Ecological Modelling 278 (2014) 52–66

Contents lists available at ScienceDirect

Ecological Modelling

journal homepage: www.elsevier.com/locate/ecolmodel

# Assessing changes in the southern Humboldt in the 20th century using food web models

Sergio Neira[a,b,c,*], Coleen Moloney[c], Lynne J. Shannon[c], Villy Christensen[d], Hugo Arancibia[a], Astrid Jarre[c]

[a] Departamento de Oceanografía, Universidad de Concepción, Chile
[b] Programa COPAS S-A and Centro INCAR, Universidad de Concepción, Chile
[c] Marine Research Institute and Department of Biological Sciences, University of Cape Town, South Africa
[d] Fisheries Centre, University of British Columbia, Canada

ARTICLE INFO

*Article history:*
Received 14 September 2012
Received in revised form 3 January 2014
Accepted 7 January 2014
Available online 26 February 2014

*Keywords:*
Southern Humboldt
Food web reconstruction
Ecopath with Ecosim
Ecological network analysis
Trophic interactions
Upwelling system

ABSTRACT

This paper assesses changes in the southern Humboldt system (33–39 °S) in the 20th century by constructing and comparing food web models representing four historical periods: (i) lightly exploited (<1900s), (ii) altered by removal of marine mammals (1950), (iii) development of industrial fisheries (1992), and (iv) the more recent state (2005) when main stocks are fully or overexploited. Models are constructed with a standard structure in terms of functional groups using the Ecopath with Ecosim (EwE) software version 5.1. Model components include: the fisheries, cetaceans, sea lions, marine birds, cephalopods, large-sized pelagic fish (swordfish), medium-sized pelagic fish (e.g., horse mackerel, mackerel, and Pacific grenadier), small-sized pelagic fish (e.g., anchovy and Araucanian herring), demersal fish (e.g., Chilean hake, black conger-eel), benthic invertebrates (red squat lobster, yellow squat lobster, pink shrimp) and other groups such as zooplankton, phytoplankton and detritus. Input data are gathered from published and unpublished (grey) literature. Inter-model comparison is based on individual and community indicators obtained by means of network analysis. The biomasses of target species (with the exception of small pelagic fish) and top predators decrease in more recent models. Although predation mortality is the main cause of total mortality for the majority of the fish groups in all models, fishing mortality of target species is high in the 1992 and 2005 models. This has led to a decrease in the importance of predation removal of the most important fish stocks compared with fishing, which is found to assume the role of the main predator in the system in the more recent state. Changes in system energetics could have resulted in loss of productivity by increased flow to detritus. Prior to heavy fishing, long-living and high-trophic level species were abundant compared to the most recent models in which species with small body size, short life span and low trophic level dominate. These fishing-induced trends are in accordance with what is theoretically expected in stressed ecosystems and, if correct, the food web could now be more susceptible to external forcing and negative ecological interactions.



## 1. Introduction

Interactions between humans and the marine environment have a long history. Gathering and fishing were probably one of the first anthropogenic activities impacting coastal ecosystems (Nicholson, 1979). With time, other human activities linked to the sea (e.g., transport, trading, and exploitation of new sources of energy, among others) were significant for the development of human societies, but also important stressors for marine ecosystems.

Despite these ancient bonds, anthropogenic impacts on many marine ecosystems have become significant only since the industrial era (Pauly et al., 1998; Caddy and Cochrane, 2001; Valdemarsen, 2001; Christensen et al., 2003; Myers and Worm, 2003). More powerful ships equipped with improved technology (refrigeration, navigation, and fish detection systems) and fishing gears transformed humans into successful predators able to catch almost anything that is abundant and/or valuable in the sea (Smith, 1994; Pauly et al., 2002). As a result, the world fish catch increased from 2 million tonnes in 1850 to a maximum of 95.5 million tonnes in 2001 (McGoodwin, 1995; FAO, 2005). This fast growth

* Corresponding author at: Cabina 10, Departamento de Oceanografía, Universidad de Concepción, P.O. Box 160-C, Concepción, Chile; Zoology Department, University of Cape Town, Rondebosch, 7701, South Africa. Tel.: +56 41 2661329; fax: +56 41 2225400.
*E-mail address:* seneira@udec.cl (S. Neira).

0304-3800/$ – see front matter 
http://dx.doi.org/10.1016/j.ecolmodel.2014.01.003

was possible by first exploiting coastal resources and then by expanding the geographic and bathymetric range of fleets to new resources (Watson and Pauly, 2001). In this process, almost every important stock has been driven to collapse (Beverton, 1998), many of them without clear signs of recovery (Hutchings, 2000).

Fishing removal of target and non-target species (the category of by-catch accounts for a quarter of the world catch) has directly and indirectly altered the food webs where fisheries are embedded. Fishery-induced impacts extend far beyond target and by-catch species, decreasing the length (Bianchi et al., 2000; Daan et al., 2005) and trophic level (Pauly et al., 1998) of fish assemblages, along with habitat destruction (Jennings and Kaiser, 1998; Trimmer et al., 2005). Evidence of trophic cascades in marine ecosystems (Daskalov, 2002; Frank et al., 2005; Daskalov et al., 2007) could indicate that humans are starting to impact the ocean as in big lakes, where cascades are common. Fishing has degraded and simplified marine ecosystems towards food webs dominated by species with high turnover rate, small body size, short life span and low trophic level.

As fishing is increasingly considered the main threat to marine ecosystems (Jackson et al., 2001; Halpern et al., 2008), traditional fisheries management is experiencing a worldwide crisis due among other to the limited ability of single-species models to assess the effects of fishing on ecosystems and the effects of the environment (physical and biological) on fishery resources (McGoodwin, 1995; Buckworth, 1998; Botsford et al., 1997). Consequently, the ecosystem approach to fisheries (EAF) has been emerging as a new paradigm recognizing that if fisheries are to be sustainable, then wider ecosystem considerations must be explicitly considered in management (FAO, 2003).

The understanding of ecosystem changes, with fisheries included as an important component, is a key aspect of EAF and, along with conservation of biodiversity and sustainability, ecosystem reconstruction is often considered a major goal for EAF (Pitcher and Pauly, 1998; Pitcher, 2001, 2005). Although it is unlikely, even under sustainable fishing pressure, that ecosystems can be reconstructed to, for example, their pristine states (we do not know how ecosystems were in the past, changes in the ecosystem structure and function may be irreversible and some components could be already extinct), historical ecosystem-based reference points can potentially increase our understanding of ecosystem dynamics and the roles of fishing and other forcing (i.e., environmental variability) in driving ecosystem change. This understanding is important when assessing the current status of ecosystems and in developing strategies aimed to reconstruct or rehabilitate ecosystems and/or to alleviate negative impacts caused by external and internal forcing (Pitcher, 2001, 2005).

However, little attention has been paid to the state of harvested populations (communities) beyond comparatively recent times. This is because fisheries scientists have been rather contemporary in their scope, disregarding historical trends and precedents, and tending to perceive the productive potential of a stock as the amount of fish present at the beginning of their careers. This baseline change (so-called "shifting baseline", sensu Pauly, 1995) implies that the perception of the productive capacity of stocks (and ecosystems) decreases with successive generations.

Although reconstructions of time series of exploited species can be indicative of past ecosystem changes, retrospective analysis based only on biomass, catch and effort data can be misleading since, generally, these data are more related to the behaviour of the human predator rather than the natural prey (Heath, 2005). In addition, fishery-induced changes in ecosystem components can be delayed for several years (Daan et al., 2005). Therefore, in addition to abundance and fisheries-based indices, there is also a need for flux-based and food web metrics in order to gain insights into mechanisms by which climate and fishing might affect marine food webs (Shannon et al., 2009a).

Food web models and network analysis are useful for following and assessing the pathways through which energy flows up and down the food web, and for comparing the relative magnitudes of these flows through, for example, the benthos, plankton, predators and fisheries, and how fishing alters them (e.g., contributions in Christensen and Pauly, 1993; Pauly and Christensen, 1995; Bundy, 2005; Coll et al., 2009; Heymans et al., 2004; Shannon et al., 2003; Watermeyer et al., 2008a,b). Network and food web based metrics are key measures of ecosystem structure and function (Garrison and Link, 2000; Cury et al., 2005a), conveying useful information in the process of translating our understanding of ecosystem dynamics, patterns and processes into fisheries management.

The southern section of the Humboldt Current System off central Chile (33–39 °S) is a typical eastern boundary ecosystem, supporting one of the highest levels of primary productivity recorded for the open ocean (1.7 gC $m^{-2}$ $y^{-1}$; Daneri et al., 2000), and globally significant landings (>4.5 million tonnes in 1995). Despite this high biological productivity, the state of exploited stocks is far from being healthy (Cubillos et al., 1998; Arancibia and Neira, 2005a,b). In fact, the main target species have been fully exploited or overexploited for years, leading to a series of recent stock depletions, e.g., for horse mackerel (1998), red squat lobster and yellow squat lobster (1999), and Chilean hake (2004).

When analyzing the whole food web, Arancibia and Neira (2005a,b) observed a significant decrease in the mean trophic level of the catch, which indicates a likely change in community structure induced by fishing. In addition, small pelagic fish species hypothesized to play a key role in all major upwelling systems as "wasp-waist" populations (Rice, 1995; Cury et al., 2000, 2005b; Shannon et al., 2009b), have been under increasing fishing pressure, and a recent and unexplained outbreak of jumbo squid (*Dosidicus gigas*) in the system has been associated with steep declines in some demersal and pelagic fish species (SSP, 2004; Arancibia and Neira, 2008). Sources and consequences of these changes are poorly known, but it is unlikely that they can be fully understood from a single-species perspective. Rather, a more holistic approach is needed. Unfortunately, little is known about the ecosystem status of the southern Humboldt system before and at the beginning of heavy industrial fishing.

Future ecosystem-based strategies applied in the southern Humboldt will rely on historical reference points to compare the performance of management strategies. In this study we assess changes at the functional group and food web levels in the southern Humboldt by constructing and comparing Ecopath models representing four historical periods, of which the first two in particular, by virtue of limited available data, should be viewed only as possible representations of the ecosystem in that era: (i) the lightly exploited state (<1900s); (ii) the altered state induced by removal of big marine mammals (1950); (iii) the heavy industrial fishing period (1992); and the more recent state (2005) when individual stocks and the whole food web are considered significantly altered by fishing (Arancibia and Neira, 2005a,b).

## 2. Materials and methods

### 2.1. Study area

The southern Humboldt system off central Chile extends from 33 °S to 39 °S and from the coastline up to 30 nautical miles offshore, covering approximately 50,000 $km^2$ (Fig. 1). This geographical unit corresponds to the "Mediterranean District" of the Humboldt system, that is ecologically independent from the "Peruvian Province" and the "Southern District" located northward and southward,