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## *Aquaculture Systems*

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**Bruno Díaz López**

*The Bottlenose Dolphin Research Institute  
Golfo Aranci, Italy*

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### 9.1 Introduction

The worldwide expansion of marine aquaculture industries has caused growing concern regarding their environmental impact. Marine aquaculture is a significant industry that continues to grow more rapidly than all other animal food-producing sectors, with an average annual growth rate worldwide of 8.8% per year since 1970, compared with only 1.2% for capture fisheries and 2.8% for terrestrial farmed meat production systems (FAO 2007). This increase in marine aquaculture activities is thought to be in response to the increase in demand for fish, which cannot be fulfilled by traditional fisheries because of the decline in wild populations. Moreover, marine aquaculture is essentially an economic development within small- and medium-sized enterprises in areas where alternative employment is scarce.

In many cases, the ability to predict the effects of aquaculture on the marine environment is a prerequisite to establishing and expanding culture operations. Consequently, the study of aquaculture ecosystems requires consideration of biological, physical, chemical, and geological factors. According to FAO (1995), "The achievement of real marine ecosystem-based management of fisheries implies the regulation of the use of the living resources based on the understanding of the structure and dynamics of the ecosystem of which the resource is a part." This premise requires an improvement of our understanding of the structure of marine ecosystems, and the interactions between ecosystem compartments and their changes due to large-scale culturing operations (Díaz López et al. 2008).

Published literature shows that the development of the aquaculture industry has been accompanied by an increase in environmental impacts (reviewed by Pillay 1992; Fernandes et al. 2002; Cole 2002). The effects of aquaculture on the marine environment may be categorized into three types: eutrophication, sedimentation, and effects on the food web (Hargrave 2003). It has often been noted that the type of cultivated organisms, the locations of cultivation, the cultivated biomass, the quality and quantity of supplied food, and management practices are the main factors determining the extent of these effects (Beveridge 1996; Hargrave 2003; Pillay 2004; Machias et al. 2005).

Marine finfish aquaculture differs from that of shellfish farming in that bivalve culture requires minimal additions to the environment, except for the animals themselves and the infrastructures used to grow them. Their food is supplied by the environment and their wastes return nutrients and minerals to the ecosystem. Conversely, the marine finfish aquaculture, commonly practiced in cages, involves the supply of a substantial amount of nutrients with consequent impacts on the environment (Holby and Hall 1991; Hall et al. 1992). Fish production can also generate considerable amounts of effluent, such as waste feed and feces, medications, and pesticides, which can have undesirable impacts on the environment (Wu 1995; Lemarié et al. 1998; Read and Fernandes 2003). Consequently, it would be reasonable to expect effects at large spatial scales, particularly when a finfish farm is established in a coastal bay (Díaz López et al. 2008). In addition, effects on wild fish have been investigated at short spatial scales (Carss 1990, 1994; Dempster et al. 2004), indicating a considerable increase in wild fish abundance and biomass in the immediate vicinity of fish cages. There may also be undesirable effects on wild fish populations, such as genetic interactions between escaped farmed fish and wild fish (Youngson et al. 2001), disease transfer by escaped fish, or through ingestion of contaminated waste by wild fish (Heggberget et al. 1993). Additionally, aquaculture activities cause potential impacts on top predators such as modification of habitat use (Watson-Capps and Mann 2005; Díaz López et al. 2005), death, and injury through entanglement in gear (Díaz López and Shirai 2007).

Considerations of the sustainability of aquaculture include the ecological resources required to sustain the industry, that is, fish food for farmed species, and environmental capacity to assimilate waste (Read et al. 2001). However, to assess the sustainability, it is necessary to estimate the impact of aquaculture activities and to predict the change on the environment.

The impact that marine finfish aquaculture produces in the environment requires research involving field measurements as well as comprehensive modeling studies that integrate available knowledge about natural and anthropogenic parts of coastal ecosystems (Cruz-Escalona et al. 2007). Thus, any attempt to assess ecosystem-level effects of finfish aquaculture must consider the complexity of natural and human actions in coastal systems.

Most modeling studies of effects of finfish aquaculture on the environment have focused on organic matter, dissolved inorganic nutrients and oxygen, the impact of organic matter on the benthic system, and the relative potential for environmental effects of new and existing fish farm sites (i.e., Silvert 1992; Silvert and Sowles 1996; Hevia et al. 1996; Ervik et al. 1997; Findlay and Watling 1997; Black 2001; Cromey et al. 2002; Stigebrandt et al. 2004). Although these models are good tools to study the impact that finfish aquaculture has on the environment, ecosystem-level studies are needed on many areas, particularly the long-term responses of ecosystem components (phytoplankton, zooplankton, fish, marine top predators, benthos, as well as the farmed fish) to aquaculture-induced changes in system energy flow and biomass.

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## 9.2 Ecosystem-Level Studies: Ecopath Modeling Approach

To investigate the potential effect of finfish culture on coastal ecosystems, I advocate the use of an alternative approach based on a mass-balance modeling framework. The make use of mass-balance models in aquaculture is currently at a relatively early stage of development compared to fisheries and for many other anthropogenic activities. Thus, few studies have been completed which adequately assess these potential environmental interactions of this newly developed industry (Jiang and Gibbs 2005; Díaz López et al. 2008). Ecosystem mass-balance models are very important because the complexity of ecosystems makes it difficult to gain an insight into their structure based on direct observations (Niquil et al. 1999). The major advantage of the mass-balance model is that it can be used to study the broad spectrum of ecosystem theories, including the description of trophic levels, network analysis, information theory, and thermodynamic concepts (Müller 1997). This modeling framework can also be used to investigate the possible impacts of some species and how these may affect the ecosystem as a whole (Pauly et al. 2000).

Ecopath software (Christensen and Pauly 1992; <http://www.ecopath.org>) was developed as a useful tool incorporating algorithms for the retrieval of the ecological, thermodynamic, and informational indices needed for network analysis (Ulanowicz 1993). Through a system of linear equations describing the mass balance for each functional component of the system, the overall ecosystem balance is obtained (Christensen and Pauly 1992; Pauly et al. 1993).

Ecopath models rely on the truism that:

1. Production by (*i*) = all losses by predation on (*i*) + nonpredation losses on (*i*) + export of (*i*).

This applies for any group (e.g., a given fish population) and time (e.g., a year or season).

Groups are linked through predators consuming prey, where:

2. Consumption = production + nonassimilated food + respiration.

The basic equation that represents the balance for each trophic group, *i*, of the network is:

$$3. B_i \frac{P_i}{B_i} EE_i - \sum_{j=1}^n B_j \frac{Q_j}{B_j} DC_{ji} - EX_i = 0$$

where DC is the diet matrix, which describes the relationships among groups whose elements  $DC_{ji}$  represent the fraction of the prey *i* in the average diet of the predator *j*;  $B_i$  is the biomass of each group;  $P_i/B_i$  is the production/biomass ratio (equal to the instantaneous rate of total mortality *Z* in steady-state systems) and  $Q_j/B_j$  is the consumption/biomass ratio of predator; and  $EE_i$  is the ecotrophic efficiency, which represents the part of the total production that is consumed by predators or exported; and  $EX_i$  is the export of the compartment *i* toward other ecosystems such as net migration and harvest by fishery (Christensen and Pauly 1993). Since the currency of the model is energy-related, the unassimilated/consumption ratio ( $UN/Q$ ) is used to quantify the fraction of the food ( $Q_x$ ) that is not assimilated. More details on capabilities and limitations of the Ecopath software are given by Christensen and Walters (2000).

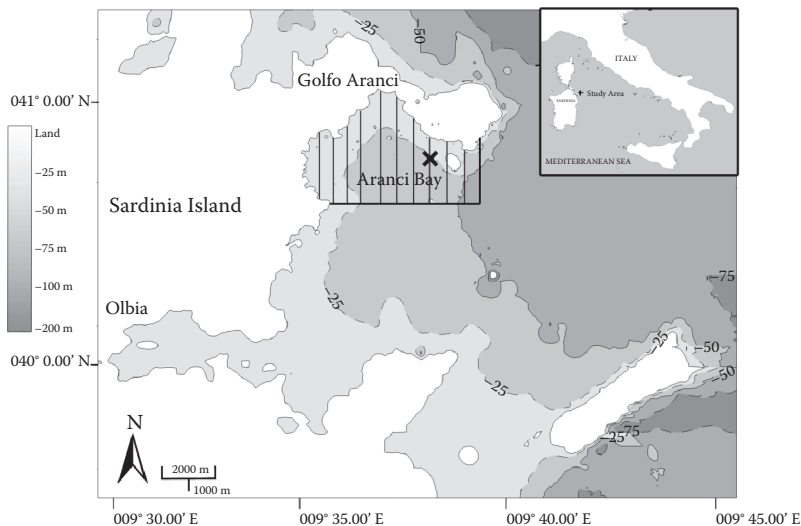
Ecopath is a steady-state model representation for a given period of the energy flows of an ecosystem, and therefore cannot be used to simulate changes to flows with time. By contrast, this modeling framework has been used to investigate the functioning of the system and how this has changed with the introduction of intensive finfish culturing (Díaz López et al. 2008) and selfish culturing (Jiang and Gibbs 2005).

### 9.3 Impact of Aquaculture on a Coastal Ecosystem Predicted by the Mass-Balance Model

The next section describes the Ecopath mass-balance model determined by Díaz López et al. (2008), refined and shaped to estimate the potential effects of finfish culture on a coastal ecosystem and, therefore, to identify the species that play a key role in the processes of ecosystems affected by aquaculture. The functioning of the Ecopath model is illustrated for two examples, and is followed by a general discussion. These examples are derived from model applications in the same area during two different periods “before” and “after” the beginning of culturing operations. The availability of published data provided the opportunity to compare these two distinct ecosystem states. Both scenarios were made in such a manner that they provide the best information on the functioning of the modeling framework.

#### 9.3.1 Defining the System: Aranci Bay

The coastal area of 16.25 km<sup>2</sup> considered in the study was the Aranci Bay, located on the northeastern coast of Sardinia (Italy) (Figure 9.1). The selection of this area was based on the environment assessment and field studies conducted before and after the establishment of a marine finfish farm (Díaz



**FIGURE 9.1**

Map of the Aranci Bay (Sardinia, Italy), showing the area influenced by aquaculture with a line pattern. A cross indicates the location of the marine finfish farm (40°59.98'N, 9°37.09'E).

López et al. 2008). This finfish farm was set up in 1995; it covers an area of 0.04 km<sup>2</sup>, which is approximately 0.25% of the Aranci Bay, and contains 850 tons of ichthyic biomass [mostly sea bass (*Dicentrarchus labrax*) and gilthead sea bream (*Sparus auratus*)].

### 9.3.2 Application of the Ecopath Mass-Balance Model

Definition of the functional trophic groups was based on similarities in their ecological and biological features (Pinnegar and Polunin 2004), based on their abundance and how they are affected by aquaculture. For each functional group, three out of four of the basic parameters [biomass, consumption ratio, production ratio, ecotrophic efficiency (EE)] were required to construct the Ecopath mass-balance models. All the available data for biomass, annual harvesting, and discards were converted into the same unit (t km<sup>-2</sup>) and expressed as wet weight. Published and unpublished sources concerning the system were used to generate input parameters; however, in some cases, it was necessary to assume from the wider literature derived for values of similar coastal systems. Furthermore, for both scenarios, the assumption was followed that the artisanal fishery effort, although low in the area, was constant.

The model was considered balanced when: (1) realistic estimates of the missing parameters of EE were calculated ( $EE < 1$ ); (2) gross efficiency values ( $GE = P/Q$ ) for functional groups were between 0.1 and 0.35 with the exception of fast growing groups with higher values and top predators with lower values; and (3) values of  $R/B$  were consistent with the group's activities with high values for small organisms and top predators (Christensen and Walters 2004).

#### 9.3.2.1 First Mass-Balance Scenario (Scenario A): State of the System Influenced by Aquaculture Activities

The first scenario includes 14 living and 3 detritus groups spanning the main trophic components of a coastal ecosystem influenced by finfish aquaculture (Table 9.1). In order to consider fish farm effects, the fish nourishment, harvesting, and discards, as well as outflow from the group representing farmed fish species, were introduced into the model. In addition, the mortality of some groups affected by incidental captures in the fish farm was increased according to information provided by field studies (Díaz López 2006; Díaz López and Shirai 2007), and in function of the data reported by the fish farm manager. This scenario provides a summary of current knowledge of the biomass, consumption, production, food web, and trophic structure in this area after the establishment of a marine finfish farm. Similarly, diet data for the aggregated functional trophic groups were obtained from the diet compositions reported in the model of Díaz López et al. (2008) and data gathered from published sources.

TABLE 9.1

Input Values (in *Italic*) and Estimates of Some Parameters in the Ecopath Mass Balance Model of the System after the Establishment of Marine Finfish Farm (Scenario A)

Group Name	TL	<i>B</i> (t km <sup>-2</sup> )	<i>P/B</i> (/year)	<i>Q/B</i> (/year)	EE	Flow to Detritus (t km <sup>-2</sup> year <sup>-1</sup> )
Bottlenose dolphins	3.82	6.246	0.331	3.766	0	6.772
Cormorants	3.55	0.554	0.315	4.225	0	0.643
Seabirds	3.17	0.583	0.251	1.19	0	0.285
Cephalopods	3.17	8.889	2.34	5.3	0.961	10.227
Mugil cephalus	2.54	13.549	0.624	8.587	0.95	58.599
<i>Mugil cephalus</i> (juveniles)	2.54	1.866	1.74	23.45	0.781	22.585
Piscivorous fish	3.74	27.227	0.729	2.88	0.908	17.512
Zooplanktivorous fish	2.57	31.479	1.5	8.86	0.966	57.38
Farmed fish	2	52.308	1.138	2.4	0.656	0
Polychaetes	2	5.79	4.8	11.53	0.95	28.093
Macrobenthos detritivorous	2	8.325	5.23	18	0.837	0
Mussels	2.03	3.614	1.8	6.629	0.019	11.174
Zooplankton	2.09	6.476	50	170	0.97	560.263
Phytoplankton	1	6.57	112.65	–	0.95	37.006
Fish farm discards	1	1.231	–	–	0	0
Nourishment	1	156.923	–	–	0	0
Detritus	1	631.73	–	–	0.823	0

Note: Trophic level (TL), biomasses (*B*), production rates (*P/B*), consumption rates (*Q/B*), ecotrophic efficiency (EE), and flow to detritus used in the mass-balance model.

### 9.3.2.2 Second Mass-Balance Scenario (Scenario B): State of the System Before the Start of Aquaculture Activities

This mass-balance model includes 11 living and 1 detritus groups spanning the main trophic components of a Mediterranean coastal ecosystem (Table 9.2). Biomasses were estimated for most trophic groups based on previous field studies, and in function of the data reported in coastal areas of similar characteristics to Aranci Bay before the aquaculture operations began. Information gathered from published sources was useful in rejecting all groups directly related with the presence of aquaculture (i.e., cormorants, farmed fish, mussels). For example, species such as farmed fish and mussels were absent in the area before the beginning of aquaculture operations. Similarly, cormorants were not present in the area because their presence was directly related with predation in the finfish farm. Likewise, diet matrix was

TABLE 9.2

Input Values (in *Italic*) and Estimates of Some Parameters in the Ecopath Mass Balance Model of the System before the Establishment of the Marine Finfish Farm (Scenario B)

Group Name	TL	<i>B</i> (t km <sup>-2</sup> )	<i>P/B</i> (/year)	<i>Q/B</i> (/year)	EE	Flow to Detritus (t km <sup>-2</sup> year <sup>-1</sup> )
Bottlenose dolphins	4.04	1.679	0.875	14	0	6.172
Seabirds	3.75	0.012	0.251	1.19	0	0.006
Cephalopods	3.4	4.875	2.34	5.1	0.903	6.08
Mugil cephalus	2.54	2.887	2.824	8.087	0.967	4.939
<i>Mugil cephalus</i> (juvenile)	2.54	1.186	1.74	23.45	0.473	0
Piscivorous fish	4	13.02	1.56	2.88	0.902	9.496
Zooplanktivorous fish	2.72	14.95	1.5	8.86	0.938	27.885
Macrobenthos detritivorous	2	1.928	5.23	18	0.837	0
Polychaetes	2	4.90	2.67	13.36	0.909	27.403
Zooplankton	2.09	3.5	50	170	0.969	306.477
Phytoplankton	1	3.6	112.65	–	0.95	20.238
Detritus	1	356	–	–	0.824	0

Note: Trophic level (TL), biomasses (*B*), production rates (*P/B*), consumption rates (*Q/B*), ecotrophic efficiency (EE), and flow to detritus used in the mass-balance model.

obtained from the diet compositions reported in the model of Díaz López et al. (2008) and data gathered from published sources.

Where biomass estimates for this scenario were not available, the EE estimated by Ecopath for the first scenario was used as input parameter in the second scenario. In this way, Ecopath estimated biomasses under the assumption that the fraction of production used within the system is the same in the first and the second scenarios. Furthermore, although the first scenario was inherently more accurate than the second scenario, the latter represented useful estimates of the possible community states before aquaculture activities.

### 9.3.3 Predicted Effects of the Introduction of Finfish Culture Determined from the Ecopath Mass-Balance Model

To predict the effects of finfish aquaculture on the system, the status of the two ecosystem scenarios was compared and the system response investigated. The model as presently formulated makes it possible to predict increases in biomass groups as a result of nutrient loadings in agreement with field studies.



The structure of the ecosystem in both scenarios showed substantial differences in biomass values estimated for each group (Table 9.3). This change in all trophic groups demonstrates an increase in the biomass after the start of aquaculture activities. Biomass values were estimated by the model for zooplankton, polychaetes, macrobenthos detritivorous (amphipods and isopods), zooplanktivorous fish, piscivorous fish, cephalopods, and common gray mullets (adults and juveniles). Augment in biomass seen in benthic (i.e., polychaetes and macrobenthos detritivorous) and pelagic subsystems (i.e., fish species such as common gray mullets and zooplanktivorous) was in accordance with field studies (Dempster et al. 2004; Klaoudatos et al. 2006). Increased nutrient loading into the fish farm area resulted in greater biological activity and induced a strong coupling between the pelagic and benthic subsystems (Díaz López et al. 2008).

The calculation of “Trophic aggregations” (Ulanowicz 1995) with Ecopath, provided an accurate picture of the system and allocated the different dietary interactions to discrete trophic levels (Figures 9.2 and 9.3). This increase in biomass after the start of the aquaculture was nonmonotonic, although an increase in primary producer biomass should propagate monotonically through all trophic levels in a system based almost entirely on primary producers (Odum 1971). In cases like this (of strong coupling between the pelagic and benthic subsystems), the bottom-up control of phytoplankton

**TABLE 9.3**Estimated Biomass (t km<sup>-2</sup>) of Trophic Groups for the Created Scenarios

Group	Scenario B	Scenario A	Variation (%)
Bottlenose dolphin	1.679	6.246	272
Cormorant	0	0.554	
Seabirds	0.012	0.583	4 758
Cephalopods	4.875	8.889	82.3
Mugil cephalus	2.887	13.549	369
<i>Mugil cephalus</i> (Juveniles)	1.186	1.866	57.3
Piscivorous fish	13.023	27.227	109
Zooplanktivorous fish	14.95	31.479	110
Farmed fish	0	52.308	
Polychaetes	4.905	5.79	18
Macrobenthos detritivorous	1.928	8.325	331
Mussels	0	3.614	
Zooplankton	3.541	6.476	82.9
Phytoplankton	3.593	6.57	82.8
Discards	0	1.231	
Nourishment	0	156.923	
Detritus	356.015	631.73	77.4
Total biomass	408.594	963.36	135.7

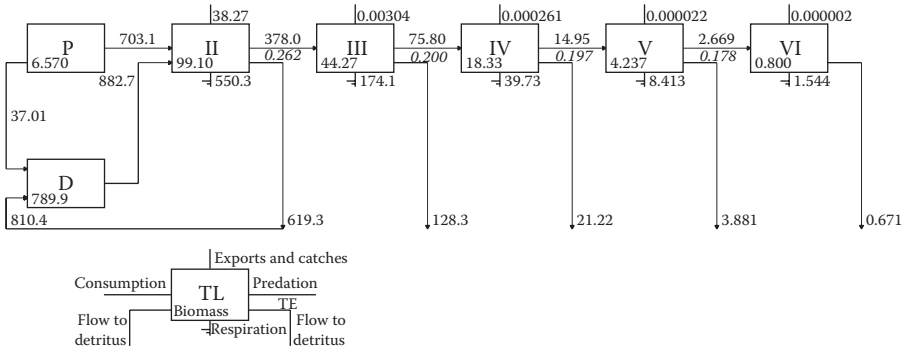


FIGURE 9.2

The complex food web for the first scenario (after the establishment of the marine finfish farm) is represented schematically in the form of the Lindeman spine, where biomasses and flows are aggregated into integer trophic levels (TL). At the first level of the chain, the primary producers (P) and detritus (D) are split for clarity.

development (i.e., nutrient loadings from fish farms) becomes less important, and the ecosystem could be more resilient to changes in external nutrient loading (Prins et al. 1998). This modeling framework confirms the important role that is played by detritus groups, and in particular those related with aquaculture (fish farm nourishment and discarded fish) in this trophic network (Díaz López et al. 2008).

Although the presence of aquaculture added two detritus groups to the Aranci Bay (nourishment and discards), the biomass stored in detritus groups in relation with the total biomass was lower after the start of

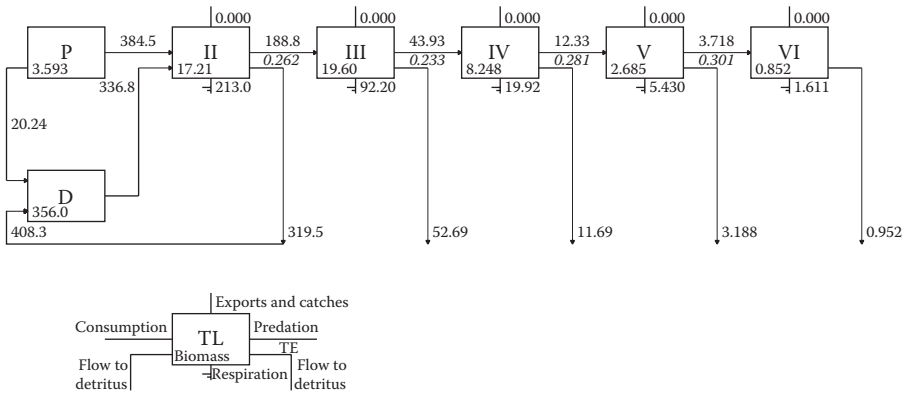


FIGURE 9.3

The complex food web for the second scenario (before the establishment of the marine finfish farm) is represented schematically in the form of the Lindeman spine, where biomasses and flows are aggregated into integer trophic levels (TL). At the first level of the chain, the primary producers (P) and detritus (D) are split for clarity.

**TABLE 9.4**

Summary of the Cycling Indices Estimated for the Created Scenarios

	Unit	Scenario B	Scenario A	Variance (%)
Throughput cycled (excluding detritus)	t km <sup>-2</sup> year <sup>-1</sup>	63.61	107.78	69.43
Predatory cycling index	% of throughput without detritus	6.57	5.42	-17.50
Throughput cycled (including detritus)	t km <sup>-2</sup> year <sup>-1</sup>	438.15	800.07	82.60
Finn's cycling index	% of total throughput	24.54	20.93	-14.71
Finn's mean path length	none	4.411	4.008	-9.13

aquaculture activities. This could be explained by the high concentration of species described as buffers to the eutrophication process (i.e., common gray mullets, mussels) feeding around sea cages, which may diminish the amount of organic matter that reaches the sea floor (Porter et al. 1996; Dempster et al. 2004; Lupatsch et al. 2003; Nizzoli et al. 2005; Mazzola and Sara 2001). These results support the notion that the contextual cultivation of species from different trophic levels (e.g., bivalves together with carnivorous fish) would reduce the impact that would emerge from the cultivation of only carnivorous species (Troell et al. 1999). Díaz López et al. (2008) also suggested that the role that top predators (marine mammals and seabirds) could play in the elimination of discarded fish (reducing the organic matter) implies that these species may be considered as buffers to the eutrophication process, reducing the organic matter present in the area.

The capacity of an ecosystem to entrap, withhold, and cycle nutrients increases with system "maturity" (Odum 1969), and this "maturity" has been correlated with Finn's cycling index (FCI) (Christensen 1995). The FCI, estimated through Ecopath for both trophic networks, were relatively high (Table 9.4), indicating a substantial degree of recycling before and after the start of aquaculture activities. Furthermore, the higher FCI estimated for the second scenario (before the establishment of aquaculture) confirms that recycling played an important role in the maintenance of coastal ecosystem stability. In other words, the establishment of aquaculture could reduce the recycling, and the lower the recycling level, the more slowly will the effects of perturbations be eliminated from the system.

## 9.4 Summary

Long-term growth of the aquaculture industry requires both ecologically sound practices and sustainable resource management. Thus, any decision

related to aquaculture and environmental systems will necessarily involve management issues. In the face of the scale of aquaculture impact on coastal ecosystems, scientists have to find ways and tools to predict these effects and account for ecological interactions, especially those of a trophic nature.

The Ecopath mass-balance model provides a rigorous and relatively simple framework that is capable of accounting for the major components and trophic interactions of the ecosystem, and produce results coherent with field studies and ecological theory on ecosystem development properties. The Ecopath mass-balance model can be a valuable tool for understanding ecosystem functioning, and for designing ecosystem-scale adaptive management experiments. This modeling framework could provide new insights into the understanding of how aquaculture influences coastal ecosystems, and hence support the design of policies aimed at implementing ecosystem management principles.

The case study presented in this chapter has shown the appropriateness of the model in describing the modifications induced, at an ecosystem level, by nutrient loading into the area. Increased nutrient loading is the most obvious predicted effect from fish farms, and measures of this effect comprise the main method of regulating and controlling the size of fish farms such that the local environment is not overwhelmed. It must be highlighted that the biomass estimates from the mass balance models could serve as a guide to investigate changes in the environment attributed to the start, recovery, or cessation of farming. In addition, these estimates could provide important additional information complementary to the normal environmental assessment impact studies, and before selecting polyculture as a potential solution to some aspects of eutrophication. Finally, mass-balance models could be a useful tool, from the viewpoint of predicting how increased nutrients will affect the plankton community, particularly if there is a risk of toxic algal blooms in the area.

The accuracy of parameter estimates depends on information available to the groups. Much more refined models than this can be developed if sufficient biological data are available. Bias resulting from estimating the model parameters would have an impact on the model output. The main limitation to using this type of model is the availability of field data and the time and resources needed to obtain this information. In particular, the limited availability of parameter estimates on a seasonal or annual basis for some groups (particularly multistanza groups) reflects a need for future models. Some groups may include hundreds of species, and it is almost impossible to have an accurate parameter estimate for such groups. Data needed to estimate the model parameters (including dietary composition) are often incomplete, and estimates have to be based on studies conducted elsewhere for similar species, or rely on qualitative descriptions. These would certainly affect the model output.

Apart from above-mentioned factors, the prediction of aquaculture's effects on the ecosystem is likely to be influenced by other factors not

explicitly addressed in this model. For example, climatological variability, hydrographic conditions, bottom topography, and geography may also lead to variability in the input of new nutrients that may also ultimately influence the environmental impact of aquaculture.

Using Ecosim and Ecospace routines, in a future step, it could be possible to simulate the consequences of certain management measures, such as changes in aquaculture, on the ecosystem. Nevertheless, further research is required in order to improve input data and to support or refute the results presented in this model. Although there remains a great deal of research and development work yet to be done to improve our understanding of the environmental impacts of fish farming in marine waters, Ecopath models can help us assess these impacts and make reasonable management decisions.

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