PANAMJAS
Pan-American Journal of Aquatic Sciences

# Assessing fishing policies for northeastern Brazil

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**Abstract.** This study is a first contribution towards the development of ecosystem-based fisheries management in northeastern Brazil, through the exploration of fishing policies based on a trophic model (Ecopath with Ecosim). Our simulations for 1978-2028 indicated that current fishing effort is completely unsustainable for lobsters (*Panulirus argus* (Latreille, 1804) and *Panulirus laevicauda* (Latreille, 1817)) and swordfish (*Xiphias gladius* Linnaeus, 1758). The simulation of optimum fishing policies led to a diverse fleet configuration when ecosystem health (ecological scenario) was emphasized. If the main objective was economic or social (or a combination of both and ecosystem health), manual gathering of coastal resources and demersal industrial fisheries could be boosted, while the lobster and longline fisheries should be phased out. A 50% reduction in effort for lobster fisheries would not produce significant changes in lobster biomass; a reduction in effort to the 1978-level ($f_{MSY}$ – effort leval corresponding to the maximum sustainable yield) would lead to biomass recovery. An improvement in the collection system of fishery statistics (catch and effort) is recommended, as well as further gathering information on biological, economic, and social components of this ecosystem and its fisheries.

**Key words:** ecosystem modelling, Ecopath with Ecosim, fisheries management, trophic model, East Brazil, LME.

**Resumo. Avaliação de políticas pesqueiras para o nordeste do Brasil**. Este estudo é a primeira contribuição para a gestão pesqueira ecossistêmica do nordeste do Brasil, através da exploração de políticas pesqueiras baseadas em um modelo trófico (Ecopath with Ecosim). Simulações realizadas para o período de 1978-2028 indicaram que o atual esforço de pesca é completamente insustentável para a lagosta (*Panulirus argus* (Latreille, 1804) e *Panulirus laevicauda* (Latreille, 1817)) e para o espadarte (*Xiphias gladius* Linnaeus, 1758). A simulação das políticas ótimas de pesca levou a uma configuração de frota diversa quando a saúde do ecossistema (cenário ecológico) foi enfatizada. Se o objetivo principal era econômico ou social (ou uma combinação de ambos com a saúde do ecossistema), a coleta manual de recursos costeiros e a pesca industrial demersal poderiam ser incentivadas, enquanto a pesca da lagosta e a de espinhel deveriam ser fechadas. Uma redução de 50% do esforço da pesca da lagosta não produziria mudanças significativas na sua biomassa; uma redução no esforço para o nível de 1978 ($f_{RMS}$ – esforço correspondente ao rendimento máximo sustentável) levaria a uma recuperação da biomassa. Recomenda-se uma melhoria do sistema de coleta de estatística pesqueira (captura e esforço), assim como a coleta de informações mais detalhadas relativas aos aspectos biológicos, econômicos e sociais desse ecossistema e de suas pescarias.

**Palavras-chave:** modelagem ecológica, Ecopath with Ecosim, gestão pesqueira, modelo trófico, Plataforma do Leste Brasileiro, Grande Ecossistema Marinho.

## Introduction

There is a growing interest in ecosystem-based fisheries management (EBFM) in recent years resulting from a better understanding of the near impossibility of optimally exploiting a number of target stocks (a common management goal) that interact with other target and non-target species (Walters *et al.* 2005). There is also an increased concern with the effect of fisheries on the habitats and on non-target species, mainly those long-lived and with low reproductive rates such as cetaceans, seabirds, sea turtles, and sharks (see, e.g., Hall *et al.* 2000).

The importance of ecosystem considerations was recognized as early as the mid-1950s, even though a low priority was assigned to them (Schaefer 1956). EBFM is thus not a new idea from the theoretical point of view, but there is much to be learned about how to make it operational and ultimately how to measure its success. Several issues should be considered and ranked by priority in specific ecosystems: geography of the system, abiotic factors, key species, species interactions, aggregate and system level properties, and fisheries (Link 2002). The analysis of each of these issues requires the establishment of an open dialogue among all stakeholders involved in ecosystem-related activities in order to decide on goals, objectives, strategies, indicators of performance at the ecosystem level, monitoring system, and feedback mechanisms.

The complexity of marine ecosystems often leads to responses to management measures that are not always foreseen in single species management plans. In attempts to protect stocks from the direct impact of fisheries, technical measures such as the use of nets with larger mesh sizes are one of the most traditional measures recommended since the establishment of fisheries science in the late 1800s. In a search for a better understanding of the trophic interactions between the species in an ecosystem, it was found that larger meshes may decrease the impact on non-target species that act as competitors of target species, and thus have a deleterious effect on the latter. Along these lines, Walters & Kitchell (2001) postulated a phenomenon of ‘cultivation/depensation’: if high fishing pressure reduces the abundance of a previously abundant species, then even after a reduction in fishing effort, the population may be unable to recover due to the predation release of some prey of the target species that act as competitors (or predators) of juveniles of the target species. Testing for this effect is possible using multi-species models, and the results have profound policy implications.

Fisheries are not only about ‘fishes’ and their competitors and/or predators. Fisheries are intended to generate economic benefits, including employment for people. With a growing world population, particularly in developing countries, and an increasing demand for fish protein (Delgado *et al.* 2003), there is a need to consider different ecological, economic, and social goals under a much higher pressure than earlier. One way this can be accommodated is through multi-objective functions that include these three dimensions of fisheries to identify an ‘optimum’ use of a natural resource (see, e.g., Healey 1984).

Considering the importance of ecosystem modelling, several models were built for southern and southeastern Brazil in the late 1990s and early 2000s (Rocha *et al.* 1998, Vasconcellos 2000, Vasconcellos & Gasalla 2001, Gasalla & Rossi-Wongtschowski 2004, Gasalla 2004, Velasco & Castello 2005). However, most of them did not explore the capability of these models to be used in fishing policy exploration, except for the most recent ones. To the north of the study area, Manickchand-Heileman *et al.* (2004) analyzed the ecosystem impacts in the region between Venezuela and Trinidad. For northeastern Brazil, models were built for very small areas (Wiedemeyer 1997, Telles 1998) and none considered fishery components.

In northeastern Brazil, fisheries targeting tunas and tuna-like fishes (*Thunnus albacares* (Bonnaterre, 1788), *Thunnus alalunga* (Bonnaterre, 1788), *Thunnus obesus* (Lowe, 1839)), lobsters (*Panulirus argus* (Latreille, 1804) and *P. laevicauda* (Latreille, 1817)), southern red snapper (*Lutjanus purpureus* Poey, 1876), shrimps (*Xiphopenaeus kroyeri* (Heller, 1862), *Litopenaeus schmitti* (Burkenroad, 1936), *Farfantepenaeus subtilis* (Pérez-Farfante, 1967), and *Farfantepenaeus brasiliensis* Latreille, 1970, and demersal fishes (Sciaenidae) are very important as a result of their bulk catch or the revenue generated by their exports (Paiva 1997). Many of these stocks are overexploited or their exploitation level is unknown (CNIO 1998, Lima & Dias Neto 2002). Lobster is one of the most valuable resources due to its very high price in the international market. Swordfish (*Xiphias gladius* Linnaeus, 1758), shrimps, and southern red snapper catches are also, at least partially, exported. In this paper, the effect of several fishing practices over these stocks will be analyzed using a trophic model, which includes a multi-objective function. Also, the degree in which the current information allows for increasing fishing effort as proposed by national plans will be assessed. Some scenarios for future fishing policies will be investigated, with emphasis on the lobster fishery for which long time series of catch and effort are available.

## Materials and Methods

The Ecopath model constructed by Freire (2005) for the marine ecosystem off northeastern Brazil (Fig. 1) was used as the basis for the simulations of fishing policies for the same region. This model was originally built for 1978, and covers an area of 1,075,000 km$^2$. This region is characterized by rocky substrates and low primary production due to the influence of the warm North

Brazil and Brazil currents (Matsuura 1995). The shelf is mostly narrow (down to 20 km), but reaches up to 220 km at the southernmost part (Ekau & Knoppers 1999). A total of 1,200 km$^2$ of coral reefs are found in this region (Spalding *et al.* 2001) and a seagrass coverage of about 175 km$^2$ (Joel Creed, Laboratory of Benthic Marine Ecology, Rio de Janeiro-Brazil, pers. comm.). The model encompasses a large area that is recognized by its homogeneous features in relation to the areas to the north and south and due to the unability of defining the fishing grounds where catches recorded by state are originating from; within this large area, the fleet has high mobility but catches are mainly recorded in the home port. Additionally, for most resources, there is no detailed information for different stocks.

Some additional fisheries-related data were required and are presented in the following sub-sections. Temporal changes in biomass were assessed through the Ecosim module included in the Ecopath with Ecosim (EwE) software, version 5.1 (Pauly *et al.* 2000, Christensen *et al.* 2004):

$$\frac{dB_i}{dt} = g_i \sum_j^n Q_{ji} - \sum_j^n Q_{ij} + I_i - (M0_i + F_i + e_i) \cdot B_i \qquad (1)$$

where: $dB_i/dt$ = change in biomass of group *i*; $g_i$ = net growth efficiency; $Q_{ji}$ = consumption of group *j* by group *i*; *n* = number of functional groups; $Q_{ij}$ = consumption of group *i* by group *j*; $I_i$ = immigration of group *i*; $M0_i$ = non-predation natural mortality rate of group *i*; $F_i$ = fishing mortality rate of group *i*; $e_i$ = emigration of group *i*; and $B_i$ = biomass of group *i*.

All simulations were performed for a fifty years period through changes in fishing effort for each fleet or fishing mortality for each exploited species (according to data availability) among the 41 groups included in the model (Table I). During the simulations, fishing effort and mortality were allowed to change only for the 2001-2028 period. The period from 1978 to 2000 was used to set vulnerabilities values *sensu* Walters & Juanes (1993) for the main species for which direct or indirect estimates of biomass were available (for details, see Freire 2005). For the remaining species, default values of 2 were used according to Christensen *et al.* (2004). Some of the functional groups (tunas, swordfish, and other large pelagics) have a distribution area beyond the studied area, imposing some limitations to the model.

### *Fleet definition and landing data*

The fishing fleets operating in northeastern Brazil were divided into twelve groups: large pelagic artisanal, small pelagic artisanal, gathering (‘manual collection’), demersal artisanal, longline, reef, lobster, demersal industrial, turtle, toothed cetaceans (‘whaling toothed’), baleen whales (‘whaling baleen’), and snapper. This division was chosen to represent some of the basic dynamics of the fleets operating off northeastern Brazil under the limitation of data availability for several more specific fisheries. Landing data compiled by Freire (2005) were used here after being divided among the twelve fleet types based on the knowledge about the dynamics of each species and fleet/gear. Catches recorded by any broad category indicating ‘other fishes’ and originating from subsistence, recreational, ornamental, and research fisheries were not considered here. ‘Other fishes’ corresponded to less than 2% of the total catch from the region for most of the period, but could reach up to 12% and 16% by the late 70s and 90s, respectively. For the other fisheries, there is no estimate available for the studied area.

### *Ex-vessel prices of ‘fish’ products*

Ex-vessel prices (at dockside) were obtained from SUDEPE (1980) (Table II). No distinction was made for price of products originating from artisanal and industrial fisheries, as prices are estimated by the ratio of total value to total catch (after combining both fisheries) in the original sources. Non-market value was not considered in this analysis.

**Figure 1**: Location of the East Brazil Large Marine Ecosystem (gray) along the Brazilian coast.

**Table I**. Groups used in the simulations of fishing policies and their respective trophic level, biomass, and ecotrophic efficiency in the total area of the East Brazil Large Marine Ecosystem (After Freire, 2005).

| Group name | Trophic level | Biomass* ($t{\cdot}km^{-2}$) | Ecotrophic efficency |
|---|---|---|---|
| 1. Manatee | 2.02 | 0.000 | 0.00 |
| 2. Baleen whales | 3.72 | 0.385 | 0.30 |
| 3. Toothed cetaceans | 4.45 | 0.143 | 0.50 |
| 4. Seabirds | 3.45 | 0.015 | 0.38 |
| 5. Sea turtles | 3.15 | 0.163 | 0.50 |
| 6. Tunas | 4.31 | 0.035 | 0.99 |
| 7. Other large pelagics | 4.50 | 0.026 | 0.72 |
| 8. Dolphinfish | 4.58 | 0.000 | 0.43 |
| 9. Dolphinfish juveniles | 4.42 | 0.001 | 0.31 |
| 10. Swordfish | 4.56 | 0.009 | 0.99 |
| 11. Sharks | 4.65 | 0.032 | 0.60 |
| 12. Rays | 3.88 | 0.101 | 0.41 |
| 13. Small pelagics | 3.05 | 0.604 | 0.99 |
| 14. Needlefishes | 3.43 | 0.115 | 0.99 |
| 15. Southern red snapper | 4.21 | 0.014 | 0.90 |
| 16. Large carnivorous reef fishes | 4.01 | 0.234 | 0.81 |
| 17. Small carnivorous reef fishes | 3.68 | 0.973 | 0.86 |
| 18. Herbivorous reef fishes | 2.00 | 1.116 | 0.39 |
| 19. Omnivorous reef fishes | 3.33 | 1.140 | 0.95 |
| 20. Demersal fishes | 3.36 | 1.347 | 0.95 |
| 21. Mullets | 2.04 | 0.763 | 0.86 |
| 22. Spotted goatfish | 3.50 | 0.050 | 0.24 |
| 23. Benthopelagic fishes | 3.58 | 0.075 | 0.80 |
| 24. Bathypelagic fishes | 3.58 | 1.171 | 0.71 |
| 25. Spiny lobsters | 3.30 | 0.014 | 0.99 |
| 26. Other lobsters | 3.25 | 0.611 | 0.90 |
| 27. Shrimps | 2.73 | 3.901 | 0.99 |
| 28. Crabs | 2.61 | 1.508 | 0.99 |
| 29. Squids | 3.64 | 0.177 | 0.90 |
| 30. Octopus | 3.58 | 0.151 | 0.85 |
| 31. Other molluscs | 2.35 | 2.531 | 0.95 |
| 32. Other crustaceans | 2.17 | 1.591 | 0.95 |
| 33. Other invertebrates | 2.16 | 6.998 | 0.91 |
| 34. Zooplankton | 2.47 | 2.166 | 0.95 |
| 35. Corals | 2.83 | 0.063 | 0.98 |
| 36. Microfauna | 2.00 | 5.989 | 0.09 |
| 37. Phytoplankton | 1.00 | 12.086 | 0.08 |
| 38. Macroalgae | 1.00 | 98.406 | 0.09 |
| 39. Mangroves | 1.00 | 77.762 | 0.00 |
| 40. Seagrasses | 1.00 | 0.052 | 0.09 |
| 41. Detritus | 1.00 | 201.913 | 0.33 |

*Only significant digits are presented.

### *Fishing costs and profits*

Fishing costs (fixed and variable) and profits were obtained from local studies for those fleets for which data were available, and from foreign studies when not (Table III). Data from the crab fishery were considered as representative of manual collection fisheries (Glaser & Diele 2004). For demersal artisanal and lobsters fisheries, data were obtained from Carvalho *et al.* (1996, 2000). No local data were found for the other fleet types. For large pelagic artisanal and industrial demersal fisheries, the information available in Arreguín-Sanchéz (2002) was used as an indication of potential costs and benefits for these fleets. For small pelagic artisanal fisheries, estimates provided by Trinidad *et al.* (1993) were used. Pedrosa & Carvalho (2000)

**Table II**. Price[1] per kilogram of each group caught by each fleet[2] operating off northeastern Brazil in 1978[3].

| Group name / Fleet | LPelArt | SPelArt | Manual | DemArt | Longline | Reef | Lobster | DemInd | Turtle | Whaling Toothed | Whaling Baleen | Snapper |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Baleen whales | – | – | – | – | – | – | – | – | – | – | 4.6 | – |
| Toothed cetaceans | – | – | – | – | – | – | – | – | – | 4.6 | – | – |
| Sea turtles | – | – | – | – | – | – | – | – | 10.8 | – | – | – |
| Tunas | 20.3 | – | – | – | 20.3 | – | – | – | – | – | – | – |
| Other large pelagics | 22.4 | – | – | – | 22.4 | – | – | – | – | – | – | 22.4 |
| Dolphinfish | 26.8 | – | – | – | 26.8 | – | – | – | – | – | – | – |
| Swordfish | – | – | – | – | 21.4 | – | – | – | – | – | – | – |
| Sharks | 10.9 | – | – | – | 10.9 | – | – | – | – | – | – | – |
| Rays | – | – | – | 11.05 | – | – | – | 11.0 | – | – | – | – |
| Small pelagics | – | 12.7 | – | – | 12.7 | – | – | – | – | – | – | – |
| Needlefishes | – | 16.9 | – | – | – | – | – | – | – | – | – | – |
| Southern red snapper | – | – | – | – | – | – | – | – | – | – | – | 23.4 |
| Large carnivorous reef fishes | – | – | – | – | – | 21.6 | – | – | – | – | – | 21.6 |
| Small carnivorous reef fishes | – | – | – | – | – | 16.9 | – | – | – | – | – | 16.9 |
| Herbivorous reef fishes | – | – | – | – | – | 19.9 | – | – | – | – | – | – |
| Omnivorous reef fishes | – | – | – | – | – | 17.9 | – | – | – | – | – | 17.9 |
| Demersal fishes | – | – | – | 16.1 | – | – | – | 16.1 | – | – | – | 16.1 |
| Mullets | – | – | – | 24.8 | – | – | – | – | – | – | – | – |
| Spotted goatfish | – | – | – | – | – | 22.4 | – | – | – | – | – | – |
| Benthopelagic fishes | 14.2 | – | – | – | 14.2 | – | – | – | – | – | – | – |
| Spiny lobsters | – | – | – | – | – | – | 116.4 | – | – | – | – | – |
| Shrimps | – | – | – | 25.0 | – | – | – | 25.0 | – | – | – | – |
| Crabs | – | – | 11.7 | – | – | – | – | – | – | – | – | – |
| Octopus | – | – | 25.9 | – | – | – | – | – | – | – | – | – |
| Other molluscs | – | – | 29.3 | – | – | – | – | – | – | – | – | – |
| Other crustaceans | – | – | – | 35.0 | – | – | – | – | – | – | – | – |

1. Price in cruzeiros (Cr$), the Brazilian currency in 1978 (US$ 1 = Cr$ 17.98 in 1978; www.bcb.gov.br); 2. In this paper, for convenience, the names of the fleets are derived either from their size ("small scale pelagic artisanal"), their gear ("longline") or their target organisms ("lobsters"); and 3. Dashes indicate no catch. Source: SUDEPE (1980).

**Table III**. Cost of each fishing fleet operating off northeastern Brazil for the simulation performed based on the 1970s Ecopath model.

| Fleet | Abbreviation | Fixed cost (%) | Variable cost (%) | Profit (%) |
|---|---|---|---|---|
| 1. Large pelagic artisanal | LPelArt | 2.0 | 33.0 | 65.0 |
| 2. Small pelagic artisanal | SPelArt | 0.6 | 80.0 | 19.4 |
| 3. Manual collection | Manual | 0.0 | 32.9 | 67.2 |
| 4. Demersal artisanal | DemArt | 8.3 | 75.0 | 16.7 |
| 5. Longline | Longline | 18.6 | 73.0 | 8.4 |
| 6. Reef | Reef | 2.9 | 47.1 | 50.0 |
| 7. Lobster | Lobster | 7.3 | 92.5 | 0.2 |
| 8. Demersal industrial | DemInd | 4.5 | 45.5 | 50.0 |
| 9. Turtle | Turtle | 0.0 | 32.9 | 67.2 |
| 10. Whaling toothed | Toothed | 10.0 | 70.0 | 20.0 |
| 11. Whaling baleen | Baleen | 10.0 | 70.0 | 20.0 |
| 12. Snapper | Snapper | 7.3 | 92.5 | 0.2 |

analyzed the cost of longline fisheries based in the state of Rio Grande do Norte. However, it was not possible to incorporate their results in this model due to the lack of a revenue estimate. Similarly, results from Mattos & Hazin (1997) for shark longline could not be used due to limitations imposed by the use of research vessels (as also pointed out by these authors). Costs and benefits from longline fisheries were then obtained for a combination of foreign fisheries targeting tuna and swordfish (O'Malley & Pooley 2003).

Burke & Maidens (2004) provide an estimate of cost of reef fisheries in relation to gross revenue, but do not mention the relation between variable and fixed costs; this partition was based in Arreguín-Sánchez (2002). For turtle fisheries, in the absence of better alternative, the estimate for crab fisheries was used, considering that individuals are collected by the beach in both cases, and are sold or consumed locally. Even though southern red snapper is considered an important target of fisheries in northeastern Brazil, no local data on the economics of this fishery were found. It was assumed that the breakdown of costs and benefits are similar to the lobster fisheries, as both fisheries target the external market (export), use traps (although not exclusively), and the stocks are considered overexploited.

*Optimum fishing policy search*

The first simulation was run considering the maintenance of the 2000-level of fishing effort or mortality for all exploited fishing groups. Secondly, an optimum fishing policy for all fleets operating off northeastern Brazil was assessed based on economic, social, and ecological criteria using the 'fishing policy search' routine available in Ecosim (Christensen & Walters 2004, Christensen *et al.* 2004). The optimum policy search was based on a non-linear search procedure for maximizing the following multi-criterion function:

(2)

$$Max_{output} = w_1 \sum_{i=1}^{k} \sum_{j=1}^{l} NPV_{ij} + w_2 \sum_{j=1}^{l} NJ_j + w_3 \sum_{i=1}^{k} B_i \times \left( \frac{P}{B} \right)_i^{-1}$$

where $NPV_{ij}$ = net present value of the catches of functional group *i* by fleet *j*; *l* = number of fleets = 12; *k* = number of exploited groups = 26; $NJ_j$ = number of jobs generated by unit of monetary value of catches; $B_i$ = biomass; $(P/B)_i$ = production over biomass ratio, obtained from the basic input of Ecopath; and $w_1$, $w_2$ and $w_3$ are weights for the economic, social and ecological criteria (see below). Each criterion was standardized in relation to the value of that criterion in the base-year. A nonlinear optimization method known as Fletch was used in this search (Fletcher 1970). Changes in fishing effort and mortality were applied to the levels observed in 2001 and onwards for all fleets used in the model, except for turtle fisheries and whaling, as these were banned during the 1980s (Singarajah 1997, Marcovaldi & Marcovaldi 1999).

The economic criterion was assessed through the net present value (NPV) of the resources exploited in the future, partially based on the opportunity cost of the capital. NPV was calculated as follows:

(3)

$$NPV = \sum_{t=o}^{T} V_t W_t$$

where: T = simulation period (2001-2028); $V_t$ = net benefit in year *t* = gross revenue minus cost; $W_t$ = weight used to discount the benefit ($V_t$) = $d^t$; d = 1/(1+r); and r = conventional discount rate. In order to consider future generations, an intergenerational discount factor ($d_{fg}$) was added to the simulations, through changes in the weight $W_t$ (Sumaila & Walters 2005):

$$W_t = d^t + \frac{d_{fg} d^{t-1} t}{G} \tag{4}$$

where $d_{fg}$ = intergenerational discount factor in year $t$ and is equal to $1/(1+r_{fg})$; $r_{fg}$ = intergenerational discount rate and it was assumed equal to $r$; and G = generation time (assumed equal to 20 years). Discount rates (r and $r_{fg}$) of 10% were used as Sathaye *et al.* (cited in Van Vliet *et al.* 2003) considers that 8-12% is the usual rate for developing countries.

Gross revenue was calculated as the product between catch and price per kilogram for each functional group. Catch was obtained for 1978 as previously shown and price was obtained from SUDEPE (1980) (Table II).

The social criterion was defined through the number of jobs provided per catch value for each fleet. The estimated number of jobs provided by fishery type in northeastern Brazil is presented in Table IV.

**Table IV**. Estimated number of jobs provided by each fishery type in northeastern Brazil.

| Fishery | No. jobs per R$ 10,000 of catch* | Sources |
|---|---|---|
| 1. Large pelagic artisanal | 1.9 | CEPENE (2000a), Database in Freire (2005) |
| 2. Small pelagic artisanal | 7.6 | CEPENE (2000a), Database in Freire (2005) |
| 3. Manual collection | 7.1 | Costa-Neto & Lima (2000), Alves & Nishida (2003) |
| 4. Demersal artisanal | 0.8 | CEPENE (2000a), CEPENE (2000b) |
| 5. Longline | 0.1 | Evangelista *et al.* (1998), Database in Freire (2005) |
| 6. Reef | 1.4 | CEPENE (2000a), Database in Freire (2005) |
| 7. Lobster | 3.2 | Castro e Silva *et al.* (2003), CEPENE (2000a) |
| 8. Demersal industrial | 7.4 | CEPENE (2000a), CEPENE (2000b) |
| 9. Turtle | 3.4 | Based on manual collection and the price/kg for turtle |
| 10. Whaling toothed | 2.1 | SUDEPE (1979), Singarajah (1985), Rabay (1985) |
| 11. Whaling baleen | 0.4 | SUDEPE (1979), Singarajah (1985), Rabay (1985) |
| 12. Snapper | 0.6 | Based on lobsters, considering the price/kg for snapper |

* In 'reais' = Brazilian currency from 1994 onwards; US$ 1 = R$ 2.47 (As in May 2005).

The ecological criterion intended to maximize the biomass of long-lived animals, i.e., the biomass of each functional group was weighted by the inverse of its production/biomass ratio. The rationale behind this approach is that the presence of long-lived organisms with low P/B ratios is associated with the maturity of an ecosystem (Christensen *et al.* 2004).

The partial effect of each criterion was assessed through five scenarios depending on the management objectives assumed for the study area:

(a) Economic: maximization of the economic rent (net benefit = revenue-cost); $w_1$ was set to 1; $w_2$ and $w_3$ were set to zero;

(b) Social: maximization of number of jobs; $w_2$ was set to 1; $w_1$ and $w_3$ were set to zero;

(c) Ecological: maximization of ecological benefits; $w_3$ was set to 1; $w_1$ and $w_2$ were set to zero.

(d) Compromise: in this scenario, all three components (economic, social, and ecological) were considered equally important and $w_1$, $w_2$, and $w_3$ were set to 1.

(e) Mandated rebuilding: in this scenario, $w_1$ and $w_2$ were equal to 1; $w_3$ was set to 5 and the model was forced towards a policy aiming to increase the biomass of sharks. The mandated relative biomass of sharks was set to 1, indicating that the biomass of this group should maintain the value estimated for the year of the base model (1978).

### *Lobster fishery and future fishing policies*

A simulation was run in Ecosim for 50 years (1978-2028) in order to analyze the effect of changes in the effort employed by lobster fisheries in relation to the 2000 effort level under four scenarios: (a) same (constant) effort level as observed in 2000 (*status quo*); (b) decrease of 50% in effort over the first year and constant thereafter (1/2 effort); (c) ban of lobster fisheries; (d) recovery plan (to the 1978 biomass level). For all other fleets, effort was considered constant at 2000 levels for the period 2001-2028.

## Results

The simulation of the performance of the main functional groups in the marine ecosystem off northeastern Brazil under the current fishing regime indicates that spiny lobsters will be extirpated from the system within a few years (Fig. 2). Biomass of swordfish would also decrease to unsustainable levels in a short time. Groups such as sharks and other large pelagics will continue to decline slightly with their biomass stabilizing at 81% and 69% of the 1978-level, respectively. Decreasing biomass of sharks may lead to a slight increase in tunas' biomass due to the decreasing predation pressure. An increase in biomass would be expected for baleen whales. Southern red snapper biomass would stabilize at the 2000-level.

The simulations for the optimum configuration of the fleet of northeastern Brazil as a whole indicated that very similar configurations would result when trying to maximize rent, number of jobs, and the compromise (rent, jobs, and biomass of less productive groups, all equally weighted). In all three cases, the effort of manual collection for crabs and other molluscs could be increased to more than 100 times the current level (Fig. 3). The effort of demersal industrial fisheries could also be largely increased, although to an extent less than the effort of manual collectors. One important difference among these three scenarios is that small pelagic artisanal fisheries are recommended to increase in order to produce jobs under the social and compromise scenarios, but not from the economic point of view. In terms of maximizing number of jobs per catch value and profit, fisheries such as lobsters and reef fisheries would have to be shut down. In order to maximize the ecological structure of the system, five fisheries could maintain their current fishing pressure: small pelagic artisanal, manual collection, artisanal and industrial demersal, and reef fisheries (Fig. 3). Longline and lobster fisheries would have to be practically shut down. On the other hand, large pelagic artisanal and snapper fisheries could increase the fishing pressure to 6 and 2 times the 2000-level, respectively.

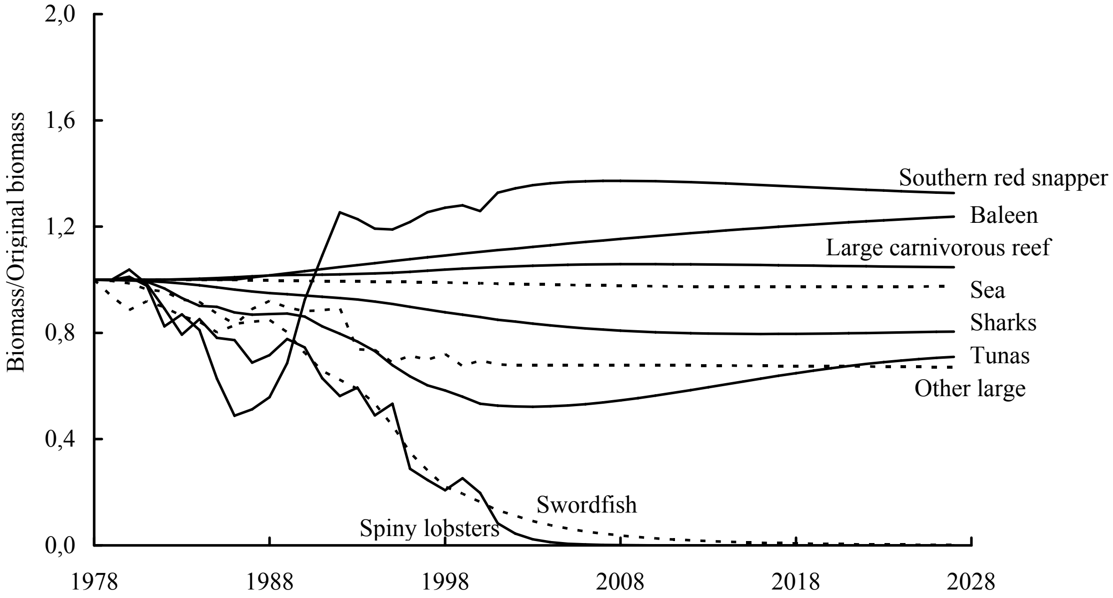

**Figure 2**: Biomass of functional groups that showed variation in relation to the original biomass (1978) for northeastern Brazil during the simulations for 50 years. Fishing rates for 2001-2028 were set at levels observed in 2000.

The recommended configuration of the fleet would result in changes in biomass very similar among the four scenarios. Particularly, one can note that the biomass of swordfish, tunas, and spiny lobsters would increase in all scenarios, with higher rates in the ecological scenario (Fig. 4). Most of the remaining groups would have their biomass constant during the simulated period or slightly decreased. Note that lesser groups in the ecological scenario reach lower biomass than in 2000.

All scenarios discussed seem to be somewhat masked by the interaction between sharks and swordfish/tunas. In order to increase the biomass of tunas and swordfish (two groups with a high trophic level and high market value), sharks (which prey on tunas and swordfish) had to be fished out. As this is an unacceptable scenario due to the current effort towards the conservation of sharks (FAO 1999), a fifth scenario was run where a recovery of shark biomass was aimed for. The results of this scenario, which considers economic, social, and mandated rebuilding as equally important and attributes a higher weight in the multi-criterion function to the ecological component, are similar to the compromise scenario. However, fishing effort of large pelagic artisanal and longline fisheries should

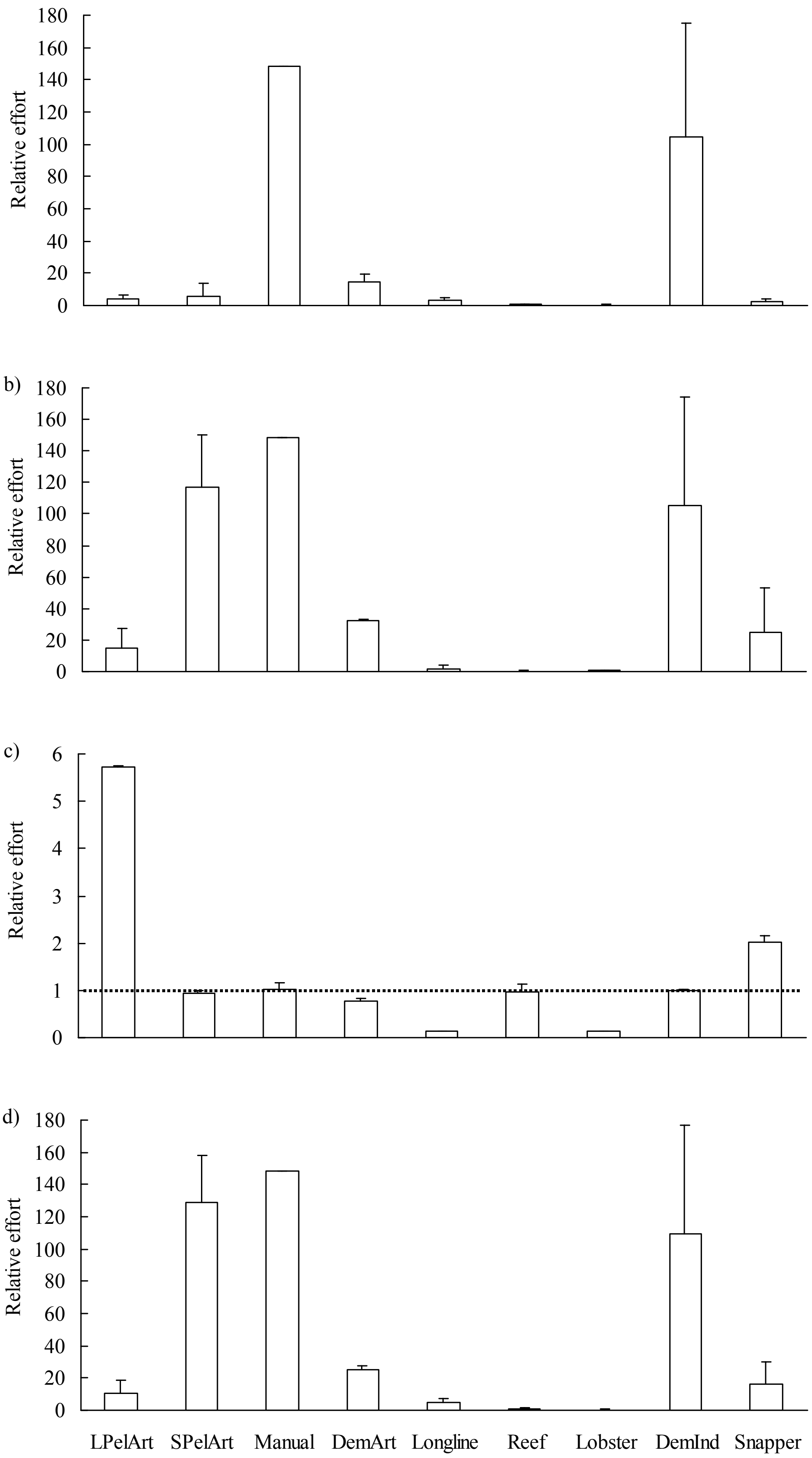

**Figure 3**: Relative changes in fishing effort ($f_{2028}/f_{2000}$)for each fleet included in the 1978 model for northeastern Brazil after a simulation from 2001 to 2028 under four scenarios: a) economic, b) social, c) ecological, and d) compromise ($n_{runs}$ = 10). Columns represent means and whiskers are means plus one standard deviation. Note the different scale used in the vertical axis for the ecological scenario (c). The horizontal dotted line in (c) indicates the 2000 effort level (not shown for the other scenarios).

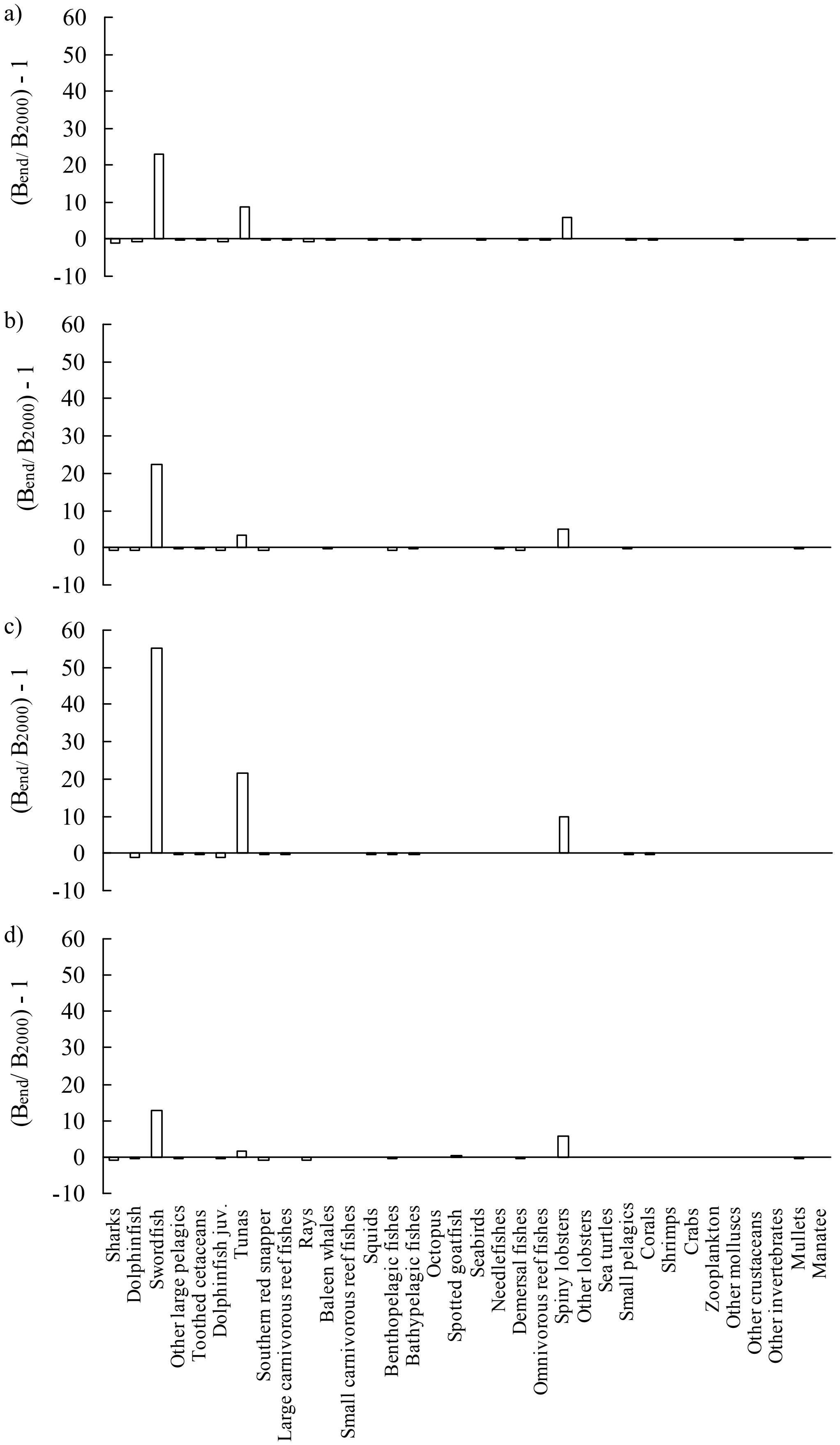

**Figure 4**: Mean relative change ($B_{end}/B_{2000} - 1$) in biomass for all functional groups (trophic level ≥ 2; arranged in decreasing order) obtained by simulations for the period 2000-2028 under four optimization scenarios: a) economic, b) social, c) ecological, and d) compromise ($n_{runs} = 10$). Compromise indicates that an equal weight was given to the components a-c.

**Table V**. Changes in biomass and catch for spiny lobsters caught by lobster fisheries in northeastern Brazil estimated in relation to 1978 and 2000 baselines (Year$_{start}$) resulting from changes in effort representing four scenarios (*status quo*, decreasing fishing pressure by 50% in relation to 2000-level, recovery of biomass to the 1978-level, and ban of lobster fisheries).

| | Year$_{start}$ | Year$_{end}$ | Biomass | Catch | Effort |
|---|---|---|---|---|---|
| Observed | 1978 | 2000 | 0.10 | 0.47 | 4.80 |
| | | | | | |
| *Status quo* | 1978 | 2028 | 0.00 | 0.00 | 4.80 |
| < 50% effort | 1978 | 2028 | 0.41 | 1.01 | 2.40 |
| Recovery | 1978 | 2028 | 1.00 | 0.96 | 0.96 |
| Ban | 1978 | 2028 | 1.22 | 0.00 | 0.00 |
| | | | | | |
| *Status quo* | 2000 | 2028 | 0.00 | 0.00 | 1.00 |
| < 50% effort | 2000 | 2028 | 3.99 | 2.01 | 0.50 |
| Recovery | 2000 | 2028 | 9.66 | 1.92 | 0.20 |
| Ban | 2000 | 2028 | 11.83 | 0.00 | 0.00 |

be decreased even further in order to protect sharks (Fig. 5). Lobster fisheries are expected to be shut down as well due to its low profitability. As in most of the other scenarios, manual collection effort could be increased in order to maximize profitability and number of jobs generated by this fishing sector.

In terms of biomass, the mandated rebuilding scenario would lead to an increase of 21% in the biomass of sharks (Fig. 6). On the other hand, it would be expected a decrease of 52% in the biomass of tunas, 62% for southern red snapper, 38% for demersal fishes, 20% for small pelagics, and 29% for mullets. Swordfish would benefit from such a fleet configuration, probably due to a combination of the release from the competition

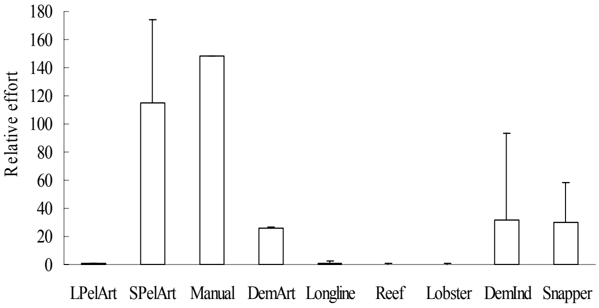

**Figure 5**: Relative changes in fishing effort ($f_{2028}/f_{2000}$)in relation to 2000 for each fleet included in the 1978 model for northeastern Brazil after a simulation for 2001-2028 under the mandated rebuilding scenario aiming at the recovery of shark populations ($n_{runs}$ = 10). Columns represent means and whiskers are means plus one standard deviation.

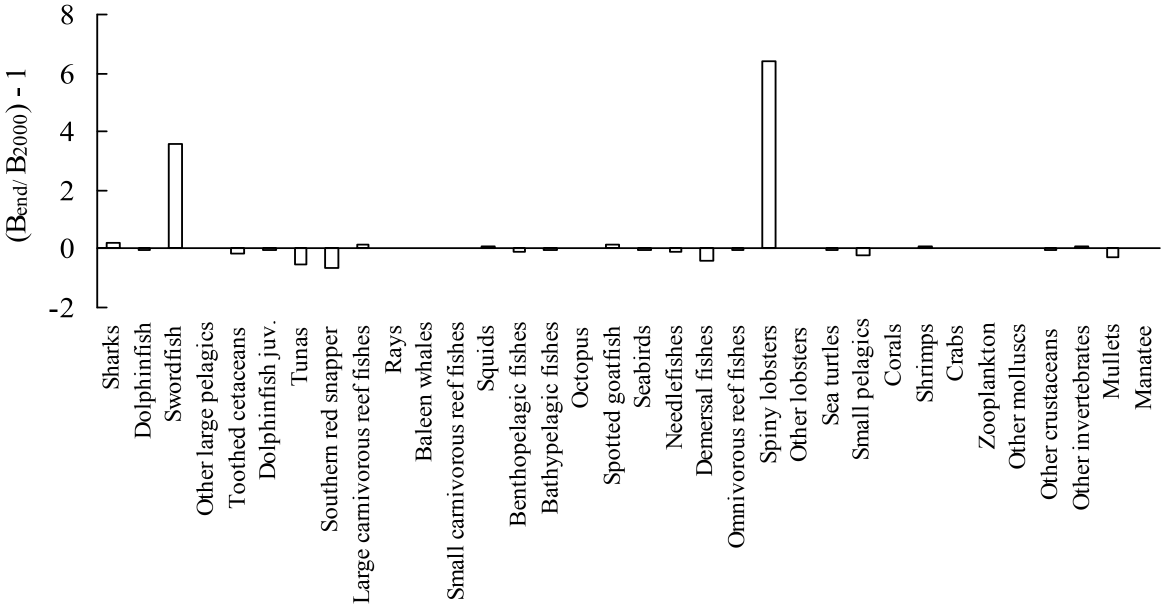

**Figure 6**: Mean relative change in biomass for all functional groups with trophic level (TL) equals or higher than 2 (arranged in decreasing order), obtained by the simulation for 2001-2028 under the mandated rebuilding scenario ($n_{runs}$ = 10). This scenario aims to recover shark biomass to the 1978-level and considers an equal weight for the economic, social, and mandated rebuilding component (1). A weight of 5 was attributed to the ecological component.

imposed by tunas and from the fishing pressure by longliners. A strong increase in biomass of lobsters also resulted from this configuration due to the closure of non-profitable lobster fisheries.

A closer look at lobster fisheries indicates that a reduction of 50% in effort would avoid the looming collapse of the spiny lobsters' stocks and would lead to a slight increase in biomass (Fig. 7). These stocks would reach the same level of biomass as observed in 1978 only if the fishing effort were reduced to the 1978-level. A complete ban of lobster fisheries would result in a biomass 22% higher than the recovery scenario (Fig. 7 and Table V). The more realistic recovery plan would lead to a very high gain in biomass (more than 9 times higher than the 2000-level) and to gains in catch (92%).

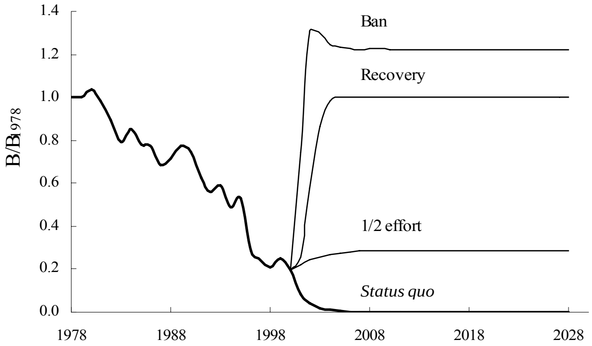

**Figure 7**: Changes is biomass of spiny lobsters in relation to the 1978 baseline after four policy strategies: maintenance of current fishing effort (*status quo*); reduction of effort to 50% of the 2000 effort level (1/2 effort); effort required to reach the 1978 biomass level (recovery); and banning of lobster fisheries (ban).

## Discussion

The results of the simulations presented here indicate that the current levels of fishing pressure would lead to even further decline and eventual collapse of important stocks such as lobsters and swordfish. As these stocks are mainly caught to supply international markets, their collapse would represent an important loss in terms of acquisition of foreign currency through exports. For other target groups such as sharks and other large pelagics, biomass would continue declining slightly for the first years and would stabilize after that.

Avoiding the collapse of main fisheries requires that some decisions are made about the perceived 'value' of different components of an ecosystem and the benefits expected from its natural resources. As simulated here using a defined set of values (reflected in the weights used in the multi-criterion function), the ecological scenario resulted in a more diverse fleet configuration in relation to the other scenarios. Curiously, an increase in fishing effort of the artisanal large pelagic fishery is recommended in this scenario. This probably results from the high catches of sharks by this fishery in relation to the industrial sector (longliners). Sharks prey on groups that have a high B/P ratio, which served as a criterion to define the ecological structure of the system. Thus, the increasing effort for the artisanal pelagic fishery represents an attempt to decrease the biomass of sharks to obtain higher biomass of such groups. However, this scenario is not realistic as the low proportion of sharks recorded by longliners results from an artifact due to the non-inclusion of shark discards resulting from finning practices (Lessa *et al.* 2004). Some effort should be put into determining the magnitude of these discards or hope for their complete ban after international initiatives such as the recent position of the International Commission for the Conservation of Atlantic Tuna (ICCAT) favorable to ban finning practices (Ocean Conservancy 2004).

The compromise scenario suggested a fleet configuration equal to the social scenario and very close to the economic scenario. This reveals the importance of the way the multi-criterion function is defined and the weights attributed to each component included in the function. Zetina-Rejón *et al.* (2004) and others had already mentioned the similarity of the economic and social scenarios when fisheries that result in higher rent are the same that produce more jobs; in this case, manual collection fisheries targeting crabs and molluscs. Promoting manual collection does not seem plausible in terms of fishing policy, due to its essential subsistence feature, but these results indicate that there is potential for growth from a social and economic perspective. However, the biological limits of the target stocks have to be respected and are likely to be affected by pollution in coastal areas caused by urban and industry development and by destruction of mangrove areas (Leão & Dominguez 2000, Marques *et al.* 2004). Additionally, the introduction of a new gear for collecting crabs ('redinha') has resulted in a three-fold increase in catchability, raising concern for the eventual overexploitation of these resources (Ivo & Vasconcelos 2000).

One factor that affects the simulation of future economic gains of alternative fishing policies is the discount rate. A rate of 10% was used in all simulations, which reflects the high discount of future benefits in developing countries. The use of high values usually leads to the degradation of natural resources (Field & Olewiler 2002). This opinion is shared by Costanza *et al.* (1997), who indicate that rates higher than 5% result in unsustainable practices. In fact, some simulations for the extraction of wild palmito in Brazil compiled by Orlande *et al.* (1996) indicated that rates in excess of 8% favored total extraction. All scenarios simulated here, but the ecological one, did suggest a very high increase in the most profitable fisheries. It is recommended that the effect of lower and higher discount rates be assessed in determining the trade-offs among different fishing sectors in northeastern Brazil.

All simulations presented here assumed no change to the ban of manatee, turtle, and whale fisheries, but did not explicitly consider the non-market value of these groups. This value was partially recognized through the use of the inverse of the P/B ratio in the definition of the desired ecosystem structure in the ecological and compromise scenarios. In the mandated rebuilding scenario, this value was set up even higher by the use of a weight of 5 for the ecological component (and 1 for all others). Ultimately, the decision on the relative weight of each component included in the multi-criterion function is to be reached through an agreement among all stakeholders involved in a direct or indirect way with the allocation of natural resources (Healey 1984), within a framework of integrated coastal management that up to now has been considered deficient in northeastern Brazil and, in fact, in the whole country (Marques *et al.* 2004).

The results generated by the simulations indicate that the biomass of several groups would be at lower levels by 2028 than in the 1970s, if current fishing pressure is maintained. Some of them would even collapse in a short time. The bulk of the increase in the fishing fleet proposed by the Brazilian government is directed towards increasing the fleet targeting tuna and tuna-like fishes. The model used as the basis for the simulations run here are not able to correctly duplicate the dynamics of these groups due to their highly migratory behaviour (Freire 2005). Thus, much of the response of the stock biomass to the fishing pressure within the East Brazil Large Marine Ecosystem (off northeastern Brazil) will depend on the fishing effort applied along the whole distribution area of the respective stocks. If the increase in the oceanic fleet size is coupled with negotiations of Brazil within ICCAT to increase its quotas, the scenario would be more positive as the total effort exerted over the stock is not expected to increase. If not, further decline of these stocks, beyond the documented by ICCAT (2004 a,b, 2005) is expected.

For snappers and shrimps, the incompleteness of information evidenced through this simulation exercise indicates that increases in effort proposed by the 'Profrota Pesqueira' program have not been a result of a thorough analysis of these fisheries. No cost-benefit analysis was found for snapper fisheries and the times series of catch and effort are not complete enough to allow for an analysis of CPUE trends (even with all its limitations). Additionally, the interchangeability between vessels targeting snappers and lobsters may pose additional fishing pressure on lobster stocks. The number of licenses allowed for shrimp trawlers in northern Brazil (for which catch and effort data are available) decreased from 250 to 185 in 1997 (Negreiros-Aragão & Silva 2000). The proposal to build new vessels for this fishery counteracts previous attempts to control fishing effort; old vessels may remain in this sector or be re-directed to other fisheries increasing the fishing pressure. Detailed information about the status of shrimp stocks in northeastern Brazil is lacking.

The lobster fishery is one of the best-studied fisheries in Brazil. In 2004, lobster exports yielded US$ 75 million, indicating the importance of this fishery for the local economy. The simulations indicated that this stock is expected to collapse within a few years if current fishing pressure is maintained. The biomass level of spiny lobsters would recover to the level observed in 1978 only if there were a significant decrease in fishing effort to the 1978-level. The resulting fishing effort is equivalent to the effort that would produce the maximum sustainable yield of 8,962 tonnes of spiny lobsters (Ivo & Pereira 1996), a value well above the 6,500 tonnes officially recorded for 2000 (Freire 2005).

One factor not captured in these simulations was the dynamics of industrial and artisanal lobster fisheries due to the absence of effort data for each component. Castro e Silva *et al.* (2003) comment on the transfer of boat ownership from large companies targeting lobsters to artisanal fishers, through special arrangements. Indeed the database presented in Freire (2003) was able to capture this transfer through a change in the bulk of catches originating from industrial fisheries up to 1995 and from artisanal fisheries since then. It is also well known that effort was displaced to the coast of other States where the lobster fishery was not fully developed (IBAMA 1994), and this could mask even more severe local depletions.

The analysis of the dynamics of lobster fisheries as presented here is also limited due to inclusion of one single fleet called 'lobster fisheries'. However, this sector is far from homogeneous as three basic gears are used in northeastern Brazil: traps, diving, and gillnets ('caçoeira'), each one with its own dynamics. Traps were introduced since the establishment of this fishery as an industrial activity in the late 1950s and the latter two were introduced in the 1970s after the decline in the CPUE obtained by lobster traps. Traps maintained legal status since its introduction, but gillnets and diving have oscillated between legal and illegal status in the last two decades. Gillnets are responsible for destruction of the substrate (Paiva *et al.* 1973, cited in Ivo & Pereira 1996, Vasconcelos & Lins-Oliveira 1996). In relation to the size caught, Ivo & Ribeiro Neto (1996) did not find any statistical difference between the mean size of lobsters caught by gillnets and traps in the state of Ceará.

Diving is responsible for the capture of immature lobsters when practiced in waters shallower than 20 m (Lins-Oliveira *et al.* 1997). Diving also leads to numerous deaths and disabilities caused by physiologically inappropriate diving profiles (social cost). It is estimated that 90% of all lobster divers in the state of Rio Grande do Norte have suffered at least one diving accident (Procuradoria Regional do Trabalho-RN 2005). Considering that lobsters have a very high market price, the gains are perceived as higher than the losses, and are worth the risk. This perception is emphasized by higher rents obtained by divers (and gillnets) in relation to trap fisheries (Carvalho *et al.* 1996). A penalization system for such risky fisheries should be included in the maximization of the objective function, which would allow for more realistic simulations, as was done for the New England herring fishery (Healey 1984).

Thus, future developments of this model require the incorporation of the three gear types involved in lobster fisheries. This was not possible in this version of the model due to the lack of effort data for gillnet and diving sectors. Additionally, the recording system of Brazilian fisheries precluded the split of total lobsters catches among these three gear types. The incorporation of the dynamics of each lobster fishery will be a hard task due to the illegal nature of diving activities in northeastern Brazil (Vasconcelos & Lins-Oliveira 1996). The analysis of trade-offs for lobster fisheries in relation to another fisheries occurring in this region also requires a better understanding of relative importance of artisanal and industrial lobster fisheries. Finally, the spatial dynamics of the fleet should be incorporated, possibly through the use of Ecospace, a module available in EwE that allows for spatial simulation (Walters *et al.* 1999).

All the simulations presented here inherited the high degree of uncertainty associated with the input data used in the Ecopath model built by Freire (2005). This uncertainty, associated with institutional instability and the lack of information on the social and economic aspects for most of the local fisheries, leads to a high degree of uncertainty in the results of the simulations. The scenarios simulated here did not consider differences in ex-vessel prices between artisanal and industrial fisheries, as national fisheries statistics only provide combined prices. Additionally, the results are valid only for a scenario of constant price for each exploited group. Negreiros Aragão (2005) points out that the price for Brazilian spiny lobsters has increased in the last years. This increase in price can precipitate the collapse of this stock due to the attraction of additional fishing effort.

The institutional instability in Brazil has some negative effects on the collection of basic data required for managing Brazilian fisheries. First, there is a lack of continuity in the collection of catch data (Lima & Dias Neto 2002). Second, effort data are not properly gathered and/or lack continuity (CEPENE 2000a). Third, there is the chronic problem of attributing catch records to the proper species, a problem long recognized (Welcomme *et al.* 1979). Finally, there is no attempt to estimate unreported or illegal catches at a national level from Brazilian waters. This probably reflects a disconnection between the system responsible for data collection and data users. This negative impact is expected to increase due to the current split of responsibility between two institutions with diverse agendas: SEAP (Special Secretary for Aquaculture and Fisheries) and IBAMA (Institute for Environment and Renewable Natural Resources).

The underestimation of catches also leads to somewhat unrealistic scenarios. For example, catches of dolphinfish (*Coryphaena hippurus* Linnaeus, 1758) are much higher than officially recorded; a sampling program run in northeastern Brazil was able to sample much more dolphinfish than was recorded in statistical bulletins (Rosângela Lessa, pers. comm., Universidade Federal Rural de Pernambuco, Recife/Brazil). Shrimps are mainly exploited by artisanal fishers (84% on average, according to Freire, 2005) and the statistics collection system may be failing to cover it all. The same problem is expected for crabs and other molluscs, which are mainly caught by artisanal fishers in remote areas. Finally, the non-inclusion of by-catch discards in this preliminary version of the model contributes to low exploitation rates.

The simulations presented here will also benefit from splitting tuna and swordfish groups into juveniles and adults, with predation by sharks heavily concentrated on juveniles. Fishing pressure on these groups would be somewhat shared by juveniles and adults: for swordfish, slightly higher proportion of juveniles is caught by longliners in this area (Lessa *et al.* 2004); for yellowfin tuna, the composition is reversed, with slightly higher proportion of adults in the catches (Lessa & Duarte-Neto 2004); for the two other tuna species (bigeye and albacore), no local data were found. Another change recommended is to split the sharks group into two: coastal and pelagic. This would avoid the existence of a 'super shark' group that prey upon both the pelagic and demersal components of the system and thus has an impact on the system higher than expected. This task will be possible only if some effort is spent towards obtaining more detail in the identification of sharks' landings. Fisheries statistics in Brazil record 80-100% of sharks' landings (about 18,000 tonnes for 1978-2000, based on Freire, 2005) as 'cação' or 'caçonete' (unidentified sharks).

Even though Ecosim has showed its usefulness for the exploration of fishing policies in other areas (Christensen & Walters 2004), its application to areas such as northeastern Brazil will depend on more effort to improve the basic trophic model of the region and the collection of basic social and economic data for many of its fisheries. This is not a challenge for an ecosystem-based fisheries management approach only, but reveals the weakness of the current single-species based fisheries management and that many fishery-related decisions are rather based on guesses and on the belief that the large extension of the Brazilian marine waters would always support a production higher than what is biologically feasible.

## Acknowledgements

The authors are grateful to R. Carvalho, V. Isaac, S. Lucato, E. Muto, A. Monteiro, R. Schärer, and G. Velasco for helping with bibliographic support; R. Sumaila for providing support on fisheries economics; to the referees J. N. Araújo and M. Coll for valuable comments and suggestions; CNPq (National Council for the Scientific and Technological Development) and the Pew Charitable Trusts, through the Sea Around Us Project, for financial support.

## References

Alves, R. R. N. & Nishida, A. K. 2003. Aspectos socioeconômicos e percepção ambiental dos catadores de caranguejo-uçá *Ucides cordatus cordatus* (L. 1763) (Decapoda, Brachyura) do estuário do Rio Mamanguape, nordeste do Brasil. **Interciência** 28(1): 36-43.

Arreguín-Sánchez, F. 2002. Impact of harvesting strategies on fisheries and community structure on the continental shelf of the Campeche Sound, southern Gulf of Mexico. **Fisheries Centre Research Reports** 10(2): 127-134.

Burke, L. & Maidens, J. 2004. **Reefs at risk in the Caribbean**. World Resouces Institute, Washington, DC: 80 p.

Carvalho, R. C. A., Ferreira, C. R. C., Vasconcelos, J.A., Oliveira, M.Y.S. & Campos, L.M.A. 1996. Custos e rentabilidade de embarcações envolvidas na pesca da lagosta no nordeste do Brasil, 1995. **Boletim Técnico Científico do CEPENE** 4(1): 233-262.

Carvalho, R. C. A., Oliveira, M. Y. S., Campos, L. M. A., Freitas, S. W. & Silva, E. C. S. 2000. Análise de custo e rentabilidade da captura e beneficiamento de camarão, estados de Pernambuco e Alagoas, nordeste do Brasil, 1997-1998. **Boletim Técnico Científico do CEPENE** 8(1): 279-296.

Castro e Silva, S. M. M., Pontes, A. C. P. & Rocha, C. A. S. 2003. Report on the spiny lobster fishery in Brazil. *In*: Report of the second workshop on the management of Caribbean spiny lobster fisheries in the WECAFC area. Havana, Cuba, 30 September - 4 October, 2002. **FAO Fisheries Report no. 715**: 56-67.

CEPENE 2000a. **Boletim estatístico da pesca marítima e estuarina do nordeste do Brasil - 1999**. Centro de Pesquisa e Extensão Pesqueira do Nordeste/IBAMA, Tamandaré: 150 p.

CEPENE 2000b. **Relatório da reunião técnica sobre o estado da arte da pesquisa e ordenamento da pesca de camarões peneídeos na região nordeste do Brasil**. Centro de Pesquisa e Extensão Pesqueira do Nordeste - CEPENE, Tamandaré, Pernambuco, Brasil: 64 p.

Christensen, V., Walters, C. & Pauly, D. 2004. **Ecopath with Ecosim: a user's guide**. Fisheries Centre, University of British Columbia, Vancouver: 154 p.

Christensen, V. & Walters, C. J. 2004. Trade-offs in ecosystem-scale optimization of fisheries management policies. **Bulletin of Marine Science** 74(3): 549-562.

CNIO, 1998. O Brasil e o mar no século XXI:

Relatório aos tomadores de decisão do país. Rio de Janeiro, Comissão Nacional Independente sobre os Oceanos (CNIO). 408 p.

Costa-Neto, E. M. & Lima, K. L. G. 2000. Contribuição ao estudo da interação entre pescadores e caranguejos (Crustacea, Decapoda, Brachyura): considerações etnobiológicas em uma comunidade pesqueira do estado da Bahia, Brasil. **Actualidades Biologicas** 22(73): 195-202.

Costanza, R., Cumberland, J., Daly, H., Goodland, R. & Nogaard, R. 1997. **An introduction to ecological economics**. St. Lucie Press, Boca Raton, Florida: 275 p.

Delgado, C. L., Wada, N., Rosegrant, M. W., Meijer, S. & Ahmed, M. 2003. **Fish to 2020: supply and demand in changing global markets**. International Food Policy Research Institute/WorldFish Center, Washington, D.C./Penang: 226 p.

Ekau, W. & Knoppers, B. 1999. An introduction to the pelagic system of the North-East and East Brazilian shelf. **Archive of Fishery and Marine Research** 47(2/3): 113-132.

Evangelista, J. E. V., Vasconcelos, J. A. & Oliveira, G. M. 1998. Evolução da pesca de atuns no nordeste do Brasil. **Boletim Técnico Científico do CEPENE** 6(1): 77-108.

FAO 1999. **International Plan of Action for reducing incidental catch of seabirds in longline fisheries. International Plan of Action for the conservation and management of sharks. International Plan of Action for the Management of Fishing Capacity**. FAO, Rome. 26 p.

Field, B. C. & Olewiler, N. D. 2002. **Environmental economics**. McGraw-Hill Ryerson, Toronto: 498 p.

Fletcher, R. 1970. A new approach to variable metric algorithms. **The Computer Journal** 13(3): 317-322.

Freire, K. M. F. 2003. A database of landing data on Brazilian marine fisheries from 1980 to 2000. **Fisheries Centre Research Reports** 11(6): 181-189.

Freire, K. M. F. 2005. Fishing impacts on marine ecosystems off Brazil, with emphasis on the northeastern region. **PhD thesis**, Resource Management and Environmental Studies. University of British Columbia, Vancouver - Canada: 254 p.

Gasalla, M. A. 2004. Impactos da pesca industrial no ecossistema da plataforma continental interna do sudeste do Brasil: a abordagem ecossistêmica e a integração do conhecimento. **PhD thesis**, Universidade de São Paulo, São Paulo, Brazil: 276 p.

Gasalla, M. A. & Rossi-Wongtschowski, C.L.D.B. 2004. Contribution of ecosystem analysis to investigating the effects of changes in fishing strategies in the South Brazil Bight coastal ecosystem. **Ecological Modelling** 172(2-4): 283-306.

Glaser, M. & Diele, K. 2004. Asymmetric outcomes: assessing central aspects of the biological, economic and social sustainability of a mangrove crab fishery, *Ucides cordatus* (Ocypodidae), in North Brazil. **Ecological Economics** 49: 361-373.

Hall, M. A., Alverson, D. & Metuzals, K. I. 2000. By-catch: problems and solutions. **Marine Pollution Bulletin** 41(1-6): 204-219.

Healey, M. C. 1984. Multiattribute analysis and the concept of optimum yield. **Canadian Journal of Fisheries and Aquatic Science** 41: 1393-1406.

IBAMA 1994. **Lagosta, caranguejo-uçá e camarão do nordeste**. Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis, Brasília: 190 p.

ICCAT 2004a. 2003 ICCAT albacore stock assessment session. **Collective Volume of Scientific Papers ICCAT** 56(4): 1223-1311.

ICCAT 2004b. 2003 ICCAT Atlantic yellowfin tuna stock assessment session. **Collective Volume of Scientific Papers ICCAT** 56(2): 443-527.

ICCAT 2005. Report of the 2004 ICCAT bigeye tuna stock assessment session. **Collective Volume of Scientific Papers ICCAT** 57 (2): 3-38.

Ivo, C. T. C. & Pereira, J. A. 1996. Sinopse das principais observações sobre as lagostas *Panulirus argus* (Latreille) e *Panulirus laevicauda* (Latreille), capturadas em águas costeiras do Brasil, entre os estados do Amapá e do Espírito Santo. B**oletim Técnico Científico do CEPENE** 4(1): 7-94.

Ivo, C. T. C. & Ribeiro Neto, J. 1996. Estudo comparativo sobre a pesca de lagostas com covo e rede de espera no estado do Ceará. **Boletim Técnico Científico do CEPENE** 4(1): 95-134.

Ivo, C. T. C. & Vasconcellos, E. M. S. 2000. Potencial reprodutivo do caranguejo-uçá, *Ucides cordatus cordatus* (Linnaeus, 1763), capturado na região estuarina do rio Curimatau (Canguaretama, Rio Grande do

Norte, Brasil). **Boletim Técnico Científico do CEPENE** 8(1): 45-53.

Leão, Z. M. A. N. & Dominguez, J. M. L. 2000. Tropical coast of Brazil. **Marine Pollution Bulletin** 41(1-6): 112-122.

Lessa, R. & Duarte-Neto, P. 2004. Age and growth of yellowfin-tuna (*Thunnus albacares*) in the western equatorial Atlantic, using dorsal fin spines. **Fisheries Research** 69: 157-170.

Lessa, R. P., Nóbrega, M. F. & Bezerra Júnior, J. L. 2004. **Dinâmica de populações e avaliação de estoques dos recursos pesqueiros da região nordeste**. Volume II. DIMAR, Departamento de Pesca - Universidade Federal Rural de Pernambuco, Recife-Brazil: 245 p.

Lima, J. H. M. & Dias Neto, J. 2002. O ordenamento da pesca marítima no Brasil. **Boletim Técnico Científico do CEPENE** 10(1): 265-330.

Link, J. S. 2002a. Ecological considerations in fisheries management: when does it matter? **Fisheries** 27(4): 10-17.

Lins-Oliveira, J. E., Vasconcelos, J. A., Freire, K. M. F., Cunha, K. M. F., Henriques, V. M. C. & Freire, F. A. M. 1997. **Aspectos legais da captura de lagostas no nordeste do Brasil**. VII COLACMAR. Congresso Latino-americano sobre Ciências do Mar. Volume II. 22 a 26 September, 1997, Santos, Brazil.

Manickchand-Heileman, S. C., Mendoza-Hill, J., Kong, A. L. & Arocha, F. 2004. A trophic model for exploring possible ecosystem impacts of fishing in the Gulf of Paria, between Venezuela and Trinidad. **Ecological Modelling** 172(2-4): 307-322.

Marcovaldi, M. A. & Marcovaldi, G. G. 1999. Marine turtles of Brazil: the history and structure of Projeto TAMAR-IBAMA. **Biological Conservation** 91: 35-41.

Marques, M., Costa, M. F., Mayorga, M. I. O. & Pinheiro, P. R. C. 2004. Water environments: anthropogenic pressures and ecosystem changes in the Atlantic drainage basins of Brazil. **Ambio** 33(1-2): 68-77.

Matsuura, Y. 1995. **Os ecossistemas brasileiros e os principais macrovetores de desenvolvimento. Subsídios ao planejamento da gestão ambiental**. Projeto Cenários para o Planejamento da Gestão Ambiental (MMA/PNMA). Brasília, DF, MMA: p. 39-104.

Mattos, S. M. G. & Hazin, F. H. V. 1997. Análise da viabilidade econômica da pesca de tubarões no litoral do estado de Pernambuco. **Boletim Técnico Científico do CEPENE** 5(1): 89-114.

Negreiros Aragão, J. A. 2005. **Liberação da pesca da lagosta com rede caçoeira - uma decisão preocupante**. Digest 119.

Negreiros-Aragão, J. A. & Silva, K. C. A. 2000. Northern Brazil shrimp fisheries. **FAO Fisheries Reports** 628: 37-44.

O'Malley, J. M. & Pooley, S. G. 2003. **Economic and operational characteristics of the Hawaii-based longline fleet in 2000**. Joint Institute for Marine and Atmospheric Research, Honolulu: 31 p.

Ocean Conservancy 2004. **First international ban on shark finning adopted**. www. oceanconservancy.org. Accessed in May 18th, 2005.

Orlande, T., Laarman, J. & Mortimer, J. 1996. Palmito sustainability and economics in Brazil's Atlantic coastal forest. **Forest Ecology and Management** 80(1-3): 257-265.

Paiva, M. P. 1997. **Recursos pesqueiros estuarinos e marinhos do Brasil**. UFC, Fortaleza. 278 p.

Pauly, D., Christensen, V. & Walters, C. 2000. Ecopath, Ecosim, and Ecospace as tools for evaluating ecosystem impact of fisheries. **ICES Journal of Marine Science** 57: 697-706.

Pedrosa, B. M. J. & Carvalho, R. C. A. 2000. **Análise de viabilidade econômica da pesca de atuns e afins no nordeste brasileiro**. Anais do XXXVIII Congresso Brasileiro de Economia e Sociologia Rural, Rio de Janeiro, 2000. Universidade Federal do Ceará, Fortaleza, Brazil: 1-15 p.

Procuradoria Regional do Trabalho - RN 2005. **Acidentes do trabalho**. http://www.prt21 .gov. br/inform2/. Accessed in May 18th, 2005.

Rabay, G.C. 1985. **Baleias. Fatos e mitos**. João Pessoa, Brazil: 34 p.

Rocha, G.R.A., Gasalla, M.A., Rossi-Wongtschowski, C.L.D., Soares, L.S.H., Pires-Vanin, A.M.S., Muto, E.Y., Cergole, M.C., Aidar, E., Mesquita, H.S.L., Gianesella-Galvão, S.M.F., Vega-Perez, L.A. & Jarre-Teichmann, A. 1998. Quantitative model of trophic interactions in the Ubatuba shelf system (Southeastern Brazil). **Naga** 21(4): 25-32.

Schaefer, M. B. 1956. **The scientific basis for a conservation program**. In: Papers presented at the International Technical Conference on the Conservation of the Living Resources of the Sea, United Nations Publication 1956. II.B.1: 15-55.

Singarajah, K. V. 1985. **A review of Brazilian whaling: aspects of biology, exploitation and utilization**, Cochin, India: 131-148.

Singarajah, K. V. 1997. Whales and whaling. In: K. V. Singarajah (ed.). **Fundamentals of marine biology with particular emphasis on the tropical realm**. Minerva Press, London: 276-304.

Spalding, M., Ravilious, C. & Green, E. P. 2001. **World atlas of coral reefs**. University of California Press, Berkeley. 424 p.

SUDEPE 1979. **Anuário do registro geral da pesca - 1978**. Ministério da Agricultura, Brasília: 407 p.

SUDEPE 1980. **Estatística da pesca. Produção - 1978**. Ministério da Agricultura, Brasília: 84 p.

Sumaila, U. R. & Walters, C. 2005. Intergenerational discounting: a new intuitive approach. **Ecological Economics** 52: 135-142.

Telles, M.D. 1998. Modelo trofodinâmico dos recifes em franja do Parque Nacional Marinho dos Abrolhos - Bahia. **Masters thesis**. Departamento de Oceanografia, Rio Grande, Brazil, Fundação Universidade do Rio Grande: 150 p.

Trinidad, A. C., Pomeroy, R. S., Corpuz, P. V. & Agüero, M. 1993. Bioeconomics of the Philippine small pelagics fisheries. **ICLARM Technical Reports** 38: 1-73.

Van Vliet, O. P. R., Faaij, A. P. C. & Dieperink, C. 2003. Forestry projects under the clean development mechanism? Modelling of the uncertainties in carbon mitigation and related costs of plantation forestry projects. **Climate change** 61: 123-156.

Vasconcellos, M. 2000. Ecosystem impacts of fishing forage fishes: an analysis of harvest strategies for the Brazilian sardine. **PhD thesis**, University of British Columbia, Vancouver, Canada: 188 p.

Vasconcellos, M. & Gasalla, M. A. 2001. Fisheries catches and the carrying capacity of marine ecosystems in southern Brazil. **Fisheries Research** 50: 279-295.

Vasconcelos, J. A. & Lins-Oliveira, J. E. 1996. Estudo comparativo sobre a pesca de lagostas com rede-de-espera e mergulho no estado do Rio Grande do Norte. **Boletim Técnico Científico do CEPENE** 4(1): 135-172.

Velasco, G. & Castelo, J. P. 2005. An ecotrophic model of southern Brazil continental shelf and fisheries scenarios for *Engraulis anchoita* (Pisces, Engraulididae). **Atlântica** 27(1): 59-68.

Walters, C. J. & Juanes, F. 1993. Recruitment limitation as a consequence of natural selection for use of restricted feeding habitats and predation risk taking by juvenile fishes. **Canadian Journal of Fisheries and Aquatic Science** 50: 2058-2070.

Walters, C. & Kitchell, J. F. 2001. Cultivation/ depensation effects on juvenile survival and recruitment: implications for the theory of fishing. **Canadian Journal of Fisheries and Aquatic Science** 58: 39-50.

Walters, C., Pauly, D. & Christensen, V. 1999. Ecospace: prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. **Ecosystems** 2: 539-554.

Walters, C. J., Christensen, V., Martell, S. J. & Kitchell, J. F. 2005. Possible ecosystem impacts of applying MSY policies from single-species assessment. **ICES Journal of Marine Science** 62**:** 558-568.

Welcomme, R. L., Richards, R. & Neiva, G. S. 1979. Relatório da missão sobre peixes ornamentais. **PDP Série Documentos Traduzidos** 9: 1-41.

Wiedemeyer, W. 1997. Analysis of the benthic food web of a mangrove ecosystem at northeastern Brazil. **PhD Thesis**, Kiel University, IfM, Germany: 155 p.

Zetina-Rejon, M. J., Arreguín-Sánchez, F. & Chavez, E. A. 2004. Exploration of harvesting strategies for the management of a Mexican coastal lagoon fishery. **Ecological Modelling** 172(2-4): 361-372.

Received April 2007
Accepted June 2007