Other abbreviations (belonging to internal compartments) are shown in the Appendix 5.1.

	1990s		2000s			1990s		2000s	
Flow	min	max	min	max	Flow	min	max	min	max
MOR->DET	0.000E+00	7.692E+02	0.00E+00	6.45E+02	CEP->DOC	0.000E+00	1.531E-02	0.00E+00	1.20E-02
DOC->BAC	1.385E+01	8.400E+02	8.92E+00	6.75E+02	CEP->DIC	1.115E+00	1.130E+00	8.55E-01	8.67E-01
BAC->ZOP	2.743E+00	1.663E+02	1.77E+00	1.34E+02	CEP->DET	1.393E-01	1.566E-01	1.07E-01	1.20E-01
BAC->DIC	5.541E+00	3.360E+02	3.57E+00	2.70E+02	CEP->CATCH	2.600E-02	1.360E-01	4.20E-02	8.30E-02
BAC->DOC	1.828E+00	1.109E+02	1.18E+00	8.91E+01	CEP->MOR	0.000E+00	1.531E-02	0.00E+00	1.20E-02
BAC->MOR	0.000E+00	2.268E+02	0.00E+00	1.82E+02	SMP->MAM	0.000E+00	2.857E-01	0.00E+00	5.75E-01
DET->SHR	0.000E+00	3.587E+00	0.00E+00	5.48E+00	SMP->TUN	0.000E+00	2.857E-01	0.00E+00	5.75E-01
DET->CRU	0.000E+00	2.127E+00	0.00E+00	4.22E+00	SMP->LAR	0.000E+00	2.857E-01	0.00E+00	5.75E-01
DET->ZOB	0.000E+00	7.475E+01	0.00E+00	7.90E+01	SMP->LAD	0.000E+00	2.857E-01	0.00E+00	4.95E-01
DET->LAD	0.000E+00	3.988E-01	0.00E+00	4.95E-01	SMP->OTD	0.000E+00	2.857E-01	0.00E+00	5.75E-01
DET->LAP	0.000E+00	2.037E+00	0.00E+00	3.39E+00	SMP->MEP	0.000E+00	2.857E-01	0.00E+00	5.75E-01
DET->SMP	0.000E+00	9.923E-01	0.00E+00	1.95E+00	SMP->LAP	0.000E+00	2.857E-01	0.00E+00	5.75E-01
DET->DOC	0.000E+00	7.171E+02	0.00E+00	5.78E+02	SMP->DOC	1.840E-02	1.488E-01	3.61E-02	2.92E-01
DIC->PHY	1.936E+02	5.807E+02	1.94E+02	5.82E+02	SMP->DIC	1.942E-01	2.453E-01	3.81E-01	4.82E-01
PHY->CEP	0.000E+00	1.413E+00	0.00E+00	1.08E+00	SMP->DET	3.679E-02	2.977E-01	7.23E-02	5.85E-01
PHY->CRU	0.000E+00	2.127E+00	0.00E+00	4.22E+00	SMP->CATCH	1.200E-02	6.800E-02	1.00E-02	4.10E-02
PHY->ZOP	0.000E+00	1.632E+02	0.00E+00	1.32E+02	SMP->MOR	2.240E-03	2.800E-03	4.40E-03	5.50E-03
PHY->DIC	9.678E+00	1.742E+02	9.70E+00	1.75E+02	MEP->MAM	0.000E+00	3.348E-01	0.00E+00	9.94E-02
PHY->DOC	6.774E+00	3.310E+02	6.79E+00	3.32E+02	MEP->LAR	0.000E+00	3.348E-01	0.00E+00	9.94E-02
PHY->MOR	0.000E+00	5.240E+02	0.00E+00	5.25E+02	MEP->TUN	0.000E+00	3.348E-01	0.00E+00	9.94E-02
ZOP->MAM	0.000E+00	5.475E-01	0.00E+00	6.69E+01	MEP->MEP	0.000E+00	3.348E-01	0.00E+00	9.94E-02
ZOP->SEA	0.000E+00	8.316E+01	0.00E+00	6.69E+01	MEP->LAP	0.000E+00	3.348E-01	0.00E+00	9.94E-02
ZOP->MEP	0.000E+00	1.293E+00	0.00E+00	9.98E-01	MEP->DOC	0.000E+00	8.454E-01	0.00E+00	6.26E-02
ZOP->SMP	0.000E+00	9.923E-01	0.00E+00	1.95E+00	MEP->DIC	1.760E-01	1.021E+00	7.26E-01	7.88E-01

ZOP->LAP	0.000E+00	2.037E+00	0.00E+00	3.39E+00	MEP->DET	4.500E-02	3.878E-01	9.19E-02	1.62E-01
ZOP->CEP	0.000E+00	1.413E+00	0.00E+00	1.08E+00	MEP->CATCH	5.300E-02	1.360E-01	6.30E-02	1.05E-01
ZOP->LAD	0.000E+00	3.988E-01	0.00E+00	4.95E-01	MEP->MOR	1.288E-02	1.564E-02	9.94E-03	1.21E-02
ZOP->OTD	0.000E+00	9.358E-01	0.00E+00	1.25E+00	LAP->LAR	0.000E+00	5.990E-01	0.00E+00	1.01E+00
ZOP->SHR	0.000E+00	3.587E+00	0.00E+00	5.48E+00	LAP->LAP	0.000E+00	5.990E-01	0.00E+00	1.01E+00
ZOP->ZOB	0.000E+00	7.475E+01	0.00E+00	6.69E+01	LAP->DOC	0.000E+00	1.622E+00	0.00E+00	2.70E+00
ZOP->ZOP	0.000E+00	8.316E+01	0.00E+00	6.69E+01	LAP->DIC	0.000E+00	1.622E+00	0.00E+00	2.70E+00
ZOP->DOC	5.231E+00	2.596E+01	9.14E-01	7.02E+00	LAP->DET	6.570E-03	6.110E-01	1.10E-02	1.02E+00
ZOP->DIC	1.585E+01	2.596E+01	2.77E+00	7.02E+00	LAP->CATCH	1.200E-02	4.800E-02	1.10E-02	3.90E-02
ZOP->DET	3.243E+00	8.316E+01	1.74E+00	6.69E+01	LAP->MOR	6.900E-03	7.800E-03	1.15E-02	1.30E-02
ZOP->MOR	0.000E+00	8.703E+01	0.00E+00	8.32E+01	OTD->LAR	0.000E+00	2.537E-01	0.00E+00	3.46E-01
ZOB->LAD	0.000E+00	3.988E-01	0.00E+00	4.95E-01	OTD->LAD	0.000E+00	2.537E-01	0.00E+00	3.46E-01
ZOB->OTD	0.000E+00	9.358E-01	0.00E+00	1.25E+00	OTD->DOC	0.000E+00	7.450E-01	0.00E+00	9.93E-01
ZOB->CRU	0.000E+00	2.127E+00	0.00E+00	4.22E+00	OTD->DIC	0.000E+00	7.450E-01	0.00E+00	9.93E-01
ZOB->ZOB	0.000E+00	6.727E+01	0.00E+00	7.11E+01	OTD->DET	9.000E-03	2.807E-01	9.33E-03	3.74E-01
ZOB->DOC	0.000E+00	6.727E+01	0.00E+00	7.11E+01	OTD->CATCH	2.700E-02	1.200E-01	2.80E-02	7.90E-02
ZOB->DIC	0.000E+00	6.727E+01	0.00E+00	7.11E+01	OTD->MOR	3.600E-03	4.200E-03	4.80E-03	5.60E-03
ZOB->DET	1.601E+00	2.242E+01	1.69E+00	2.37E+01	LAD->DOC	0.000E+00	4.370E-03	0.00E+00	1.54E-01
ZOB->MOR	0.000E+00	6.727E+01	0.00E+00	7.11E+01	LAD->DIC	2.966E-01	3.010E-01	2.15E-01	3.69E-01
CRU->MAM	0.000E+00	5.475E-01	0.00E+00	1.21E+00	LAD->DET	3.933E-02	4.425E-02	3.92E-02	1.49E-01
CRU->SEA	0.000E+00	6.082E-01	0.00E+00	1.21E+00	LAD->CATCH	3.933E-02	4.425E-02	5.40E-02	1.05E-01
CRU->TUN	0.000E+00	6.082E-01	0.00E+00	1.21E+00	LAD->MOR	1.805E-02	2.090E-02	2.24E-02	2.60E-02
CRU->LAR	0.000E+00	6.082E-01	0.00E+00	1.21E+00	LAR->LAR	0.000E+00	5.859E-01	8.60E-02	6.57E-01
CRU->LAD	0.000E+00	3.988E-01	0.00E+00	4.95E-01	LAR->DOC	0.000E+00	1.491E+00	0.00E+00	4.79E-01
CRU->OTD	0.000E+00	6.082E-01	0.00E+00	1.21E+00	LAR->DIC	1.618E-01	3.534E-01	1.47E+00	1.95E+00
CRU->DOC	0.000E+00	1.681E+00	0.00E+00	3.33E+00	LAR->DET	2.176E-02	6.199E-01	1.84E-01	7.23E-01
CRU->DIC	0.000E+00	1.681E+00	0.00E+00	3.33E+00	LAR->CATCH	3.400E-02	1.570E-01	6.60E-02	9.80E-02
CRU->DET	1.222E-01	6.382E-01	2.42E-01	1.27E+00	LAR->MOR	0.000E+00	1.491E+00	0.00E+00	4.79E-01
CRU->CATCH	3.000E-02	1.380E-01	6.10E-02	1.00E-01	TUN->MAM	0.000E+00	5.475E-01	0.00E+00	9.47E-01
CRU->MOR	2.108E-02	3.596E-02	4.18E-02	7.13E-02	TUN->LAR	0.000E+00	6.980E-01	0.00E+00	1.06E+00
SHR->MAM	0.000E+00	5.475E-01	0.00E+00	1.64E+00	TUN->DOC	0.000E+00	1.454E+00	0.00E+00	2.46E+00

SHR->SEA	0.000E+00	1.074E+00	0.00E+00	1.64E+00	TUN->DIC	4.923E-01	8.476E-01	3.87E-01	2.85E+00
SHR->TUN	0.000E+00	1.074E+00	0.00E+00	1.64E+00	TUN->DET	6.154E-02	7.300E-01	4.84E-02	1.08E+00
SHR->LAD	0.000E+00	3.988E-01	0.00E+00	4.95E-01	TUN->CATCH	3.200E-02	7.800E-02	2.10E-02	6.70E-02
SHR->OTD	0.000E+00	9.358E-01	0.00E+00	1.25E+00	TUN->MOR	0.000E+00	1.454E+00	0.00E+00	2.46E+00
SHR->LAP	0.000E+00	1.074E+00	0.00E+00	1.64E+00	SEA->DOC	0.000E+00	7.576E+01	0.00E+00	6.18E+01
SHR->MEP	0.000E+00	1.074E+00	0.00E+00	9.98E-01	SEA->DIC	0.000E+00	7.576E+01	0.00E+00	6.18E+01
SHR->SMP	0.000E+00	9.923E-01	0.00E+00	1.64E+00	SEA->DET	6.667E-04	2.525E+01	0.00E+00	2.06E+01
SHR->DOC	0.000E+00	2.500E+00	0.00E+00	2.87E+00	SEA->CATCH	6.000E-03	1.500E-02	0.00E+00	6.18E+01
SHR->DIC	9.108E-03	5.054E-02	9.64E-01	3.83E+00	SEA->MOR	0.000E+00	7.576E+01	0.00E+00	6.18E+01
SHR->DET	1.243E-02	1.076E+00	2.76E-01	1.64E+00	MAM->DOC	0.000E+00	4.922E-01	0.00E+00	6.17E+01
SHR->CATCH	2.000E-03	4.100E-02	8.00E-03	2.80E-02	MAM->DIC	0.000E+00	4.922E-01	0.00E+00	6.17E+01
SHR->MOR	1.800E-03	2.160E-03	2.75E-03	3.30E-03	MAM->DET	1.226E-02	1.643E-01	0.00E+00	2.06E+01
CEP->TUN	0.000E+00	1.306E-01	0.00E+00	7.85E-02	MAM->CATCH	1.226E-04	5.475E-03	0.00E+00	6.86E-01
CEP->SHR	0.000E+00	1.306E-01	0.00E+00	7.85E-02	MAM->MOR	0.000E+00	4.922E-01	0.00E+00	6.17E+01
CEP->LAP	0.000E+00	1.306E-01	0.00E+00	7.85E-02					

Appendix 5.4. Inequalities for the Vietnamese coastal ecosystem models between two time periods, where Min and Max are minimum and maximum of constraint values and S_X denotes the standing stock of X ($t{\cdot}C{\cdot}km^{-2}$). DIC: dissolve inorganic carbon, CATCH: export by fisheries, and MOR: natural mortality other than predation mortality is considered as an external compartment. Other abbreviations (belonging to internal compartments) are shown in the Appendix 5.1.

Constraints of consumption:

MAMCon= [MinMAMCon, MaxMAMCon] * S_MAM (only for model in the 2000s)

SEACon = [MinSEACon, MaxSEACon] * S_SEA (only for model in the 2000s)

LARCon = [MinLARCon, MaxLARCon] * S_LAR

SMPCon = [MinSMPCon, MaxSMPCon] * S_SMP

TUNCon = [MinTUNCon, MaxTUNCon] * S_TUN

MEPCon = [MinMEPCon, MaxMEPCon] * S_MEP

LAPCon = [MinLAPCon, MaxLAPCon] * S_LAP

CEPCon = [MinCEPCon, MaxCEPCon] * S_CEP

LADCon = [MinLADCon, MaxLADCon] * S_LAD

OTDCon = [MinOTDCon, MaxOTDCon] * S_OTD

SHRCon = [MinSHRCon, MaxSHRCon] * S_SHR

CRUCon = [MinCRUCon, MaxCRUCon] * S_CRU

ZOBCon = [MinZOBCon, MaxZOBCon] * S_ZOB

ZOPCon = [MinZOPCon, MaxZOPCon] * S_ZOP

Constraints of gross efficiency:

MAMPro = [MinMAMGroeff, MaxMAMGroeff] * MAMCon

LARPro = [MinLARGroeff, MaxLARGroeff] * LARCon

TUNPro = [MinTUNGroeff, MaxTUNGroeff] * TUNCon

MEPPro = [MinMEPGroeff, MaxMEPGroeff] * MEPCon

SMPPro = [MinSMPGroeff, MaxSMPGroeff] * SMPCon

LAPPro = [MinLAPGroeff, MaxLAPGroeff] * LAPCon

CEPPro = [MinCEPGroeff, MaxCEPGroeff] * CEPCon

LADPro = [MinLADGroeff, MaxLADGroeff] * LADCon

OTDPro = [MinOTDGroeff, MaxOTDGroeff] * OTDCon

SHRPro = [MinSHRGroeff, MaxSHRGroeff] * SHRCon

CRUPro = [MinCRUGroeff, MaxCRUGroeff] * CRUCon

ZOPPro = [MinZOPGroeff, MaxZOPGroeff] * ZOPCon

Constraints of assimilation efficiency:

MAMCon - MAMEgest = [MinMAMAss, MaxMAMAss] * MAMCon

SEACon - SEAEgest = [MinSEAAss, MaxSEAAss] * SEACon

LARCon - LAREgest = [MinLARAss, MaxLARAss] * LARCon

TUNCon - TUNEgest = [MinTUNAss, MaxTUNAss] * TUNCon

MEPCon - MEPEgest = [MinMEPAss, MaxMEPAss] * MEPCon

SMPCon - SMPEgest = [MinSMPAss, MaxSMPAss] * SMPCon

LAPCon - LAPEgest = [MinLAPAss, MaxLAPAss] * LAPCon

CEPCon - CEPEgest = [MinCEPAss, MaxCEPAss] * CEPCon

LADCon - LADEgest = [MinLADAss, MaxLADAss] * LADCon

OTDCon - OTDEgest = [MinOTDAss, MaxOTDAss] * OTDCon

SHRCon - SHREgest = [MinSHRAss, MaxSHRAss] * SHRCon

CRUCon - CRUEgest = [MinCRUAss, MaxCRUAss] * CRUCon

ZOBCon - ZOBEgest = [MinZOBAss, MaxZOBAss] * ZOBCon

ZOPCon - ZOPEgest = [MinZOPAss, MaxZOPAss] * ZOPCon

Constraints of excretion:

PHYExcr = [MinPHYExcr, MaxPHYExcr] * PHYNpp

ZOPExcr = [MinZOPExcr, MaxZOPExcr] * ZOPRes

SMPExcr = [MinSMPExcr, MaxSMPExcr] * SMPIngs

BACExcr = [MinBACExcr, MaxBACExcr] * BACRes

Constraint of gross primary production

PHYGpp = [MinPHYGpp, MaxPHYGpp] * S_PHY

Constraints of respiration

PHYRes = [MinPHYRes, MaxPHYRes] * PHYGpp

ZOPRes = [MinZOPRes, MaxZOPRes] * S_ZOP

CEPRes = [MinCEPRes, MaxCEPRes] * S_CEP

SMPRes = [MinSMPResmat, MaxSMPResmat] * S_SMP

LADRes = [MinLADRes, MaxLADRes] * S_LAD

MEPRes = [MinMEPRes, MaxMEPRes] * S_MEP

TUNRes = [MinTUNRes, MaxTUNRes] * S_TUN

SHRRes = [MinSHRRes, MaxSHRRes] * S_SHR

BACRes = BACUptake - [MaxBACgroeff, MinBACgroeff] * BACUptake

LARRes = [MinLARRes, MaxLARRes] * S_LAR

MAMRes = [MinMAMRes, MaxMAMRes] * S_MAM (only for the model in the 2000s)

Constraints of ingestion

SMPIngs = [MinSMPIngs, MaxSMPIngs] * S_SMP

ZOPIngs = [MinZOPIngs, MaxZOPIngs] * S_ZOP

SHRIngs = [MinSHRIngs, MaxSHRIngs] * S_SHR

BACViral = [MinBACvir,MaxBACvir] * BACPro

Constraints of natural mortality rates

LAPMor = [MinLAPMor, MaxLAPMor] * S_LAP

MEPMor = [MinMEPMor, MaxMEPMor] * S_MEP

SMPMor = [MinSMPMor, MaxSMPMor] * S_SMP

SHRMor = [MinSHRMor, MaxSHRMor] * S_SHR

CRUMor = [MinCRUMor, MaxCRUMor] * S_CRU

OTDMor = [MinOTDMor, MaxOTDMor] * S_OTD

LADMor = [MinLADMor, MaxLADMor] * S_LAD

Constraints of fisheries landings

SEAExp = [MinSEAExp, MaxSEAExp] (only for the model in the 2000s)

TUNExp = [MinTUNExp, MaxTUNExp]

LARExp = [MinLARExp, MaxLARExp]

LADExp = [MinLADExp, MaxLADExp]

OTDExp = [MinOTDExp, MaxOTDExp]

LAPExp = [MinLAPExp, MaxLAPExp]

MEPExp = [MinMEPExp, MaxMEPExp]

SMPExp = [MinSMPExp, MaxSMPExp]

CEPExp = [MinCEPExp, MaxCEPExp]

SHRExp = [MinSHRExp, MaxSHRExp]

CRUExp = [MinCRUExp, MaxCRUExp]

Appendix 5.5. Formula to calculate functional food web indices and their code used in the study.

Order	Index name	Code	Description of index	Formula	Reference
1	Total system throughput	TSTP	TSTP is obtained by summing all flows in and out the system	$\sum_{i=1}^{n+2}\sum_{j=0}^{n} T_{ij}$	Latham (2006)
2	Ascendency	A	A is calculated by multiplying the average mutual information and TSTP	$AMI\ x\ TSTP = \sum_{i=1}^{n+2}\sum_{j=0}^{n} T_{ij}\ log_2 \frac{T_{ij} \cdot TSTP}{T_{i.} \cdot T_{.j}}$	Latham (2006)
3	Development capacity	DC	DC represents the maximum potential evenness and diversity of flows of the system in terms of all its flows	$-\sum_{i=1}^{n+2}\sum_{j=0}^{n} T_{ij}\ log_2 \frac{T_{ij}}{TSTP}$	Ulanowicz (2004)
4	Overhead	O	Overhead is calculated by difference between development capacity and ascendency	$DC - A$	Ulanowicz (2004)
5	Ratio between A/DC	A/DC	A/DC is a measure for the system's development	A/DC	Latham (2006)
6	Constraint efficiency	CE	CE is a measure of the degree of inherent network constraints to maximum network uncertainty	$\frac{\sum_{i=1}^{n} \log_2(n+2) - [-\sum_{i=1}^{n+2}\sum_{j=1}^{n} \frac{T_{ij}}{TSTP} \log_2 \frac{T_{ij}}{T_{.j}}}{H_{max}}$	Latham (2006)
7	Finn's cycling index	FCI	FCI is the fraction of all the flow in the system which is being cycled	$\frac{\sum_{j=1}^{n}(1-\frac{1}{q_{ij}}) \cdot T_j}{TST}$	Patten and Higashi (1984)
8	Average mutual information	AMI	AMI is a measure of the average amount of constraint placed upon an arbitrary unit of flow anywhere in the network	$\sum_{i=1}^{n+2}\sum_{j=0}^{n} \frac{T_{ij}}{TSTP} \log_2 \frac{T_{ij} \cdot TSTP}{T_{i.} \cdot T_{.j}}$	Ulanowicz (2004)

Where
n: Number of internal compartments in the network, excluding 0 (zero), *n + 1* and *n+2;*
j=0: external sources;
i = n + 1: Usable export from the network;
i = n + 2: Unusable export from the network (respiration, dissipation);
T_{ij}: Flow from compartment *j* to *i*, where j represents the columns of the flow matrix and *i* the rows;
T_i: Total inflows to compartment *i;*
T_j: Total outflows from compartment *j;*
TST: Total system throughflow.

Appendix 6.1. Data sources used to derive Ecopath parameter estimates and diet composition for the coastal ecosystem model in Vietnam. B is Biomass, P/B is production rate, Q/B is the consumption rate, and EE is the ecotrophic efficiency.

No.	Species	Biomass ($t \cdot km^{-2}$)	P/B ($year^{-1}$)	Q/B ($year^{-1}$)	EE	Diet matrix
1	Mammal	Estimated by Ecopath	Chen et al. (2008a)	Griffiths et al. (2010)	Duana et al. (Duana et al. 2009)	Chen et al. (2008a)
2	Sea turtle	Estimated by Ecopath	Griffiths et al. (2010)	Griffiths et al. (2010)	Griffiths et al. (2010)	Griffiths et al. (2010)
3	Tuna	RIMF (2005b)	Griffiths et al. (2010)	Calculated from empirical formula of Palomares and Pauly (1989)	Estimated by Ecopath	Griffiths et al. (2010)
4	Large predators	RIMF (2005b)	Chen et al. (2008a)	Chen et al. (2008a)	Estimated by Ecopath	Chen et al. (2008a)
5	Large demersal fish	RIMF (2005a)	Chen et al. (2008a)	Chen et al. (2008a)	Estimated by Ecopath	Chen et al. (2008a)
6	Other demersals fish	RIMF (2005a)	Chen et al. (2008a)	Chen et al. (2008a)	Estimated by Ecopath	Chen et al. (2008a)
7	Reef fish	RIMF (2005a)	M - calculated by empirical equation of Pauly (1980); F - calculated by catch/biomass (C/B)	Calculated from empirical formula of Palomares and Pauly (1989)	Estimated by Ecopath	Wang et al. (2012)
8	Large pelagic fish	RIMF (2005c)	Griffiths et al. (2010)	Wang et al. (2012)	Estimated by Ecopath	Froese and Pauly (2006)
9	Medium pelagic fish	RIMF (2005c)	M - calculated by empirical equation of Pauly (1980); F - calculated by C/B	Chen et al. (2008a)	Estimated by Ecopath	Froese and Pauly (2006)
10	Small pelagic fish	RIMF (2005c)	Chen et al. (2008a)	Wang et al. (2012)	Estimated by Ecopath	Froese and Pauly (2006)

11	Anchovy	RIMF (2005c)	M - calculated by empirical equation of Pauly (1980); F - calculated by C/B	Calculated from empirical formula of Palomares and Pauly (1989)	Estimated by Ecopath	Froese and Pauly (2006)
12	Cephalopods	RIMF (2005a)	Chen et al. (2008a)	Chen et al. (2008a)	Estimated by Ecopath	Chen et al. (2008a)
13	Shrimp	RIMF (2005a)	Wang et al. (2012)	Wang et al. (2012)	Estimated by Ecopath	Wang et al. (2012)
14	Crustaceans	RIMF (2005a)	Chen et al. (2008a)	Chen et al. (2008a)	Estimated by Ecopath	Chen et al. (2008a)
15	Zoobenthos	Estimated by Ecopath	Chen et al. (2008a)	Chen et al. (2008a)	Wang et al. (2012)	Griffiths et al. (2010)
16	Zooplankton	Estimated by Ecopath	Chen et al. (2008a)	Chen et al. (2008a)	Wang et al. (2012)	Griffiths et al. (2010)
17	Phytoplankton	Estimated by Ecopath	Griffiths et al. (2010)	Not applicable	Griffiths et al. (2010)	Not applicable
18	Detritus	Chen et al (2008a)	Not applicable	Not applicable	Estimated by Ecopath	Not applicable

Appendix 6.2. Diet matrix of ecological groups used for the coastal ecosystem model in Vietnam.

No	Prey \ predator	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
1	Mammal																
2	Sea turtle																
3	Tuna	0.122		0.090	0.111												
4	Large predators	0.300			0.006												
5	Large demersal fish	0.200	0.100		0.100												
6	Other demersals fish					0.070											
7	Reef fish					0.080											
8	Large pelagic fish			0.100	0.149												
9	Medium pelagic fish	0.080		0.130	0.190				0.020								
10	Small pelagic fish	0.050		0.113	0.271				0.310	0.100							
11	Anchovy			0.143					0.100								
12	Cephalopods	0.137	0.161	0.180	0.168				0.160								
13	Shrimp	0.100	0.180			0.063		0.265									
14	Crustaceans		0.145		0.001	0.311	0.040	0.033		0.059	0.120	0.260	0.333				
15	Zoobenthos		0.214			0.120	0.156	0.546		0.000	0.430	0.540	0.467				
16	Zooplankton	0.011	0.200	0.244	0.004	0.335	0.414	0.156	0.200	0.601	0.240	0.200	0.200	0.550	0.400	0.125	0.050
17	Phytoplankton					0.021	0.390		0.210	0.240	0.210			0.050	0.200	0.500	0.900
18	Detritus													0.400	0.400	0.375	0.050