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Production and food web efficiency decrease as fishing activity increases in a coastal ecosystem

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ABSTRACT

Fishing effort in the Vietnamese coastal ecosystem has rapidly increased from the 1990s to the 2000s, with unknown consequences for local ecosystem structure and functioning. Using ecosystem models that integrate fisheries and food webs we found profound 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 did not vary, 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 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 profound changes between the 1990s and 2000s, and emphasise the need for a closer inspection of the ecological impact of fishing.

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., 2014). 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., 2014). The average catch per horse power (HP) estimated in the 1990s was around 0.6 tons$\cdot$HP^{-1} and this number was reduced to about 0.35 tons$\cdot$HP^{-1} in recent years (Fig. 1; Anh et al., 2014). 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., 2014). At the end of the 1990s, the number of fishing vessel was only around 70,000 but

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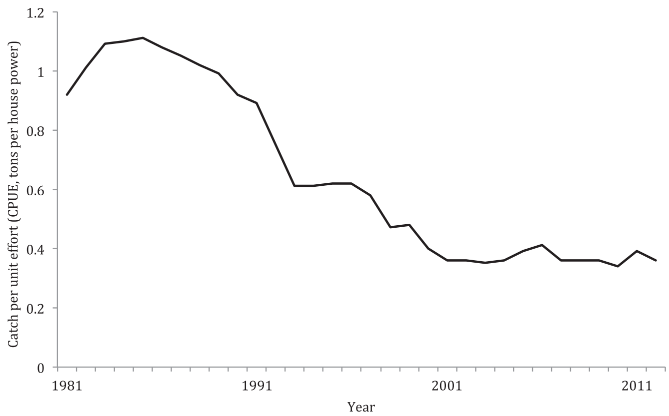

Fig. 1. Catch per unit effort (CPUE) of Vietnamese fisheries from 1981 to 2012. Adopted from Anh et al. (2014).

it reached about 85,000 in 2005 (Anh et al., 2014). 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., 2014). 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. 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; Fig. 1). 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 usually considered as relatively unreliable data sources in the present study. This is due to the limited amount of time series data and an unstandardized way of data collection. Therefore, we used a sensitivity analysis to assess the solution 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.

2. Methods

2.1. The data

In the present study, we focus on the coastal ecosystem of Vietnam with a total area of 464,748 km^2, and an average depth of around 40 m. The coastline of the present coastal ecosystem is composed of many estuaries, bays, lagoons and mangrove forests, which are considered biodiversity hotspots (Thanh et al., 2004). We considered two periods for which fish stock data and landings were available, only differing in the fishing pressure experienced: the end of 1990s and the early 2000s. 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 (Fig. 2 and Supplement No. 1).

The biomasses of pelagic fish 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). A total of 43 acoustic transects (33 transects of 60 nm and 10 transects of 30 nm) were conducted within this survey. 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 pelagic fish 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 (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), and biomass data of mammals, sea turtle, zoobenthos, zooplankton and phytoplankton were obtained from the literature (Chen et al., 2008a, 2008b; Van et al., 2010). Details on the assembled data are available in the Supplementary materials (Supplement No. 2). 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. 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.

We estimated total annual catch from landing data collected previously using a standard method of the Food and Agriculture Organization as described in Anh et al. (2014). Of those, 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) were collected monthly using questionnaires (FAO, 2002). The number of fishing vessels (FS) was based on reported data from local authorities. The monthly total catch (C) by the fisheries and by species was then estimated by:

$$C = FS \cdot CPUE \cdot BAC \cdot A \tag{1}$$

Annual total mean catch by the fisheries was then calculated per ecological group by summing monthly averages. Total catches were expressed as $t \cdot C \cdot km^{-2} \cdot year^{-1}$ using the conversion factors described above (Fig. 3 and Supplement No. 1).

2.2. The models

This work adopted the inverse methodology, adjusted from physical sciences to reconstruct coastal food webs (Vezina and Platt, 1988), and recently described by van Oevelen et al. (2010). The general structure of an inverse modelling includes: (a) functional group mass-balance equations (the food web topology), (b) data equations, and (c) constraints (Savenkoff et al., 2007; De Laender et al., 2010). The mass-balance equations specify that, for each functional group, 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