



Contribution of ecosystem analysis to investigating the effects of changes in fishing strategies in the South Brazil Bight coastal ecosystem[☆]

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Abstract

Ecosystem effects of recent changes in fishing strategies in the South Brazil Bight (SBB) area, including increasing squid catches by shrimp bottom trawlers and fishing for young sardines as bait, for the skipjack tuna pole-and-line fishery were investigated by modelling the SBB coastal ecosystem for the 1998–1999 fisheries period, using the mass-balance modelling software, Ecopath with Ecosim. Based on fisheries information and on previous models, 25 species groups were defined. The mean trophic level of all fisheries was 2.99 with small trawlers and the live baitfish fleet occupying the lowest trophic levels.

The change in squid fishing rate and the most important squid predators fishing rate were simulated by increasing fishing mortality (F) from 0 to 1 per year, while maintaining F constant for other exploited groups. Also, the impact of fishing pressure by trawlers on squids and by the live baitfish fleet on young sardines were simulated. Different hypotheses of flow control were tested by setting the maximum instantaneous mortality rate that consumers could exert on food resources by (a) “top-down” control by predators (predator control); (b) all interactions of mixed control type; and (c) “bottom-up” control of predators by their prey (donor control). For squid, predicted biomass changes were more pronounced under top-down control than under bottom-up. Biomass of the weakfish, which are important squid predators, were the most affected under all hypotheses. A similar pattern of decrease in other squid predators was seen. The ecosystem consequence of overexploiting weakfish and large pelagic fish biomass was speculated regarding the increasing squid catches.

Simulations of increasing live-baitfish fleet did not show prominent impact in the inner shelf, where present F values are low. Given young sardines concentrate in shallow waters, where the conflicts converge, a complementary model of the adjacent coastal system and artisanal fisheries could better address fleet’s impact. Nevertheless, it was noted a decrease of sharks and rays reflecting their vulnerability in the system.

A precautionary measure for fisheries management in the region would be to assume that top-down control is a dominant force in energy flows, differing from the custom emphasis placed only in environmental forces and productivity. The ecosystem approach will require the integration of information from a wide range of disciplines, levels of ecological organization and temporal and spatial scales, as well as concrete management measures.

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Keywords: Ecosystem impact of fishing; Brazil; Ecosystem analysis; Ecopath; Fisheries management; Ecosystem approach

[☆] Manuscript PFITEC-11 (EMECs 7) for Ecological Modelling, May 2003.

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1. Introduction

Ecosystem analysis has proved to be a useful tool to investigate the direct and indirect effects of fishing (Christensen and Walters, 2002, this vol.; Shannon et al., 2000; Vasconcellos and Gasalla, 2001). Nevertheless, ecosystems are extremely complex, as well as the relationships between fisheries and biological productivity changes, which makes them impossible to treat comprehensively (Jorgensen, 1992). Ecosystem analysis gives a description and assessment of the whole system in terms of interaction between species, productivity and major functional processes (Field et al., 1989; Heymans and Baird, 2000). The first step is to construct a comprehensive food web of the system based on quantitative data on biomass and flows between compartments. Then, a dynamic approach to modelling energy flow can be used to address ecosystem responses to changes in fishing strategies (Walters et al., 1997).

Previous modelling has already been carried out for specific areas and time periods for South Brazil, such as both the Ubatuba region (Rocha et al., 1998; Rocha, 1998) and the Southeastern shelf (Vasconcellos, 2000) in the 1980s, and the extreme South area (Vasconcellos and Gasalla, 2001). However, this modelling has suggested the need for a new model that focuses on the most intense fishery area of the South Brazil Bight (SBB)—the inner shelf system as a whole—and that (a) uses more complete, local, up-to-date data; (b) emphasizes the exploited part of the food-web; (c) refines the interactions between industrial fisheries, fishery stocks, bycatches and the biological components of the system, and (d) elaborates further the effect of different exploitation patterns in the productivity, integrity and catches of the SBB.

Several changes in fishing strategies have been recently noted as a result of the overexploitation of target species and oceanographic anomalies. Purse-seiners fishing for demersal fish, shifts in catch composition, and other evidences stressed the present multispecies nature of major regional coastal fisheries (Gasalla et al., 2003; Tomás et al., 2003).

The increasing squids catches by bottom trawlers is just an example. It was reported both an increase in the volume of squids caught as bycatch in shrimp fisheries off Santos (Tomás et al., 2000) and a minor pair-trawlers fleet recently targeting squids in the sum-

mer season off Santa Catarina (Pérez, 2002). It raises the question of whether this is due to oceanographic anomalies or whether it can be caused by the overexploitation of higher trophic level species.

Also, skipjack tuna pole-and-line boats enter the coast for young sardines as bait, conflicting with other users of that environment, i.e. artisanal fishers and sardine purse-seiners. The effects of these new practices on the whole system and on the fisheries are important issues that need to be addressed.

In this paper, we present a new mass-balance model of the SBB coastal ecosystem for the period 1998–1999, and the use of Ecopath with Ecosim (Christensen and Pauly, 1992) to investigate the ecosystem effect of altered fishing rates on squid mortality by trawlers, and on sardine mortality by the live-baitfish fleet.

2. Study site

The crescent-shape SBB lies between two prominent capes, Cabo Frio (23°S) and Cabo de Santa Marta (28°S) on the Southeastern coast of Brazil (Fig. 1), with a coastal length of about 1100 km cut by cliffs, small bays and many islands, and with isobaths approximately parallel to the coastline (Castro and Miranda, 1998). The northern and southern boundaries respectively limit the biogeographic distribution of many tropical coastal *taxa*, and the mangrove systems along the Brazilian coastline (Lamardo et al., 2000).

Circulation, waters interaction and meteorological regime determine major enrichment processes and productive interfaces in the South Brazil Bight, such as seasonal, locally wind-driven upwelling regimes (Valentin et al., 1987) or sparse cool intrusions that create subsurface chlorophyll maximum layers (Brandini, 1990a; Matsuura, 1989; Gaeta and Brandini, 2002). Fertilization, phytoplankton biomass accumulation and enhanced new production are dependent on the duration of physical events, such as cyclonic vortexes, summer upwellings (Pires-Vanin, 1993; Castro and Miranda, 1998), winter wind-induced sub Antarctic lateral intrusions (Campos et al., 1996), and inner waves (Gaeta and Brandini, 2002).

In the inner shelf, major oceanographic interactions occur among warm, saline, oligotrophic surface water (so-called Tropical Water); a low salinity, more



Fig. 1. Map of the study site (South Brazil Bight—20–100 m depth).

productive, Coastal Water, highly influenced by rivers runoff; and the seasonal penetration towards the coast of cold, nutrient-rich slope waters identified as the South Atlantic Central Water (SACW) (Castro and Miranda, 1998). This intrusion has been related to the occurrence of meanders and eddies, which are common features of the Brazil Current, and to winds from E-NE, which dominate in the summer (Castro Filho et al., 1987). Vertical mixing occurs controlled by cold fronts, which put nutrients back in suspension to be utilized by phytoplankton, and creating favourable reproductive habitat for coastal fish (Matsuura, 1996; Katsuragawa et al., 1993). Bakun and Parrish (1990) infer a closed gyral geostrophic circulation pattern that may aid in retaining sardine larvae within the favourable neritic habitat. A coast-ocean gradient of zooplankton distribution has been noted with its higher biomasses associated with the Coastal Water (Bonecker et al., 1990; Lopes et al., 2001).

The SBB semi-enclosed ecosystem is very diverse, with 619 benthic invertebrate species and more than 590 fish species described so far (Castello et al., 1994). It has been affected by large-scale fishing activities, especially since the 1970s. The study area is defined here

as the SBB inner shelf between the 20 and 100 m isobaths (23–28°S and 40–48°W), the principal grounds of a multispecies and multigear industrial fishery. The area, $9.7 \times 10^{10} \text{ m}^2$ (estimated using Arcview 3.1) forms an appropriate unit for ecosystem management purposes (Matsuura, 1995; Gasalla and Tomás, 1998).

3. Methodology

3.1. The mass-balance model

We constructed an original mass-balance model of the SBB inner ecosystem for 1998 and 1999, using Ecopath with Ecosim version 4.0. The Ecopath model is a system of linear equations that for any given group (*i*) can be represented for any time interval by

$$\Delta B = B_i \left(\frac{P}{B} \right)_i EE_i - Y_i + \sum_{j=1}^n B_j \left(\frac{Q}{B} \right)_j DC_{ji} \quad (1)$$

where ΔB is the biomass accumulation, is equal to 0 in the case the system is under equilibrium, B_i is the biomass of (*i*) during the period in question, $(P/B)_i$

is the production/biomass ratio, EE_i is the ecotrophic efficiency, i.e. the fraction of the production of (i) that is consumed within the system or harvested, Y_i is the yield [with $Y_i = F_i B_i$, and F as the fishing mortality], B_j is the biomass of consumers or predators, $(Q/B)_i$ is the food consumption per unit of biomass of (j), and DC_{ji} is the fraction of (i) in the diet of (j).

The model was constructed to reflect annual conditions in the SBB firstly by assessing the most important fisheries resources in 1998–1999 landings calculated for the whole area. Data were obtained from São Paulo, Rio de Janeiro, Paraná and Santa Catarina State's official catch statistics reported by the Instituto de Pesca and IBAMA. Based mainly on fisheries information and on previous models, we defined 25 species groups as described in Table 1.

Some species groups reported in other models of Southeastern Brazil (Rocha, 1998; Vasconcellos, 2000) were not included here due to: (1) the specific study area is different and we focused on the whole inner shelf, (2) species could now be rare in present study area and did not occur in the 1998–1999 catches, (e.g. jack mackerel), or (3) species could be dominant only in the outer shelf or shelf break (e.g. anchovy, Myctophidae). By the other hand, we add several important biological groups for the area (seabirds, mammals, octopus, flatfish, catfish, large pelagics) and put emphasis in catches dominant groups during the study period.

Annual primary productivity was obtained from Brandini (1990b), Giancesella-Galvão and Saldanha-Correia (1995), Gaeta (1999), and Gaeta and Brandini (2002). An average phytoplankton production of about 0.5 g C m^{-2} per day was suggested to the inner shelf (Gaeta and Brandini, 2002), ranging from 0.99 g C m^{-2} per day (generally in summer) to $0.11\text{--}0.17 \text{ g C m}^{-2}$ per day. The conversion factor of $0.06 \text{ g C} = 1 \text{ g wet weight}$ (Walsh, 1981) was adopted, considering an average carbon/biomass ratio of 30% dry weight, according to phytoplankton species composition (Parsons and Takahashi, 1973; Brandini et al., 1997).

Phytoplankton biomass was adjusted from chlorophyll *a* abundance values estimated in Gaeta (1999) for the main waters within the inner shelf area, by using a 1:25 ratio to convert to unit carbon. Detritus biomass was estimated using Pauly's empirical relationship (Pauly et al., 1993a).

Zooplankton biomass was calculated following wet weight obtained by Muxagata (1999) for winter conditions, and volumetric values from Matsuura et al. (1980) (Matsuura Prof. Yasunobu, Instituto Oceanográfico, University of São Paulo, personal communication), for autumn, spring and summer conditions, and Resgalla et al. (2001) estimates. Copepods dominate zooplankton samples in this area (Vega-Perez, 1993; Muxagata, 1999), and conversion of biovolume data to wet weight followed Pearre (1980).

Sardine biomass and P/B for sardines were obtained from VPA analysis by Cergole et al. (2001). The sardine group was split between young and adult pools, using the spawning stock biomass (SSB)/total biomass ratio of 0.75. Parameters used for the delay difference model in Ecosim were the von Bertalanffy growth parameter (k) = 0.5; and $w_k = 44$, $t_k = 1.5$, where w_k is the weight at age t_k of sardine graduates to the adult pool (Cergole, 1995; Vasconcellos, 2000). The age-structured population dynamics model embedded in the trophic model allows capturing ecosystem linkages such as changes in predation and consumption rates by juvenile and adult pools (Vasconcellos, 2000).

Benthic invertebrates were divided into "detritivorous benthos," "carnivorous benthos" and "shrimps." Estimates on the former groups were based on previous modelling (Rocha, 1998; Rocha et al., 1998; Vasconcellos, 2000) and on recent field data obtained from surveys by using box-corer samplers in the EEZ (Amaral, 1999).

The biomass of cetaceans was obtained from average population sizes (Alexandre Zerbini, School of Aquatic and Fisheries Science, University of Washington, personal communication) and average weight for each species, after Cruz (1982), Lichter and Hooper (1983) and Siciliano (Salvatore Siciliano, Museu Nacional, UFRJ, Federal University of Rio de Janeiro, personal communication). The P/B value was calculated as average mortality estimates for each species from $S = (w - 1)/w = \exp(-M)$, where S is survival, M is mortality, and w is longevity. Q/B was obtained from the daily consumption equation (Trites et al., 1997), based on Innes et al. (1987) daily ration estimate $R = 0.1 \times W^{0.8}$, where R is the daily ration and W is the mean body weight (kg).

For seabirds, species composition was based on Sick (1977) and Vooren and Brusque (1999). Biomass calculation followed three steps: (1) population size,

Table 1
Description of trophic groups components of the South Brazil Bight coastal ecosystem model

Trophic group	Dominant components	Secondary components
Phytoplankton	Phytoflagellates	Diatoms
Zooplankton	Copepods	Cladocera, Chaetognatha, Appendicularia, Sergestids, Luciferids, Salps
Shrimps	<i>Farfantepenaeus paulensis</i> <i>F. brasiliensis</i> <i>Xyphopenaeus kroyeri</i>	<i>Litopenaeus schmidt</i> <i>Pleoticus muelleri</i> <i>Artemesia longinaris</i>
Mulletts	<i>Mugil platanus</i>	<i>Mugil curema</i>
Catfish	<i>Netuma</i> spp.	<i>Bagre marinus</i> <i>Genidens genidens</i> <i>Bagre bagre</i>
Juvenile sardine	<i>Sardinella brasiliensis</i>	
Sardine	<i>Sardinella brasiliensis</i>	
Mackerel	<i>Scomber japonicus</i>	
Carnivorous benthos	Crabs (<i>Portunus</i> spp.), Echinoderms (<i>Astropecten</i> spp.), Polichaetes (Polynoidae, Nephthyidae, Glyceridae), and Stomatopods (<i>Squilla</i> spp., <i>Lysiosquilla</i> spp.)	Gastropods (<i>Olivella</i> spp)
Detritivorous benthos	Amphipods, Bivalves (<i>Pecten ziczac</i> , <i>Corbulla</i> sp) and Polichaetes (Capitellidae, Serpulidae, Sabellidae, Spionidae)	Gastropods (<i>Eulima</i> spp.), Echinoids (<i>Lytechinus</i> spp.), Cumacea (<i>Diastylis</i> spp.), and Amphipods
Squids	<i>Loligo plei</i>	<i>Loligo sanpaulensis</i>
Croaker	<i>Micropogonias furnieri</i>	
Other sciaenids	<i>Menticirrhus americanus</i>	<i>Cynoscion virescens</i>
<i>M. litoralis</i>	<i>Pogonias chromis</i> <i>Umbrina canosai</i> <i>Cynoscion leiarchus</i>	<i>Nebris microps</i> <i>Cynoscion acoupa</i> <i>Paralochurus brasiliensis</i>
Cutlass fish	<i>Trichiurus lepturus</i>	
King weakfish	<i>Macrodon ancylodon</i>	
Triggerfish	<i>Balistes capriscus</i>	
Octopus	<i>Octopus vulgaris</i>	
Large pelagic fish	<i>Auxis thazard</i> <i>Sarda</i> sp <i>Euthunnus alletteratus</i> <i>Katsuwonus pelamis</i>	<i>Coryphaena hippurus</i>
Other piscivores fish	<i>Peprilus paru</i> <i>Percophis brasiliensis</i> <i>Thyrstlops lepidopoides</i> <i>Oligoplites saliens</i>	<i>Selene setapinnis</i>
Weakfish Cynoscion	<i>Cynoscion jamaicensis</i> <i>Cynoscion guatucupa</i>	
Flatfish	<i>Paralichthys</i> spp. <i>Bothus</i> spp.	<i>Scyacium</i> spp. <i>Etropus</i> spp. <i>Citharichthys</i> spp. <i>Gymnachirus</i> spp.
Sharks and rays	<i>Carcharrhynchus brevipinna</i> <i>Carcharrhynchus limbatus</i> <i>Carcharrhynchus sygnathus</i> <i>Squatina gugenheim</i> ; <i>S. argentina</i> <i>Rhinobatos</i> spp. <i>Mustelus canis</i> <i>Squalus cubensis</i>	<i>Sphyrna</i> spp. <i>Squatina occulta</i> <i>Carcharrhynchus plumbeus</i> <i>Carcharrhynchus leucas</i> <i>Dasyatis</i> spp. <i>Rioraja agassizi</i> <i>Atlantoraja</i> spp.

Table 1 (Continued)

Trophic group	Dominant components	Secondary components
Cetaceans	<i>Sotalia fluviensis</i> <i>Pontoporia fluviatilis</i> <i>Stenella frontalis</i> <i>Tursiops truncatus</i>	<i>Steno bredanensis</i> <i>Balaenoptera edeni</i>
Seabirds	<i>Sula leucogaster</i> <i>Larus dominicanus</i> <i>Fregata magnificens</i>	<i>Sterna eurygnatha</i> <i>S. hirundinacea</i> (both seasonal)
Detritus		

Species names of invertebrates are only examples.

based on a census of reproductive colonies by Campos et al. (2000) and Moritz et al. (2000) plus 25% of juveniles and non-reproductive colonies (Dr. Fabio Olmos, Seabirds specialist, São Paulo, personal communication); (2) total population size estimate by multiplying (1) by the estimated number of other colonies in the area (Tatiana Neves, Instituto Florestal, São Paulo, personal communication); and (3) by multiplying the number of individuals by average weights, based on Nelson (1978, 1980) for each species. P/B is the default mortality coefficient for seabirds, and Q/B was calculated from annual consumption rates in Harrison and Seki (1987) and biomass values.

The diet matrix in the model (Table 4) was constructed based on information from Araújo (1984), Capitoli et al. (1995), Carneiro (1995), Chaves and Vendel (1996), Dall et al. (1990), Efe (1993), Gasalla (1995), Gasalla and Oliveira (1997), Gasalla et al. (2001b), Goitein (1983), Goitein et al. (1998), Haimovici et al. (1989), Harrison et al. (1984), Juras and Yamaguti (1985), Lima et al. (2000), Lucato (1997), Martins and Haimovici (1997), Nelson (1978, 1980), Nogueira (2000), Oliveira and Soares (1996), Paiva (1993), Rios (1994), Schmiegelow (1990), Soares et al. (1992, 1993, 1998, 1999), Santos and Haimovici (1998), San Roman (1972), Santos (1999), Soto et al. (2000), Vazzoler et al. (1999), Vanin (Prof. Ana Maria Pires Vanin, Instituto Oceanográfico, University of São Paulo, personal communication), Vazzoler (1975), Vilela (1990), Wakabara et al. (1982), Winik and Carneiro (1999), and from Fish-Base (Froese and Pauly, 1998).

Ecotrophic efficiencies were taken as 0.9 for all groups for which biomass data were not available, considering that 90% of the production is either consumed or exported from the system, as expected

for groups with abundant consumers or fully exploited.

Harvest information was grouped by trophic group and fishing fleet (Table 2), as being: *purse seiners*, targeting mostly sardines but catching several groups during the study period, including bottom-fish; *pink-shrimp bottom trawlers*, targeting pink shrimp species and catching several demersal species; *pair-bottom trawlers* for bottom fish, especially sciaenids; *small bottom trawlers*, targeting shrimps, mostly the sea-bob shrimp; *gillnetters*, seasonally targeting sharks and sciaenids; and the *live baitfish* fleet, taking young sardines for the bonito (skipjack tuna) fishery (Instituto de Pesca/IBAMA, 2000; Lin, 1992; Jablonski et al., 1998).

Finally, some adjustments had to be made in order to balance the model. High values of cannibalism were decreased to allow the model improve biomass estimates. Ecotrophic efficiency of large pelagic fish was reduced by decreasing predation mortality by cetaceans; also benthos biomass was increased in order to have estimates of transference among groups consistent with ecological and biological constraints.

3.2. Dynamic model simulations

Several reports on regional fisheries monitoring highlighted several recent changes in fishing dynamics along the coast (Gasalla et al., 2003; Tomás et al., 2003). Purse-seiners fishing for demersal fish, such as croakers; live baitfish fleet conflicting with seiners, and increasing cephalopods catches are just examples.

Local artisanal and industrial fishers are also concerned about changes in catch composition and fishing strategies, and advocate for the need of scientific investigation on its possible effects (Gasalla and Tutui,

Table 2
Catch per fleet for each trophic group.

Group name	Catch per fleet (t km ⁻² per year)						
	Purse-seiners	Gillnetters	Pair trawlers	Live-bait	Shrimp trawlers	Small trawlers	Total
Detritus	–	–	–	–	–	–	–
Phytoplankton	–	–	–	–	–	–	–
Shrimps	–	–	– ^a	–	0.60	0.18	0.78
Zooplankion	–	–	–	–	–	–	–
Mulletts	0.007	–	–	–	–	–	0.007
Detritivorous benthos	–	–	–	–	–	–	–
Juvenile sardine	0.135	–	–	0.01	–	–	0.145
Catfish	0.001	0.001	0.002	–	– ^a	–	0.004
Sardine	0.372	– ^a	–	0.004	–	–	0.376
Carnivorous benthos	–	–	–	–	– ^a	–	–
Mackerel	0.023	–	–	–	–	–	0.023
Squids	–	–	0.003	–	0.006	– ^a	0.01
Croaker	0.018	0.027	0.071	–	0.002	– ^a	0.118
Other sciaenids	–	0.002	0.02	–	0.002	– ^a	0.024
Cutlass fish	0.005	– ^a	0.002	–	–	– ^a	0.007
King wealfish	–	0.003	0.028	–	0.001	– ^a	0.032
Triggerfish	0.001	– ^a	0.019	–	0.003	–	0.023
Octopus	–	–	– ^a	–	0.003	–	0.003
Large pelagic fish	0.015	– ^a	–	–	– ^a	–	0.015
Other piscivore fish	0.003	0.001	0.003	–	0.002	–	0.009
Weakfish Cynoscion	–	0.002	0.065	–	0.001	– ^a	0.066
Sharks and rays	– ^a	0.019	0.003	–	0.001	– ^a	0.023
Flatfish	–	– ^a	0.001	–	0.003	– ^a	0.004
Cetaceans	–	–	–	–	–	–	–
Seabirds	–	–	–	–	–	–	–

^a Catch less than 0.001 t km⁻² per year.

1999, 2000). Here we chose to deal with two of the issues in a multispecies context: the increasing squid's catches and the conflicting live-baitfish for sardines in the inner shelf.

3.2.1. Squids

3.2.1.1. Impact of increasing squid fishing rates.

Squids are target species only for the nearshore artisanal jigging. Considering shelf fisheries, there have been reported an increase in the volume of squids caught as bycatch in shrimp fisheries and some targeting seasonal attempts by a small part of regional pair-bottom trawlers (Tomás et al., 2000; Pérez, 2002). Accordingly, we would like to explore the ecosystem effect of direct coastal squid exploitation.

The change in squid fishing rate was simulated by increasing fishing mortality (F) from 0 to 1 per year, while maintaining F constant for other exploited groups. To calculate the predicted changes in equi-

librium biomasses of species/groups, the ECOSIM routine was used. The model provides biomass predictions of each group in the system as affected directly by fishing and predation, change in food availability, and indirectly by fishing or predation on other groups in the system (Walters et al., 1997). Different hypotheses of flow control were tested by setting the maximum instantaneous mortality rate that consumer j could exert on food resource i (Walters et al., 1997): (a) “top-down” control by predators (predator control); (b) all interactions of mixed control type; and (c) “bottom-up” control of predators by their prey (donor control). For constructing each scenario, prey vulnerability settings were respectively: (a) $v = 1$, (b) $v = 0$, and (c) $v = 0.3$.

We also test a “wasp-waist” control (predator control) and bottom-up control of predators by mackerel, setting $v = 0.1$ for lower trophic levels, and $v = 0.4$ for higher trophic levels than mackerel, but in this case it showed similar patterns of a “bottom-up” control.

3.2.1.2. Impact of trawlers fishing rate in squid biomass. We constructed five fishing scenarios to predict relative change in squid biomass after a 10 years period. Simulations of different effort increase of shrimp trawlers and bottom-trawlers effort were tested also under “top-down” and “bottom-up” controls, following the settings above.

3.2.1.3. Decreasing squid predator’s biomass. In order to evaluate the impact of decreasing the biomass of squids predators, we test several scenarios by increasing predators fishing rates, under a mixed trophic control ($v = 0.3$).

3.2.2. Live-baitfish fleet impact

Here we considered different scenarios of flow control to explore the impact of fishing pressure by the live baitfish fleet on young sardines by simulating a situation of permanent four-fold increase of fishing rate from year 2 on in a 10-year period. Tested hypothesis here were: (a) “top-down” scenario, (b) “bottom-up”, and (c) “mixed control”, as described above.

Prey vulnerability settings for each scenario were respectively: (a) $v = 1$, (b) $v = 0$, and (c) $v = 0.3$.

4. Results

4.1. Mass-balance model

The multispecies, multigear complexity of the SBB coastal industrial fishery can be appreciated through [Tables 1 and 2](#), where trophic groups and the main biological components as well as their catches per fishing fleets are listed.

The basic structure of the trophic model including input and output parameters that represent the functioning of the balanced ecosystem’s main trophic processes from energy sources to seabirds and cetaceans, with particular emphasis on the fisheries, is shown in [Table 3](#) and [Fig. 2](#).

Within the system, phytoplanktonic primary production and detritus are the sources of energy that sustain on the one hand zooplanktonic secondary production and on the other, large biomasses of benthos that support a significant part of fishery landings. Predator groups benefits from linking benthic

and pelagic food chains at the top of the food web ([Fig. 2](#)). The role of commercial fisheries during the study period was equivalent to a predator occupying a mean trophic level of 2.99. [Fig. 3](#) shows the corresponding mean trophic levels of each fishing fleet and the relatively low values of reported annual catch for the period. Small trawlers and the live baitfish fleet occupy the lowest trophic levels within fisheries. The purse seiners that traditionally targeted sardines, target demersal species in their absence, such as croakers ([Table 2](#)).

4.2. Simulation of ecosystem responses

4.2.1. Increasing fishing mortality for squids

Equilibrium simulation outputs of the effects of increasing F for squids from 0 to 1 per year under different scenarios of flow controls on biomass of other components in the system are shown in [Fig. 4](#).

Predicted biomass changes are more pronounced under top-down control than under bottom-up. The biomass of the “weakfish” group was the most affected under both hypotheses, because they are important squid predators and compete with the fishery. A similar pattern of decrease in other squid predators can be seen. While some squid predators also eat squid prey, triangles of complex trophic interactions indicate the complexity of relationships involving squids. Under “top-down” and “bottom-up” control assumptions from the equilibrium simulation of increasing fishing mortality for squids, opposite tendencies in biomass changes can be observed, especially for higher trophic-level groups such as seabirds, cetaceans, large pelagics, sharks and rays, and flatfish ([Fig. 4](#)).

4.2.2. Effect of trawlers fishing rate in squid biomass

[Table 5](#) shows the predicted relative change in squid biomass after 10 years under the different control hypothesis. By increasing shrimp-trawlers effort two and four times, a corresponding squid biomass decrease can be noted, while reducing effort to the half, indicates its positive changes. A “top-down” control of doubling shrimp trawlers effort predicts relative higher change than a “bottom-up” control of a four-fold effort increase.

Table 3
Parameters of the trophic model of the South Brazil Bight inner ecosystem

Group name	Trophic level	Omnivory index	Biomass (t km ⁻²)	Prod./biom. (per year)	Cons./biom. (per year)	Ecotrophic efficiency
Detritus	1.0	0.16	10.000*	–	–	0.629
Phytoplankton	1.0	0.00	18.000*	166.00*	–	0.954
Shrimps	2.0	0.03	3.387	6.50 ^f	18.00 ^d	0.900
Zooplankton	2.1	0.05	10.500*	90.000 ^a	270.00 ^a	0.402
Mulletts	2.1	0.10	0.661	1.00 ^e	11.03 ^e	0.900
Detritivorous benthos	2.2	0.17	47.000*	3.00*	27.30*	0.974
Juvenile sardine	2.2	0.18	1.050*	7.00 ^d	23.33 ^d	0.228
Catfish	2.2	0.31	0.055	0.45 ^f	7.20 ^f	0.900
Sardine	2.8	0.18	3.976*	3.00*	11.20 ^d	0.161
Carnivorous benthos	3.0	0.33	34.000*	0.96*	3.28*	0.984
Mackerel	3.1	0.26	4.082	0.34 ^h	2.71 ^h	0.900
Squids	3.4	0.36	5.283	1.95 ⁱ	3.90 ^h	0.900
Croaker	3.5	0.26	1.813	1.16 ^j	3.38 ^k	0.900
Other sciaenids	3.5	0.30	5.762	1.04 ^{m,n}	5.30 ^{m,n}	0.900
Cutlass fish	3.6	0.25	4.638	0.41 ^o	2.05 ^{o,p}	0.850
King Weakfish	3.7	0.54	1.260	2.10 ^j	6.16 ^q	0.900
Triggerfish	3.7	0.18	0.699	0.90 ^d	6.13 ^d	0.900
Octopus	3.7	0.16	0.030 ^s	4.50 ^s	10.70 ^l	0.691
Large pelagic fish	3.9	0.69	0.710 ^t	1.50 ^{q,t}	6.20 ^{q,t}	0.811
Other piscivore fish	4.0	0.51	0.050	1.65 ^{q,u}	5.00 ^v	0.900
Weakfish Cynoscion	4.0	0.39	0.858	2.34 ⁿ	5.40 ^x	0.900
Sharks and rays	4.1	0.51	0.064	0.40 ^{z,m,d}	400 ^h	0.900
Flatfish	4.2	0.60	1.638	0.83 ^y	4.10 ^{m,n}	0.900
Cetaceans	4.4	0.62	0.160*	0.05*	35.50*	0.000
Seabirds	4.6	0.21	0.001*	0.10*	94.33*	0.000

Underlined values, trophic levels and omnivory index were estimated by the model. a and b: LeBorgne (1982); c: D’Incao (1991); d: Vasconcellos (2000); e: Vasconcellos (Marcelo Vasconcellos, University of Rio Grande, personal communication); f: Mendoza (1993); g: Cergole et al. (2000); h: Vasconcellos and Gasalla (2001); i: Pérez (2002), Pérez (Dr. Angel Pérez, University of Vale do Itajai, Santa Catarina, personal communication); j: Castro (2000); k: Vazzoler (1991); l: based on Buchan and Smale (1981); m: Rocha (1998); n: Godinho-Peria (1995); o: Martins and Haimovici (1997); Peterson and Wroblewski (1984); p: Palomares and Pauly (1989); q: Froese and Pauly (1998); r: Leite (2001); s: Tomás (2003); t: Chatwin (1997), Jablonski and Matsuura (1985); u: Pauly (1980); Pauly et al. (1993b); v: Aliño et al. (1993); x: Vieira (1990); y: Araujo and Haimovici (2000), z: Waring, 1984; *: see Section 3.

However, a four-fold increase in pair-bottom trawlers resulted in a considerable increase in squid biomass, associated with indirect relationships and the important effect of decreasing squid predators biomass.

4.2.3. Decreasing squid predators' biomass

Ecosim predicted percentual changes in biomass after increasing F for weakfish and large pelagic fish from 0 to 1 per year are shown in Fig. 5. As expected, after an increase of weakfish and large pelagic fish fishing rate, an increase in squids biomass and other lower trophic levels was predicted, as well as the decrease of predator groups, such as flatfish.

4.2.4. Increasing fishing rate for the live baitfish fleet

Ten-year simulation outputs of the effects of increasing the live baitfish fleet's fishing rate permanently from year 2 onward under three alternative types of flow control (top-down, mixed, and bottom-up control) are shown in Fig. 6. As present F values are low, the simulations do not show prominent impact on the shelf trophic groups.

The simulations all show a similar pattern of decrease of chondrichthyans biomass from year 2 to 4, when biomass recovery begins, being faster under bottom-up control. Under top-down control, perturbations propagate and show more pronounced responses, with also a longer recovery time for sharks and rays.

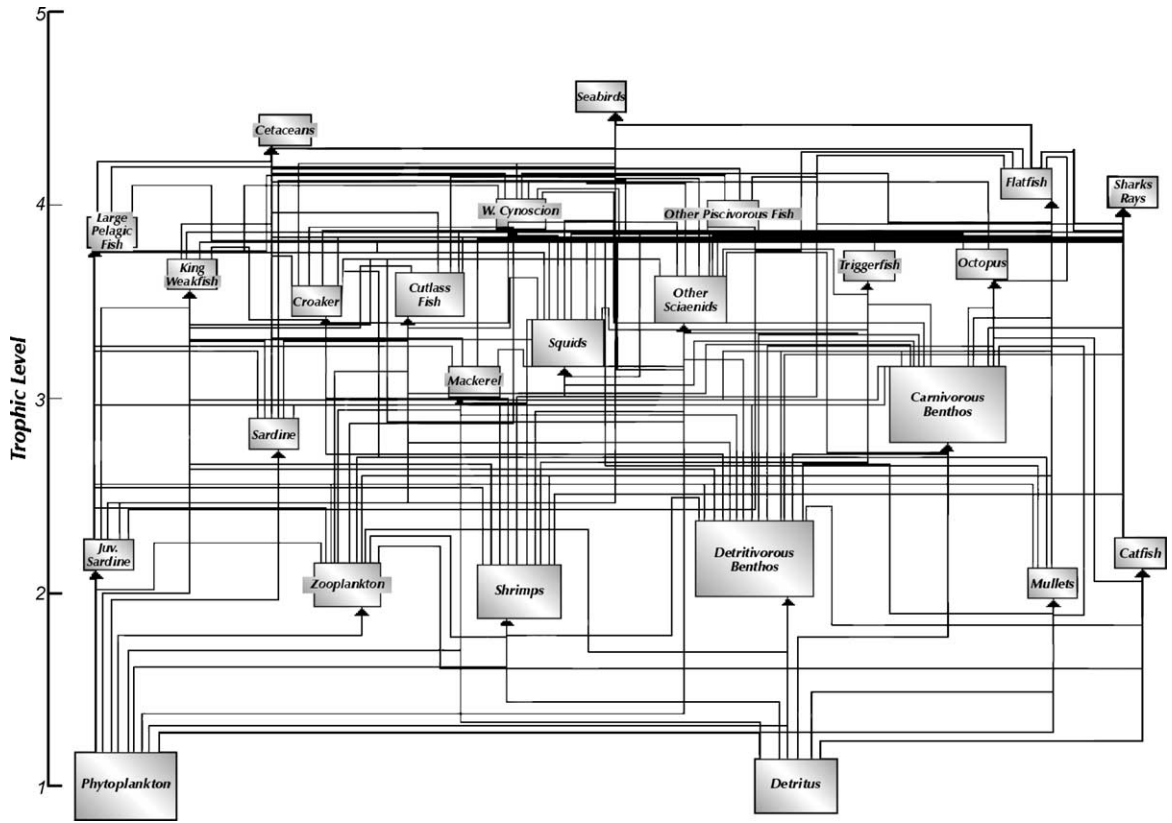


Fig. 2. Flowchart of trophic relationships in the South Brazil Bight coastal ecosystem, showing flows between groups, biomass of each group (box area proportional to log of biomass), and respective trophic levels.

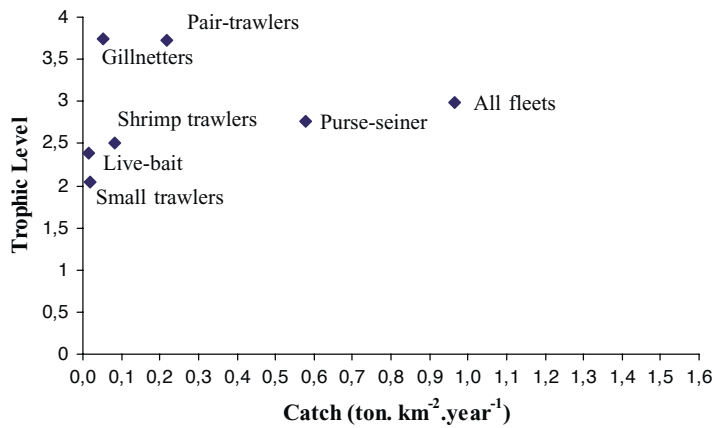


Fig. 3. SBB model 1998–1999 catch and trophic levels per fleet.

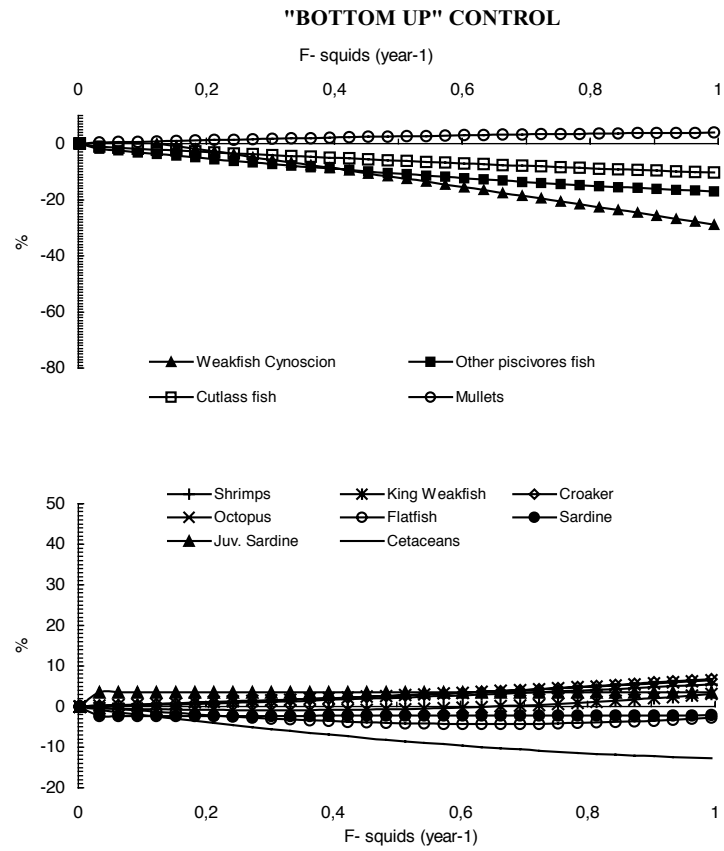
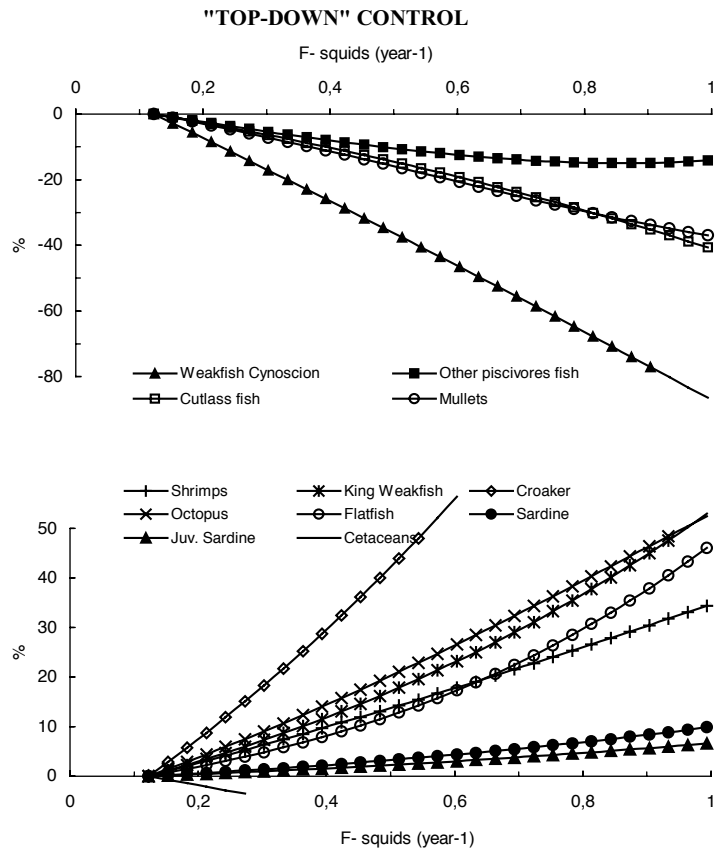


Fig. 4. Predicted relative change in biomass of other components in the system under "top-down" and "bottom-up" control assumptions from the equilibrium simulation of increasing fishing mortality for squids.

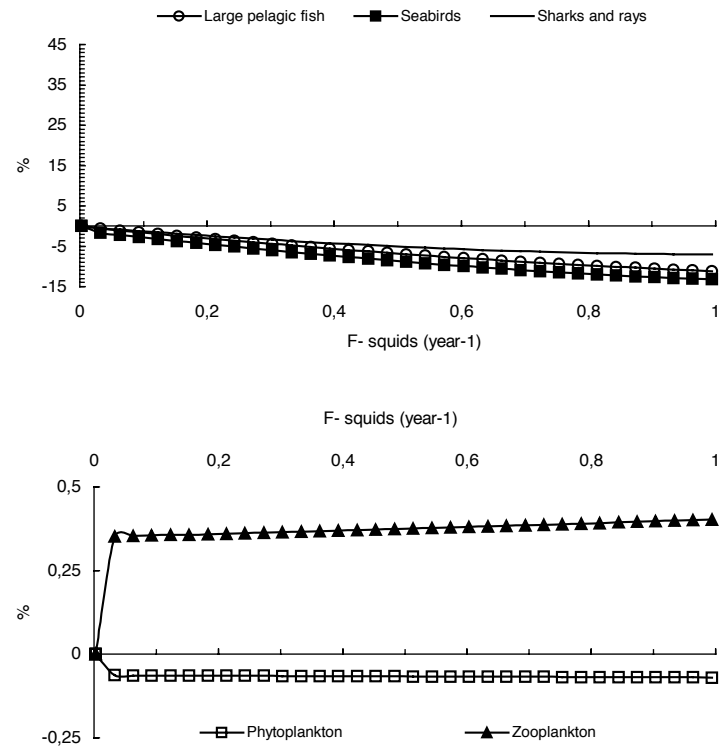
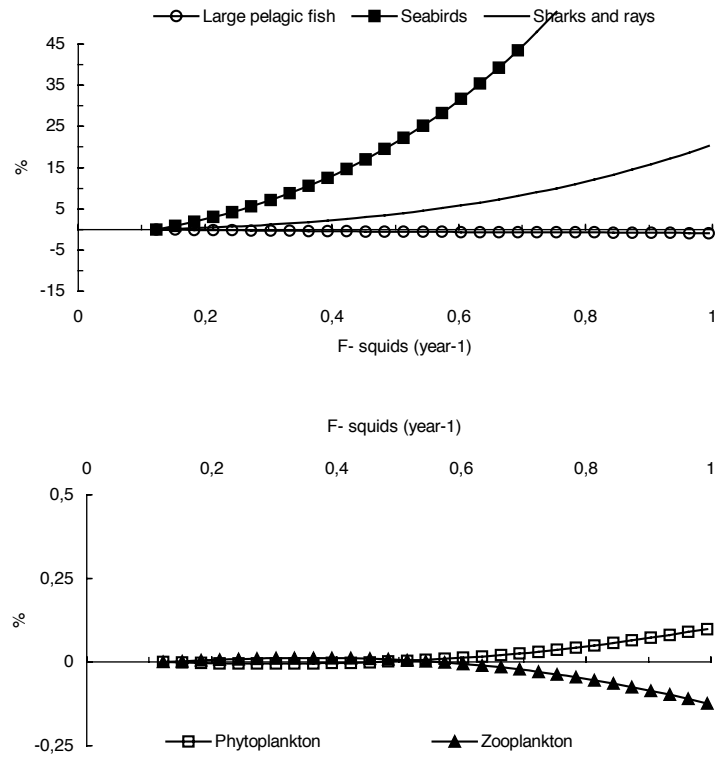


Fig. 4. (Continued).

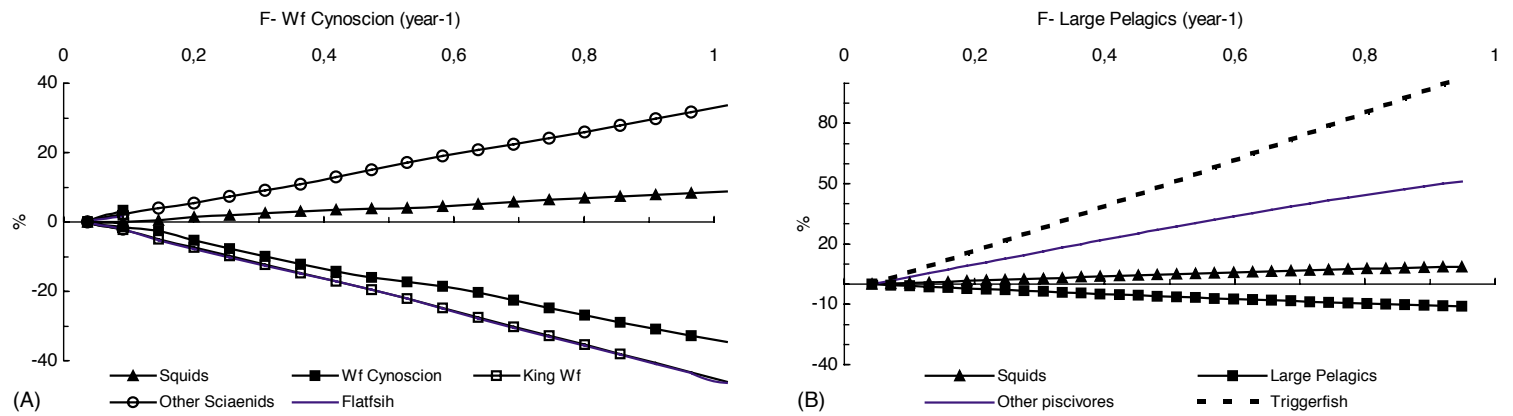


Fig. 5. Predicted change in biomass from the simulation of increasing fishing mortality for Weakfish Cynoscion (A) and large pelagic fish (B), under mixed control.

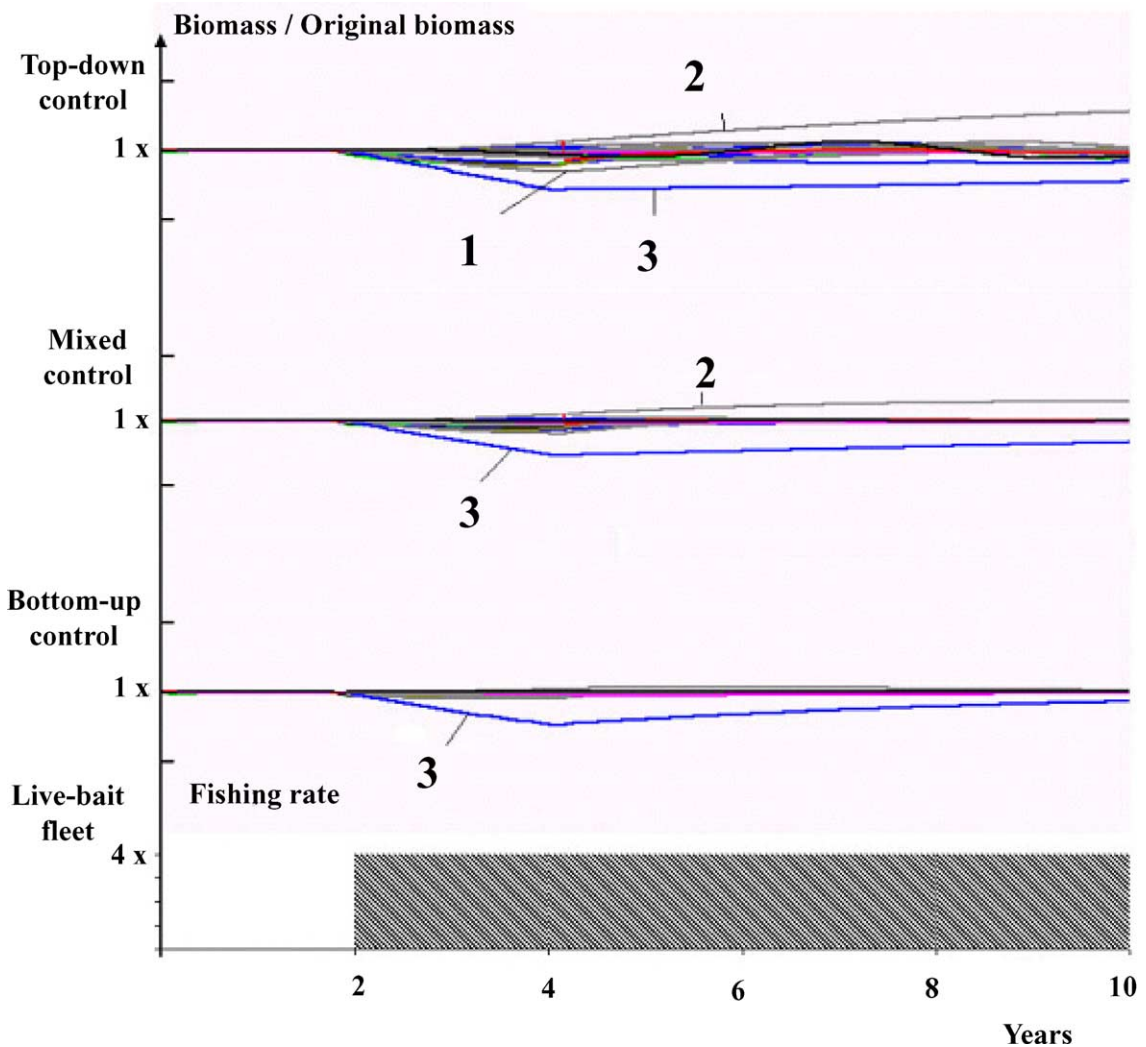


Fig. 6. Ecosim simulation outputs of the effects of increasing live baitfish fleet fishing rate from year 2 onward during a 10-year period, under three alternative types of flow control: top-down, mixed, and bottom-up. Trophic groups are indicated by numbers (1: croaker, 2: catfish, 3: sharks and rays).

5. Discussion

5.1. The mass-balance model and the “bottom-up” perspective

Oceanographic studies in the SBB highlight that physical characteristics determine the extent of the export and the size of productive habitats, i.e. the upwelling of deep, nutrient-rich water can enrich the euphotic zone by introducing new nitrogen, leading to

enhanced planktonic production and biomass (Metzler et al., 1997). The present SBB ecosystem model was constructed to represent the period 1998–1999, recognized by oceanographers as corresponding to a La Niña recovery phase.

In fact, persistent El Niño-like anomalies were evident in the tropics from 1991 to 1995 with the strongest positive temperature anomaly occurring during 1997, disrupting productivity patterns; followed by the strongest negative temperature and positive

nitrate anomalies in 1999, leading for example to the Peruvian anchoveta recovery (Chavez et al., 2002). Studies revealed that the Pacific Decadal Oscillation (PDO) changed sign in late 1990s apparently ameliorating the biogeochemical and ecological impacts of 1997 El Niño. The La Niña event began in 1998 when ocean productivity rebounded to higher than normal levels (Behrenfeld et al., 2001; Chavez et al., 2002). Therefore, fisheries catches considered in this model, may somehow be influenced by the ENSO transition, as speculated by Tomás et al. (2000) for cephalopods.

Physical complexity of the SBB increases fertilization possibilities into the sunlight zone during enough time to allow phytoplankton growth and biomass accumulation (Gaeta and Brandini, 2002). Its neritic zone is considered one of the most productive areas of the Brazilian shelf, also reaching high values of primary productivity in shallow waters and bays (Sevrin-Reyssac et al., 1979) due to coastal degradation, and influencing organic material flow through the pelagic and benthic food webs. Isotopic nitrogen experiments indicated the mesotrophic conditions of the SSB inner shelf (Gaeta, 1999), although predominated in average by regenerated production (Metzler et al., 1997). Zooplankton metabolic activity represents an organic and inorganic compounds cycling mechanism in the water column. Pelagic species recruitment is dependent on the proper availability of planktonic food, and coast-ocean zooplanktonic biomass gradient emphasises the inner shelf higher productivity comparing with its outer shelf and oceanic zone (Resgalla et al., 2001; Lopes et al., 2001).

The importance of detritus in the system is higher than in upwelling areas and represents a vital base for the benthic pathway. Also, the run-off contribution may have an important role bringing alloctonous particulate organic carbon to the shelf. Relevant organic sinking processes contribute to the benthos food supply, although new production is considered its main source (Pires-Vanin, 1993). Benthic groups were extremely important in terms of biomass. Rocha et al. (1998) estimated that the important amount of benthos biomass in summer (76.9 g m^{-2}) should have been underestimated due to sampling limitations. Therefore, although present model total benthic biomass estimates seem to be high, it is to be found between range values from Rocha (1998) ($57\text{--}101.2 \text{ g m}^{-2}$) and those calculated from Amaral (1999) database (73 g m^{-2}).

In terms of model evaluation, the present model could be improved in several aspects, such as by comparing isolated estimates of biomass and instantaneous mortality rates by other methods not presently available. Also, several groups could be split between adult and juvenile pools, and others be added in order to obtain a more complete picture of the system.

The introduction of mammals and seabirds interaction with the fisheries and within the system seems to have an important additional input to present model. Although estimates on these groups are reasonably preliminary it may not underestimate their important predation role in the ecosystem. By the other hand, the importance that the fisheries, especially the sea-bob shrimp fisheries have on seabirds survival by promoting food availability was not yet quantified.

Another important aspect of the model that could affect simulations of the effect of fishing on the ecosystem is the uncertainty about regional feeding habits of several key species. Although presently global literature and databases can help to fulfill gaps in models construction (i.e. FishBase) it is important to emphasize the need for further local feeding studies expressed as a percentage of weight. For example, the role of cephalopods is a key factor in this ecosystem functioning. The present modelling has shown high model susceptibility to changes in the input diet of cephalopods groups based on different data sources (piscivory or zooplanktivory different degrees).

Nevertheless, and when comparing other shelf models of southern Brazil, the present model seems to represent so far the most up-to-date arrange of selected trophic groups and present fisheries locally-gathered information for the area.

5.2. Simulation of ecosystem responses and the "top-down" perspective

Model simulations allowed the exploration on the effects of different trophic control assumptions in predicting the changes in the food web. The predicted biomass changes are more pronounced under top-down control that under bottom-up, as was also obtained by Mackinson et al. (1997), Shannon et al. (2000), and Vasconcellos and Gasalla (2001). The marked increase in the biomass of a competitor species due to a cascade effect in the food web was not observed under bottom-up, when competition for

food resources were limiting interactions. Walters et al. (1997) stated that bottom-up control generally predicts that stocks can sustain much higher fishing pressure before it begins to decline. This occurs because predation mortality rate tends to remain more stable while the consumption rate of lower trophic levels increases with the decrease in biomass, to make them more productive per biomass due to donor control of total food eaten.

In this case, ecosystem responses show quantitatively different results under both hypothesis, and therefore the SBB simulations appear to be very sensitive depending on the assumption.

For example, when increasing fishing mortality for squids, opposite tendencies in biomass changes can be observed especially for the upper food-web groups (Fig. 4).

Squids are important prey of fish and vertebrate species, and predators seems to be affected by their reduction, such as for instance the weakfish (Fig. 4).

Also, triangles of meshed trophic interactions may impact biomass dynamics within all simulations (Figs. 4 and 6; Table 5). Several triangles of temperate food webs have been exemplified as being either symmetrical or asymmetrical (Ursin, 1982; Beverton, 1985). Although there are some simple triangles (i.e. weakfish-squids-cutlass fish), the present model triangles are not always straightforward (Table 4).

Also, the complexity of SBB species interactions can be observed by the number of “beneficial predation” interactions in the model (more than 40). This occurs when a predator may have a direct negative impact on its prey but also an indirect positive effect, by consuming other predators and competitors of their prey (Ulanowicz and Puccia, 1990; Pauly et al., 2002). Examples of beneficial predation of squids are by large pelagic fish, shark and rays, flatfish and seabirds.

Overfished tropical marine ecosystems often show an increase in squid catches while the fisheries intensifies (Pauly, 1985; Christensen, 1998). Could the increase in squids catches in the SBB be an ecosystem consequence of overexploiting higher trophic level species, such as weakfish? Fig. 5 shows this possibility, where the decrease of the most important squid predator (weakfish) and another important predator in the system (Large pelagic fish) was simulated by increasing their fishing rates.

In fact it would not be surprising that the overfishing of higher order consumers can benefit small, widespread and rapidly growing groups, given that reducing all predators in the system is expected to increase the abundance of preys. Also, the squids' high turnover rate can allow them to withstand higher exploitation rates than slower growing species (Christensen, 1998; Hall, 1999).

Actually, the heavy exploitation of weakfish by the bottom-trawlers (Castro, 2000), could benefit the increment in squid biomass, when its effort is strongly increased (Table 5). Bottom-trawlers have a heavy impact on various squid predators, and could try to take advantage from the increasing summer squid catches, as reported by Pérez (2002). Also, other speculations can be made about the effect of shrimp-trawlers effort reduction on squids (Table 5). The extent to which a seasonal shrimp closure could affect a seasonal biomass pulse, such as the squid's, is rather questionable.

Simulations of increasing live-baitfish fleet did not show prominent impact in the inner shelf, where present F values are low. Given that young sardines concentrate in shallow waters, where the conflicts with seiners and artisanal sector converge, a complementary model of the adjacent coastal system and artisanal fisheries could better address the fleet's impact. Nevertheless, the decrease of sharks and rays reflect both their vulnerability and the indirect triangles of trophic interactions surprisingly linking them with the harvesting of young sardines (Fig. 6). Their slow recovery time, being on slow-growing with low reproductive rates, make them a vulnerable group, as already reported (Walker, 1996; Quero and Cendrero, 1996).

5.3. Flow controls

Top-down and bottom-up controls are both likely to act on ecological communities (Hunter and Price, 1992). Questions are being posed instead as to the nature of the links between bottom-up and top-down influences. The variation in flow types observed in ecosystems can be a consequence of heterogeneity of communities and species interactions. Both theories, top-down control with trophic cascades (predator control) (Carpenter et al., 1985; Carpenter and Kitchell, 1993) and bottom-up, donor control (Hunter and Price,

Table 4
Diet matrix

Prey/predator	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	
Phytoplankton	0.001	0.950	0.002	0.100	0.800		0.200		0.056			0.007		0.001								
Shrimps										0.300	0.205	0.175		0.505	0.085		0.088			0.059	0.273	
Zooplankton	0.005	0.050		0.120	0.200	0.010	0.800	0.210	0.811	0.384			0.212				0.026	0.111	0.286			
Mullets																	0.106				0.005	
Detritivorous benthos	0.015		0.002	0.050		0.080		0.480	0.022		0.379	0.409	0.212	0.024	0.253	0.348	0.015	0.056		0.125	0.009	
Juvenile sardine										0.005			0.106	0.024			0.044	0.167				
Catfish																					0.072	
Sardine										0.005			0.106	0.024			0.044	0.167		0.001		
Carnivorous benthos			0.002			0.060		0.100		0.201	0.338	0.292	0.227	0.012	0.637	0.640	0.015	0.056		0.129	0.075	
Mackerel												0.015					0.115				0.070	
Squids									0.100	0.060			0.136	0.094	0.010		0.141	0.185	0.571	0.079	0.092	
Croaker								0.001				0.029						0.143	0.012	0.005		
Other sciaenids								0.001			0.036	0.015		0.165	0.010			0.093		0.011	0.313	
Cutlass fish														0.118						0.099	0.003	
King Weakush											0.041	0.058		0.035						0.092		
Triggerfish																	0.123					
Octopus																0.010		0.002		0.002		
Large pelagic fish																	0.053			0.079		
Other piscivores fish																	0.009			0.007		
Weakfish Cynoscion															0.005		0.018	0.091		0.012	0.226	
Sharks and rays																						
Flatfish										0.045							0.002		0.074		0.053	
Cetaceans																						
Seabirds																						
Detritus	0.979		0.094	0.730		0.850		0.208	0.011													
Import			0.900														0.202			0.100		

Table 5
Predicted relative change in squid biomass after 10 years under different fishing scenarios

Scenario	Fleet	Effort multiplier	Biomass change	
			Bottom-up	Top-down
1	Shrimp trawlers	4	-0.76	-1.72
2	Shrimp trawlers	2	-0.39	-0.91
3	Shrimp trawlers	0.5	0.19	0.47
4	Pair-trawlers	4	1.85	8.8
5	Pair-trawlers	2	-0.54	-2.3

1992), have dominated the ecological debate (Matson and Hunter, 1992).

A bottom-up regime can be defined by a response to changes in productivity at the lowest trophic levels resulting from physical forcing. An increase in productivity at the bottom of the trophic system leads to an increase in the productivity and abundance at all higher trophic levels. Each trophic level is food limited not predator limited. (Bundy, 1997).

In top-down control, an increase in the abundance of a top predator leads to the prediction of a trophic cascade. That is, an increase in the abundance at trophic level 4 leads to an increase in the rate of predation mortality on trophic level 3, and thus a reduction in its biomass, and so on. An increase in productivity at the bottom would lead only to an increase in biomass at the top of the food chain (Bundy, 1997).

In the oceans, bottom up evidences are easy to find, while top-down are more difficult to follow, but it seems apparent that the trophic structure is being altered either by productivity or by the constant removal of large amounts of fish (Parsons, 1992). The effect of environmental conditions climbs upward the system to meet the effects of fishing cascading down from the top (Caddy, 1993; Rudstam et al., 1994). Leibold (1989) suggested that top-down controls especially the upper trophic levels while in bottom-up, production controls abundance in lower trophic levels. Also a “wasp-waist” control was suggested by Rice (1995) representing general upwelling ecosystems regulated up and down from their middle, which is usually occupied by small pelagic fish.

The bottom-up perspective has dominated Brazilian oceanographic literature and explanations on ecosystem’s enrichment processes. By addressing the impact of fishing through ecosystem modelling, a

complementary top-down perspective can be added. Actually, the effect of predation pressure, including exhaustive fishing effort exerted mostly during the last 40 years, had never been evaluated on a multi-species basis before. While there are wide evidences of “bottom-up” processes in the shelf ecosystem, the “top-down” type of inner flow control has been almost neglected especially due to methodological constraints. This has an important implication for management purposes, because it can be too risky to underestimate overfishing ecosystem effects (Christensen and Walters, 2002).

5.4. Methodological perspectives for management

The rudimentary task for fishery managers is the optimal utilization of a renewable resource through time. The twentieth century has witnessed a number of major management shifts in the pursuit of this goal as regulators have struggled with increasingly powerful harvesting technologies, the deleterious affects of open-access regimes, and a general failure to prevent fishing fleets from becoming overcapitalized (Hayley, 1999). Some progress has been made towards mitigating open-access inefficiencies, such as through catch or access control in many fisheries, and the recognition of the effects of fisheries on ecosystems (Botsford et al., 1997; Hall, 1999). However, ecosystem-based fishery management is still very new in world fisheries reality, remaining as an attractive concept rather than a formal activity.

Present bottom-up processes evidence in the South Brazil Bight should not dismiss the important role of a top-down control. Based on our results, we suggest that, in order to be precautionary, SBB decision-makers should not underestimate the effect of fishing effort on the whole system, and consider that a top-down control could actually dominate for the upper food chains.

Therefore, the definition of conservative exploitation rates would be required, such as by control of fishing effort by fleet type. Also, fishing access and multispecies catch restriction could be undertaken. Pauly et al. (2002) recently recommended that a “restoration ecology for the oceans” urges and the access restriction through marine reserves networks located in appropriate specific areas, as well as the enforcement improvement by satellite monitoring,

could be more effective mechanisms. Socio-economic evaluations, community involvement, ecolabelling and public education could benefit measures implementation. Further work on exploratory Ecosim management scenarios on the effect of different fleets could contribute by incorporating both interspecific interactions and physical environmental influences.

6. Conclusions

In order to improve management decisions by predicting the effect of fishing, it would be recommended to better understand the way the ecosystem functions in terms of internal control between components, especially from upper trophic levels. Nevertheless, a precautionary measure would be to assume that top-down control is the dominant force in energy flows. In order to implement such an assumption in an ecosystem approach to fisheries management, fishing effort reduction mechanisms should be evaluated and implemented.

This paper shows the importance of integrated fishery management, because the effect of a single change in fishing strategies can have consequences throughout the food web.

Methodologies, experiments and approaches that could put the management of fisheries on an ecosystem basis must be encouraged as useful scientific progress. They could, in the near future, allow the management of human uses of the ocean in a manner consistent with the existence of ecological interactions.

Acknowledgements

We are grateful to Francisco Arreguin-Sanchez and Marcelo Vasconcellos for helpful comments during modelling; and Villy Christensen and Daniel Pauly for bringing instruments through the INCO-DC concerted action “Placing fisheries in their ecosystem context”. Special thanks are given to Yasunobu Matsuura, Koichi Ara, Charles Gorri, Marcia R. Oliveira, Nilamon Leite Jr., Acacio Tomás, Angel Pérez, Cristina Cergole, Otto Gadig, Tatiana Neves, Fabio Olmos, Alexandre Zerbini and Salvatore Siciliano, for providing useful and even unpublished information on zooplankton, shrimps, octopus, squids, sardines,

sharks, seabirds and cetaceans, respectively. Also, we would like to thank all the people who produced data on which the model was based, and three anonymous referees for allowing the improvement of the article.

This is part of a project sponsored by the IFS (International Foundation for Science) and USAID (United States Agency for International Development) through WWF/Brazil. CNPq/Brazil provided sponsorship to the author. This publication was partly funded by the European Commission’s INCO-DC program through contract no. ERBIC18CT97175.

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