Ecosystem-Based Fishery Management:
A Critical Review of Concepts and Ecological Economic Models

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Abstract

An ecosystem approach means different things to different people. As a result the concept of ecosystem-based fishery management is evolving and it has no universal definition or consistent application. As regards ecosystem modeling, most economic models of fishery ignore the linkages to lower trophic levels. In particular, environmental data and other bottom-up information is widely disregarded. The objective of this paper is to provide a critical review of concepts and ecological economic models relating to ecosystem-based fishery management especially to the environmental issues. The paper started by reviewing basic concepts related to ecosystem-based fishery management and the economic value of ecosystems. Then it aimed to review the development of ecosystem modeling with emphasis on economic aspects of ecology. This was followed by a presentation of the applications of ecosystem-based fishery management in practice. The paper concluded with some critical discussions and brought together conclusions derived from previous literature reviews. We found that although the concept of ecosystem-based fishery management has no universal definition, there is a widespread agreement about the need to implement the ecosystem approach for fisheries in practice. We also revealed that nutrient flow plays a crucial role in an ecosystem. In addition, it has many properties such as recycling and exchanging between consumers and producers, which are similar to monetary flows in an economy. Therefore, nutrients should be chosen as the currency in ecological economic models.

Keywords: Ecosystem approach, fisheries management, ecological economic models

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1. Introduction

Fisheries management to date has often been ineffective since many marine fisheries are suffering from a combination of (recruitment and growth) overfishing of fish stocks[1] and overcapacity of fishing fleets (Clark 2006, pp. 1). In 2005, the Food and Agriculture Organization estimated that some 77 % of the world's fish stocks were either fully exploited, overfished or depleted[2] (FAO 2007, pp. 29). The global fishing fleet was estimated to be more than two and a half the size that the oceans can sustainably support (Porter 1998, pp. 11). In addition, the ocean's productivity has also been declining because of marine environment degradation and interference with the ecosystems through pollution (Crean and Symes 1996, pp. 4).

The collapses[3] of many fisheries is widely believed the result of a mismanagement (Costello, Gaines et al. 2008, pp. 1678). The mismanagement of fisheries is not only because of poor enforcement, but also because fisheries management traditionally focuses on managing a single target species and often ignores habitat, predators, and the prey of the target species and other physical components of ecosystems (Pikitch, Santora et al. 2004, pp. 346). The conventional single species fisheries management has failed and new approaches are needed (Beverton 1995, pp. 229-245; Hilborn 2004, pp. 275-276; Beddington, Agnew et al. 2007, pp. 1713-1714; Cardinale and Svedang 2008, pp. 244). A major element of the proposed new approaches is a move from conventional single species management to ecosystem-based fisheries management, which seeks to

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1 *Recruitment overfishing* means that the adult population was fished so heavily that the number and size of the adult population (spawning biomass) was reduced to the point that it did not have the reproductive capacity to replenish itself. *Growth overfishing* occurs when animals are harvested at an average size that is smaller than the size that would produce the maximum yield per recruit.

2 If the biomass of a fish stock falls below *Minimum Stock Size Threshold* (MSST), a threshold used by fishery managers e.g. 30-40% of spawning biomass, a stock is determined to be overfished, depleted or collapsed. Fish stock is considered fully exploited when the catch is reached the Maximum Sustainable Yield (MSY).

3 See 2.

include in the management plan not only all affected species but also abiotic factors such as water pollution, the effects of weather and climate on the ecosystem, and the effects of fishing activity on the habitat itself (Fluharty, Aparicio et al. 1998, pp. 1-10; Hilborn 2004, pp. 275).

An ecosystem approach means different things to different people (O'neill, DeAngelis et al. 1986; Larkin 1996). Grumbine (1994) summarized the ten dominant themes of ecosystem management and Arkema (2006) reviewed 17 criteria that scientists used to describe an ecosystem-based approach. Although there are many themes and definitions, the concept of ecosystem-based fishery management is still unclear and there is no agreed standard approach (Brodziak and Link 2002; Ward, Tarte et al. 2002; Babcock and Pikitch 2004). Some authors such as Larkin (1996), Brodziak (2002), Link (2002), Sumaila (2005) and Marasco (2007) have reviewed concepts related to ecosystem-based fishery management. There are also some authors reviewing ecosystem models but their papers concentrate either on habitat modeling (Knowler 2002; Amstrong 2006) or on ecological modeling in general (Larkin 1996). This paper will start by reviewing basic concepts related to ecosystem-based fishery management and the economic value of ecosystems. Then it will aim to review the development of ecosystem modeling with emphasis on economic aspects of ecology. This will be followed by a presentation of the applications of ecosystem-based fishery management in practice. The paper will conclude with some critical discussions and will bring together conclusions derived from previous literature reviews.

2. Ecosystem and Ecosystem-Based Fishery Management

In ecology, there are two ways to understand ecosystems (O'neill, DeAngelis et al. 1986; Bocking 1994). The *Population- community ecologists* tend to view ecosystems as networks of interacting populations. The biota are ecosystems, and abiotic components such as soil or sediments are external influences. The biota may interact with the abiotic environment, but the environment is largely viewed as the backdrop or context within which biotic interaction occur. The *population-community* approach is partly a result of the historical development of ecology and it is an appropriate conceptualization for some observation sets, rather than the best or most fundamental way to view ecosystems (O'neill, DeAngelis et al. 1986).

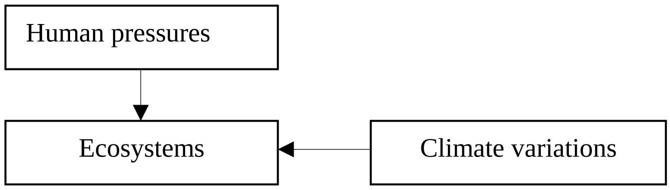
Most ecologists tend to view ecosystems by using *the Process-Functional Approach* (O'neill, DeAngelis et al. 1986; Bocking 1994). The ecosystem concept, in this approach, was originally defined by Tansley (Tansley 1935; Bocking 1994). He defines ecosystem as **"…***the whole system (in the sense of physics), including not only the organism-complex, but also the whole complex of physical factors forming what we call the environment of biome – the habitat factors in the widest sense "(Tansley 1935, pp. 299).* This ecosystem concept was further developed and clarified by Linderman (1942), Hutchinson (1948), H. T. Odum (1951) and E. P Odum (1969). In terms of energy and material flows, E. P. Odum (1969) interpreted: *"the ecosystem, or ecological system, is considered to be a unit of biological organization made up of all of the organisms in a given area (that is, community) interacting with the physical environment so that a flow of energy leads to characteristic trophic structure and material cycle within the system".* Within the ecosystem, energy and nutrients are exchanged, consumed and transformed, and feedback loops ensure that, within limits, the system will remain at equilibrium (Bocking 1994). *The process-functional approach* has limited applications in cases that deal with the effect of single populations (dominant or key species) on ecosystem function (O'neill, DeAngelis et al. 1986).

There are some important differences among ecosystems, even though they have some fundamental similarities. For those that are land based, the regional ecosystems are defined by major vegetation characteristics but this is not true for the world's oceans (Larkin 1996). For example, Large Marine Ecosystems (LMEs), which are regions of ocean space surrounding coastal areas from river basins and estuaries out to the seaward boundary of continental shelves and the outer margins of coastal current systems, are defined by distinct characteristics of depth, oceanography and productivity (Sherman, Alexander et al. 1993, pp.3; Sherman and Duda 1999).

The ecosystem concept is open with regard to spatial scale so that it allows the researchers to select the scale that is appropriate to a particular objective or interest. In one extreme, an ecosystem can be defined as an explicit unit of space such as a small pond occupied by a group of plants, animals, and microbes interacting with each other and their environment. In the other extreme, an ecosystem can occupy a coastal area of the order 200,000 km^2 or larger such as LMEs (Sherman, Alexander et al. 1993, pp.3). The difficulties often arise in attempting to measure transfers of materials, energy, and organisms into and out of (across the boundaries). Hence, scientists often choose ecosystem with well-defined physical boundaries (Franklin 1997).

There is growing evidence, which has led to recognition that coastal ecosystems are being negatively impacted by multiple driving forces or external factors (Sherman and Duda 1999; Stenseth, Ottersen et al. 2004). These external factors can be divided into two groups, namely, human pressures and climate variations (figure 1). Human pressures include overfishing, eutrophication, toxic pollution and habitat degradation (Sherman and Duda 1999). While climate variations include fluctuations of temperature, win, and residual currents as well as interactions among these (Stenseth, Ottersen et al. 2004, pp.3). Coral reef bleaching is a typical example of the impacts of climate variations (global warming) on coral reef ecosystems.

Figure 1. External factors impact on ecosystems



Source: Adapted from (Sherman and Duda 1999; Stenseth, Ottersen et al. 2004).

Ecosystem management is a framework officially adopted in US since early 1990s (Grumbine 1997). Fundamentally, ecosystem management is managing ecosystems so as to assure their sustainability (Franklin 1997). Ecosystem management is a response to today's deepening biodiversity crisis and it is still developing, at least within the academic literature (Grumbine 1994; Arkema, Abramson et al. 2006). Grumbine (1994) summarized the ten dominant themes of ecosystem management: hierarchical, ecological boundaries, ecological integrity, data collection, monitoring, adaptive management, interagency cooperation, organizational change, humans embedded in nature and values. These ten dominant themes form the basis of working definition: "*Ecosystem management integrates scientific knowledge of ecological relationships within a complex sociopolitical and values framework toward the general goal of protecting native ecosystem integrity over the long term"* (Grumbine 1994, pp. 31). We know that ecosystem perspective is desirable, but it is complex and unpredictable (Fluharty, Aparicio et al. 1998). In view of the fact that it is impossible to measure the dynamics of every species and ecosystem process or in other words, ecosystems cannot be controlled, it is scientifically more accurate to speak of "*ecosystem-based management*" or "*ecosystem approach to management*" (Christensen, Bartuska et al. 1996; Link 2002; McLeod, Lubchenco et al. 2005). Ecosystem-based management does not require that we understand all things about ecosystems since it focuses on managing human activities, rather than managing entire ecosystem (Fluharty, Aparicio et al. 1998; McLeod, Lubchenco et al. 2005).

Ecosystem-based fishery management was defined as *"a holistic approach to maintaining ecosystem quality and sustaining associated benefits"* (Fluharty, Aparicio et al. 1998). The term ecosystem management is clearly relevant for fisheries systems, however, the concept of Ecosystem-based fishery management is evolving and it has no universal definition or consistent application (Brodziak and Link 2002). Arkema (2006) reviewed definitions of marine ecosystem-based management (including ecosystem-based fishery management) and he found that there were 17 criteria that scientists used to describe an ecosystem-based approach. These criteria were divided into three categories: ecological, human dimension, and management. Ecological criteria focus on structure, function of ecosystem and recognize that ecological processes occur on temporal and spatial scales. While human dimension integrate economic factors and stakeholders into ecosystem planning processes. Management criteria include co-management and the precaution approach, as well as the use of science and technology (Arkema, Abramson et al. 2006). *Ecosystem-Based Management* (EBM) and *Ecosystem-based fishery management* (EBFM) are different, but complementary. EBM is viewed in a broader context and applied for managing cross-sectors while EBFM is applied for managing individual fishing sector (McLeod, Lubchenco et al. 2005).

The ecosystem-based approach is applied later in fisheries management comparing it to the other sectors such as land or forestry management (Grumbine 1994; Garcia, Zerbi et al. 2003; Arkema, Abramson et al. 2006). However, the ecosystem-based approach to fisheries seems to be an ambitious approach by defining humans as one of the species of ecosystems. Human populations are considered as other species populations, which have interactions with each other and their environment in ecosystems (Garcia, Zerbi et al. 2003, pp. 7). The ecosystem-based approach to fisheries, which was defined *"...to balance diverse societal objectives, by taking into account the knowledge and uncertainties about biotic, abiotic and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries"* (FAO 2005, pp. 14). In fact, the ecosystem-based approach to fisheries aims to implement sustainable development in a fisheries context (FAO 2005, pp. 6).

In this section, basic concepts related to ecosystem-based fishery management were reviewed. In order to understand ecosystem-based fishery management from the economic point of view, basic concepts related to the economic value of the ecosystems will be explored in the next section.

3. The Economic Value of Ecosystems

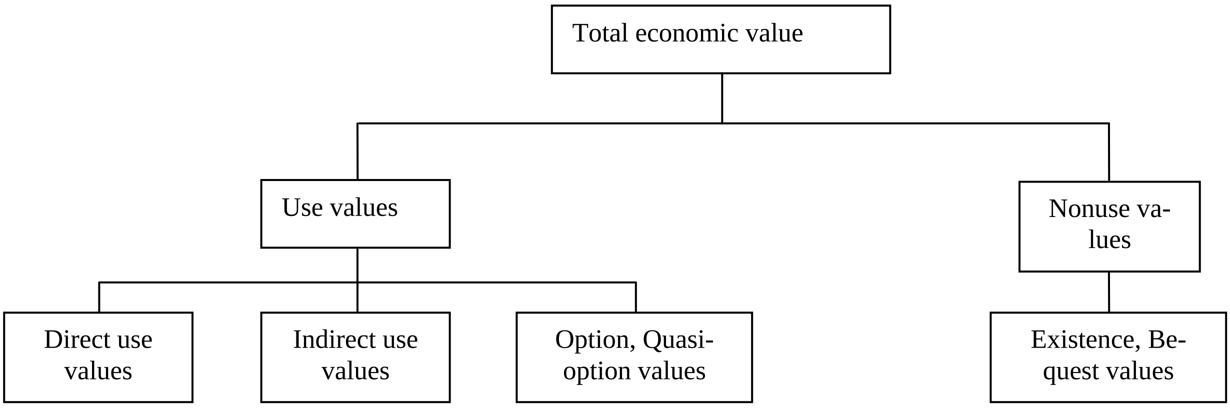
For conventional goods and services, markets provide important information about values. However, environmental amenities including ecosystems' goods and services are often not directly purchased and sold in markets. Hence theoretical research on ecosystem valuation has focused on non-market valuation, which is tightly linked to the theory of valuation of price changes. The theory for price changes is then extended to environmental quality changes (Bishop and Woodward 1995).

The total economic value of an ecosystem as an asset is the sum of the discounted present values of the flows of all services (FreemanIII 2003, pp. 5). The services of an aquatic ecosystem may include food (fish), freshwater, recreation, nutrient cycling, novel products and so on (Heal, Barbier et al. 2004; Pagiola, Ritter et al. 2004, pp. 6). The ecosystem services are divided into use values and nonuse values. The use values are further divided into direct values, indirect values, option and quasi-option values (Barbier 1994; FreemanIII 2003, pp. 12-14).

The direct use values refer to ecosystem goods and services that are directly used by human beings such as harvesting fish or recreational activities. The indirect use values are derived from ecosystem services that provide benefits outside the ecosystem its self (Pagiola, Ritter et al. 2004, pp. 10). For example, around 80 million people in the entire catchment area of the Baltic Sea may benefit indirectly from the services provided by aquatic ecosystems of the Baltic Sea such as air quality and climate stabilization. Option value is amount that consumers will be willing to pay for the option to consume ecosystem goods and services in the future (Weisbrod 1964). The concept of option value can be shown to equal the expected value of perfected information, while quasi-option value is essentially the expected value of the information gained by delaying an irreversible decision to develop a natural area (Conrad 1980; FreemanIII 1984). Option value and quasi-option value are pure public goods that the market will fail to account for, leading to a less than optimal allocation of resources (Long 1967).

Nonuse values refer to the enjoyment people experience simply by knowing that a resource exists even if they never expect to use that resource directly themselves. Illustratively, the non-use values refer to value placed on the preservation of species in the future "for reasons peculiarly our own" (Mann and Plummer 1995; as cited in Brown 2000). This kind of value is usually known as existence value or bequest value (Pagiola, Ritter et al. 2004, pp. 10). Bequest value is the value of satisfaction from preserving a natural environment for future generations (Greenley, Walsh et al. 1981). It is existence value when we may expect that our friends and relatives as well as others will have an opportunity to experience these species (Krutilla 1967; as cited in Brown 2000). Figure 2 shows the components of the total economic value of an ecosystem.

Figure 2. Total economic value of an ecosystem



Source: Adapted from Barbier (1994).

Freeman III (2003, pp. 5) argued that the economic value of a natural asset may be quite different from its market value because many of these service flows are not bought or sold in the market and therefore do not have market prices. If ecosystem services' changes make individuals "worse off", then one would like to have some measure of loss of economic values to these individuals. Otherwise, if the changes make people "better off", one would like to estimate the resulting value gain (Heal, Barbier et al. 2004, pp. 95).

There are two approaches to valuing changes in environmental goods, namely, revealed preference and stated preference (FreemanIII 2003, pp. 23-24; Maler and Vincent 2005, pp. 519-520). *Revealed preference* methods are based on actual behavior reflecting utility maximization subject to constraints (FreemanIII 2003, pp. 24). Measurement models in this approach are either based on observations of changes in market prices and quantities that resulting from changes in environmental quality or based on observation of altering purchases of goods and services that complements or substitutes for environmental quality in preference orderings of individuals (Maler and Vincent 2005, pp. 520-566). *Stated preference* methods are based on people's responses to hypothetical questions rather than from observations of real-world choices (FreemanIII 2003, pp. 24).

There are two ways to estimate values in stated preference methods (Heal, Barbier et al. 2004, pp. 119). Contingent valuation, which was developed by economists, is used to estimate values for applications, such as aquatic ecosystem services, where neither explicit nor implicit market prices exist. While conjoint analysis was developed in the marketing literature to estimate prices for new product or modifications of existing products. Contingent valuation is the commonly used approach, while the use of conjoint analysis is relatively new for nonmarket valuation and very few conjoint studies of aquatic ecosystems services have been undertaken (Heal, Barbier et al. 2004, pp. 119-123).

In general, environmental or resource quality can affect an individual's utility in three ways (FreemanIII 2003, pp. 96):

1. As an input in the household production of utility-yielding commodities;

2. Producing utility directly by being an argument in an individual's utility function;

3. Producing utility indirectly as a factor input in the production of a marketed good that yields utility.

Household production function approaches involve modeling consumer behavior, based on the assumption of a substitution or complement between an ecosystem service and one or more marketed commodities. There are three types of household production models, which have been applied to aquatic ecosystems (Heal, Barbier et al. 2004, pp. 101-113): (1) random utility or travel cost models, which are normally applied for valuing recreational fishing in freshwater lakes, rivers and marine waters; (2) averting behavior models, which analyze the rate of substitution between changes in behavior and expenditures on changes in environmental quality in order to infer the value of certain nonmarketed environmental attributes; (3) hedonic models, which analyze how the different characteristics of a marketed good, including environmental quality, might affect the price people pay for the good.

Assume that q denote some parameter of environmental or resource quality, q can produce utility directly by being an argument in an individual's utility function. In this case, q can interact with one or more market goods in the individual's preference structure in many ways. For example, there may be a substitution or complementary relationship between q and some private good (FreemanIII 2003, pp. 96).

Environmental or resource quality can also produce utility indirectly as a factor input in the production of a marketed good that yields utility. Changes in q lead to changes in production costs, which in turn affect the price and quantity of output or the returns to other factor inputs, or both (FreemanIII 2003, pp. 96-97). Assume that good x is produced with a production function

$x = x\ (k, w, \ldots, q)$ (where k and w are capital and labor, respectively)

With given prices, and assuming cost-minimizing behavior, there is a cost function

$C = C\ (p_w, p_k, x, q)$

Because q affects the production and supply of a marketed good, the benefits of changes in q can be defined and measured in terms of changes in market variables related to the x industry. A change in q will cause shifts in both cost curves and factor demand curves (FreemanIII 2003, pp. 97). The production function approach has the advantage of capturing the ecosystem functioning and dynamics of key services and can be used to value multiple services arising from aquatic ecosystem (Heal, Barbier et al. 2004, pp. 117). However, the production function approach and other revealed preference methods are not suitable for valuing nonuse values; instead, we must rely on stated preference methods (FreemanIII 2003, pp. 134).

The protection of sensitive ecosystems presumably increases the well-being of many members of society, but they generally also impose costs which translate into reductions in well-being for other members of society (Maler and Vincent 2005). For example, the establishment of marine protected areas (MPAs), which may benefit local communities by increasing the value of the ecosystems in the MPAs in the long run, however, it may directly affect livelihoods of the fishing communities in the short run.

In economic perspectives, Sumaila (2005) argue that many LMEs are shared by two or more countries. As a result, management of LMEs can be influenced by the way countries weight market and non-market values and the discount rate is applied to flows of net benefits over time from the ecosystems. Differences in discount rates and difference emphasis on market and non-market values among countries sharing the same marine ecosystem will lead to problems in implementing ecosystem-based management (Sumaila 2005).

In this section, basic concepts related to the economic value of ecosystems were reviewed. In the next section, a review of the economic models of an ecosystem will be presented.

4. Ecological Economic Models

Dynamic quantitative modeling in ecology began early in twentieth century in the form of mathematical population theory and was expanded in midcentury by the addition of systems analysis and ecosystem modeling. Population modeling peaked in the 1920s and 1950s while system analysis and ecosystem modeling peaked in the 1970s (Lauenroth, Burke et al. 2003, pp. 33).

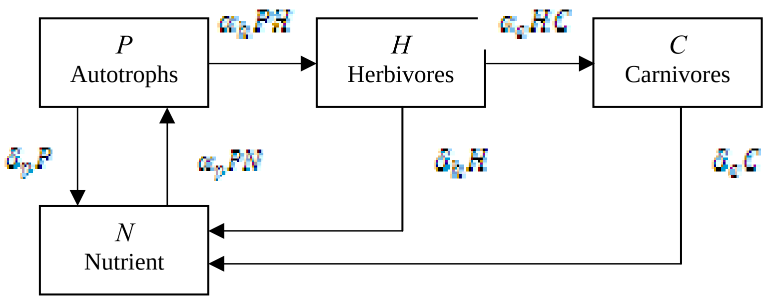
Population modeling was originally introduced by Verhulst (1838) and Pearl (1920).While the system analysis was initially introduced by Lotka (1925) and Volterra (1926) in the form of natural predator-prey model (Billard 1977; Beryman 1992; Renshaw 1993; Eichner and Pethig 2006). The Lotka-Volterra model has been applied and modified by numerous authors such as May (1979), Flaaten (1988; 1990; 1998), Yodriz (1994). The Lotka-Volterra model also has been generalized to n-species community or food web models (Polovina 1984; Tu and Wiliman 1992; Christensen, Walters et al. 2004; Pastor 2008). Regarding ecological economic aspects, there are two approaches, which have been used for population modeling, namely, *macro* and *micro* approaches (Pethig and Tschirhart 2002; Eichner and Pethig 2006). The *macro approach* takes populations as basic units of analysis. Species are presented by (differential) equations containing as variables their own populations and the populations of other species such as preys, predators and so on (May, Beddington et al. 1979; Flaaten 1988; Flaaten 1990). The limitations of choosing populations as basic endogenous variables is that it ignores the transactions of individual organisms, and does not answer the question how the interaction of individual organisms translates into population changes (Eichner and Pethig 2006). The *micro approach* takes individual organisms as basic units of analysis. The representative organisms are assumed to behave as if they maximize their net energy or biomass as price takers subject to appropriate constraints (Tschirhart 2000; Pethig and Tschirhart 2002; Eichner and Pethig 2006; Ravn-Jonsen 2009). The organisms behave as consumers who face a budget constraint requiring their expenditure on prey biomass not to exceed their revenue from supplying own biomass (Eichner and Pethig 2006). The micro approach solves the limitations of the macro approach.

Both in macro and micro approaches and in all population models, there is always one parameter, the carrying capacity of the species that forms the base of the food chain or food web. It is believed that the carrying capacity is certain to change with the environment and the abundance of predators, parasites and competitors (Hart and Reynolds 2002, pp. 130). But, the carrying capacity parameter in the population models is just a result of a particular assumption about density dependence and has nothing explicitly to do with the environment (Pastor 2008, pp.129). The complicated models of species interaction and food web simply pushed the environment problem that is constraining species interaction down to the lowest species in the food web or community, namely, to the primary producers (Pastor 2008, pp. 189) . Some population models have, however, been taking the environmental influences on the biological components of ecosystems into account. Review papers by Knowler (2002) and Amstrong (2006) are good examples of such attempts. In general, population modeling tends to view abiotic (nonliving) components as external factors of ecosystems; this is consistent with the *Population- community* approach in ecology.

Ecosystem modeling expands population modeling by integrating the biological and physical components of the environment into a single interactive system (Smith and Smith 1998, pp. 315; Pastor 2008, pp. 189-190). The interaction of living (biological component) and non living (physical component) components in the ecosystem occurs through nutrient flows. All nutrient flow from the nonliving to the living and back to the nonliving components of the ecosystem in a circular path is known as a biogeochemical cycle. This process is called *internal cycling* that represents a recycling of nutrients within the ecosystem, is an essential feature of all ecosystems (Smith and Smith 1998, pp. 343-344). By its very nature, each unit of energy can be used only once, whereas chemical nutrients can be used again, and repeatedly recycled as the building blocks of biomass (Begon, Townsend et al. 2006, pp. 525). Animals and other consumers gain their nutrients by eating producer organisms or each other. When an organism dies, its remains are broken down by decomposers. The components of their cells and tissues are utilized by decomposers and later returned to the environment and recycled (Karleskint 1998, pp. 93-100). All biological entities require nutrients (matter) for their construction and energy for their activities (Begon, Townsend et al. 2006, pp. 499).

Early ecosystem models, which used energy as their "currency", were considered unsuccessful since it is hard to define precisely the energy outflows. Modern ecosystem models thus adopt one or more essential elements, usually carbon, nitrogen, or phosphorus, as their currency (Gurney and Nisbet 1998, pp. 183). Let's consider an aquatic ecosystem with four components: producers (autotrophs), primary consumers (herbivores), secondary consumers (carnivores) and nonliving matter. We assume that the system is closed, so any nutrient taken up by the producers is lost to the free nutrient pool, and all nutrients lost by the producers, primary and secondary consumers due to death and excretion, is immediately added to the nutrient pool (regarding the processes of decay and remineralisation). We denote N (t) as the nutrient density (e.g. gram of nitrogen or phosphorus/m^2) of the nutrient pool. P (t) is denoted as the nutrient density of the primary producers (g/m^2), which is assumed to have a linear functional response, with attack rate (slope) α_p (day^{-1}), and a respiration/mortality rate δ_p(day^{-1}). $H(t)$, α_h, and δ_h are nutrient density (g/m^2), attack rate (day^{-1}) and respiration/mortality rate (day^{-1}) of the primary consumers, respectively; $C(t)$, α_c and δ_c are nutrient density (g/m^2), attack rate (day^{-1}) and respiration/mortality rate (day^{-1}) of the secondary consumers, respectively. We also assume that the secondary consumers feed exclusively on primary consumers and the primary consumers feed exclusively on the producers. With these assumptions, nutrient flows in the ecosystem are described in figure 3.

Figure 3: Nutrient flows in the ecosystem



Source: Adapted from Gurney and Nisbet (1998, pp. 196).

The dynamics of the ecosystem can be described by the following equations (Gurney and Nisbet 1998, pp. 183-200):

$$\begin{cases} \frac{dP}{dt} = \alpha_p PN - \delta_p P - \alpha_h PH \\ \frac{dH}{dt} = \alpha_h PH - \delta_h H - \alpha_c HC \\ \frac{dC}{dt} = \alpha_c CH - \delta_c C \\ \frac{dN}{dt} = \delta_p P + \delta_h H + \delta_c C - \alpha_p PN \end{cases} \quad (1)$$

Equations (1) mean that

$$\frac{dP}{dt} + \frac{dH}{dt} + \frac{dC}{dt} + \frac{dN}{dt} = \frac{d}{dt}(P + H + C + N) = 0$$

In other words, the total quantity of nutrient contained in the system remains constant as expected, given our assumption that the system is closed. If we denote S as the total amount of bounded and unbounded nutrient in the system, then the dynamics of the ecosystem can be rewritten

$$\begin{cases} \frac{dP}{dt} = P(\alpha_p N - \delta_p - \alpha_h H) \\ \frac{dH}{dt} = H(\alpha_h P - \delta_h - \alpha_c C) \\ \frac{dC}{dt} = C(\alpha_c H - \delta_c) \\ N = S - P - H - C \end{cases} \quad (2)$$

The system (2) has one stationary state ($P = H = C = 0$) and three steady states *NP, NPH*, *NPHC* indicate the compartments which contain non-zero biomass *P, (P*, *H)*, (*P, H, C),* respectively. Table 1 shows the steady states for the nutrient cycling model in the ecosystem:[4]

4 The solution is in annex 1.

Table 1. Steady states for the nutrient cycling model (Gurney and Nisbet 1998: 197)

	NP	*NPH*	*NPHC*
*P**	$S-\frac{\delta_p}{\alpha_p}$	$\frac{\delta_h}{\alpha_h}$	$\frac{\alpha_c}{\alpha_c+\alpha_h}\left(S-\frac{\delta_p}{\alpha_p}+\frac{\delta_h}{\alpha_c}-\frac{H^*(\alpha_p+\alpha_h)}{\alpha_p}\right)$
*H**	0	$\frac{\alpha_p}{\alpha_p+\alpha_h}\left(S-\frac{\delta_p}{\alpha_p}-\frac{\delta_h}{\alpha_h}\right)$	$\frac{\delta_c}{\alpha_c}$
*C**	0	0	$\frac{\alpha_h}{\alpha_c+\alpha_h}\left(S-\frac{\delta_p}{\alpha_p}-\frac{\delta_h}{\alpha_h}-\frac{H^*(\alpha_p+\alpha_h)}{\alpha_p}\right)$

With a bit of algebra (annex 2), we can see the rate of increase in nutrient density of producers, primary and secondary consumers per unit of time in system (2) following one of the popular population models, namely, the logistic model

$$\begin{cases} \frac{dP}{dt} = r_p P\left(1-\frac{P}{K_p}\right) \\ \frac{dH}{dt} = r_h H\left(1-\frac{H}{K_h}\right) \\ \frac{dC}{dt} = r_c C\left(1-\frac{C}{K_c}\right) \end{cases} \qquad (3)$$

Where $r_p = \alpha_p(S-H-C)-\delta_p-\alpha_h H$, $K_p = \frac{r_p}{\alpha_p}$, $r_h = \alpha_h(S-N-C)-\delta_h-\alpha_c C$, $K_h = \frac{r_h}{\alpha_h}$, $r_c = \alpha_c(S-N-P)-\delta_c$, $K_c = \frac{r_c}{\alpha_c}$;

The equations (3) imply that the carrying capacity assumption of the species in the logistic population models is a specific case of a closed ecosystem.

Regarding ecological economic aspects, most fishery models ignore the linkages to lower trophic levels of the ecosystems. In particular, environmental data and other bottom up information is widely disregarded (Fennel and Neumann 2004). There are few economic models dealing with the eutrophication phenomenon, which caused by excess inputs of nutrients to ecosystems (Brock and Starrett 1999; Carpenter, Ludwig et al. 1999; Knowler, Barbier et al. 2001; Brock and Zeeuw 2002; Maler, Xepapadeas et al. 2003; Smith and Crowder

2005). In cases of eutrophication, the water becomes turbid because of dense populations of phytoplankton, and large aquatic plants are outcompeted and disappear along with their associated invertebrate populations. Moreover, decomposition of the large biomass of phytoplankton cells may lead to low oxygen concentrations (hypoxia and anoxia), which kill fish and invertebrates. The outcome is a productivity community, but one with low biodiversity and low esthetic appeal (Begon, Townsend et al. 2006).

In this and the preceding sections, concepts and models related to ecosystem-based fishery management are presented. In the next section, the status of implementing ecosystem-based fishery management in practice will be assessed.

5. Implementing Ecosystem-Based Fishery Management in Practice

Ecosystem-based approaches to marine resource management including ecosystem-based fishery management has been criticized as being nonspecific, immature, invalid as a basic for decision making, and not fully supported by science (Murawski 2007). But, there is widespread agreement about the need to implement the ecosystem approach for fisheries in practice (Brodziak and Link 2002; Pikitch, Santora et al. 2004; Pitcher, Kalikoski et al. 2008). Several guidelines for implementing ecosystem-based fishery management have been published such as the papers of Fluharty (1998), Ward (2002) and FAO (2005). These guidelines give detailed instructions for implementing the principles, goals and policies of fisheries management in the ecosystem context. However, the effective application of these guidelines in practice is questionable. In a study by Pitcher *et al* (2008), two-thirds (21) of the 33 countries representing 90% of the world fish catch are unlikely to implement ecosystem-based fishery management (fail grades). Almost all countries had lower ratings for implementation of ecosystem-based fishery management because it is easier to publish good intentions for ecosystem-based fisheries management principles than to actually achieve its goals and objectives in practice (Pitcher, Kalikoski et al. 2008). Managers are just beginning to put some ecosystem-based management principles into practice and this implementation needs to be much greater (Garcia and Cochrane 2005; Arkema, Abramson et al. 2006).

Fluharty (1998) argued that ecosystem-based fishery management can be an important complement to existing fisheries management approaches. However, ecosystem-based management cannot resolve all of the underlying problems of the existing fisheries management regimes. If the political will to stop fishing and to protect habitat is removed, ecosystem-based fishery management cannot be effective (Fluharty, Aparicio et al. 1998).

According to Goodman (as cited in Marasco, Goodman et al. 2007), the move to ecosystem-based fishery management involves three stages. The first stage focuses on managing the target species and its predators and prey. The second stage takes into account environmental effects and the direct effects of fishing activities other than those on the target species (e.g., bycatch, incidental mortality, and effects on habitat). In stage three, the environment, target stock, and its predators and prey are integrated explicitly into an assessment before catch limits and other management measures are selected. Most ecosystem-based fishery models are at present in the second stage and their focus is on individual components of ecosystems. More efforts should be made to integrating different components of ecosystems in ecosystem models.

Many people argue that Marine Protected Areas (MPAs) should be a central element of ecosystem-based fishery management (eg: Palumbi 2002; Browman and Stergiou 2004). However, Sissenwine and Murawski (2004) argued that "*MPAs are just one of a suite of fishery management tools that have merit (and limitations) for either single-species approaches to management, or for ecosystem approaches".* Ecosystem-based fishery management is not synonymous with MPAs, and thus one does not have to implement MPAs in order to be successfully manage resources using ecosystem approach to management (Murawski 2007).

6. Discussions

Since the ecosystem concept is open with regard to spatial scale, it allows researchers to select the scale that is appropriate to a particular objective or interest. It also allows scientists to have different views of the ecosystem and the accuracy of these views will depend on the purpose and the time-space scale of their observations. However, as argued by O'Neill (1986), each view is limited in the specific context. The *process-functional approach* is widely accepted by ecologists but it has limited applications in cases that deal with the effect of dominant populations (species) on ecosystem function. As we know human beings are dominant species on the earth. If humans are seen as one of the species of the ecosystems, the view tends to follow the *population-community* approach, which is an appropriate conceptualization for some observation sets, rather than the best or most fundamental way to view ecosystems. In addition, if humans are one of species in the ecosystem, it is also hard to find a model for ecosystem management because the objective of ecosystem-based fishery management is managing human activities, which are now viewed as behavior of the individual species in the ecosystem model.

Ecosystem-based fishery management cannot resolve all of the underlying problems of the existing fisheries management regimes. Ecosystem-based fishery management is an important complement to existing fisheries management approaches and it should be understood in a broader context rather than individual fishing sector and its application should also take into account the impacts of relevant sectors on the ecosystem. Mcleod (2005) argued that "*managing individual sectors, such as fishing, in an ecosystem context is necessary but not sufficient to ensure the continued productivity and resilience of an ecosystem. Individual human activities should be managed in a fashion that considers the impacts of the sector on the entire ecosystem as well as on other sectors".*

Fishery models largely ignore the linkages to lower trophic levels. In particular, environmental data and other bottom-up information is widely disregarded. Nor are changes in physical environment (bottom-up) alongside both exogenous and endogenous environmental effects included in the general ecosystem models. In addition, the indirect impacts of harvesting (top-down) such as habitat degradation are also rarely taken into account in these models.

Nutrient and energy flows play a vital role in ecosystems. However, they have different characteristics. Although few ecosystems are closed to energy, many are quite close to being closed to nutrients. Even where it is not so, the inflows and outflows of nutrients tend to be easier to define and measure than energetic counterparts. In addition, a more recent observation shows that energy flows inside an ecosystem occur in the form of chemically bound energy, and are thus accompanied by flows of elemental nutrients (Gurney and Nisbet 1998). For those reasons, ecosystem models should concentrate on the internal cycling of nutrients.

The economic value of ecosystem goods and services may be quite different from its market value because many of these service flows are not bought or sold in the markets and therefore do not have market prices. As a result, the measured value is based on the "worse off" or "better off" of individuals feeling, which may lead to different of the same ecosystem goods and services. Diamond and Hausman (1993) argued that the change in well-being when a known resource is injured is not the same as that which occurs when one learns simultaneously about the existence of a resource and an injury to it. They raised a question that if an individual worse off with these two pieces of knowledge than with no knowledge at all (Diamond and Hausman 1993).

MPAs are increasing by being proposed for use as a fishery and ecosystem management tool. However, for species that are highly mobile, one would expect MPAs to be quite ineffective (Sissenwine and Murawski 2004). MPAs also often come with considerable costs to one or more affected constituencies.

Therefore, the social and environmental costs and benefits of MPAs need to be weighed carefully. In addition, other tools include prohibitions on specific activities or harvesting methods, the use of closed seasons for particular activities, input and output controls on natural resources extracted from ecosystem should also be taken into account in the fishery management plans (Murawski 2007).

7. Conclusions

Ecosystems vary in scale so that there are different views of the ecosystem and the accuracy of these views will depend on the purpose and the time-space scale of their observations.

Although the concept of ecosystem-based fishery management has no universal definition, there is a widespread agreement about the need to implement the ecosystem approach for fisheries in practice.

Most ecological economic models ignore the linkages to lower trophic levels. In particular, environmental data and other bottom-up information is widely disregarded. The models are also concentrating heavily on individual components of the ecosystem. More efforts should be paid to integrating different components of the ecosystem as well as external factors in the ecosystem models.

Nutrient flow plays a crucial role in an ecosystem. In addition, it has many properties such as recycling and exchanging between consumers and producers, which are similar to monetary flows in an economy. Therefore, nutrients should be chosen as the currency in ecological economic models.

MPAs is one of important fishery management tools, however, MPAs are not synonymous with ecosystem-based fishery management. It is not necessary to have to implement MPAs in order to be successfully implementing ecosystem-based fishery management.

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Annex 1. Steady States for the Nutrient Cycling Model

In the steady states:

$$\begin{cases} N = S - P - H - C & (1) \\ \dfrac{dP}{dt} = \alpha_p PN - \delta_p P - \alpha_h PH = 0 & (2) \\ \dfrac{dH}{dt} = \alpha_h PH - \delta_h H - \alpha_c HC = 0 & (3) \\ \dfrac{dC}{dt} = \alpha_c CH - \delta_c C \qquad = 0 & (4) \end{cases}$$

If $P^* = H^* = C^* = 0$, the system is hold ($N=S$), we have a stationary state;
If $C^*= H^* = 0$ and $P^* \neq 0$ (steady state NP), equations (*3*) and (*4*) are hold and we have:

$$N^* = S - P^* \qquad (1')$$
$$\alpha_p N^* - \delta_p = 0 \qquad (2')$$

Insert N from (*1'*) to (*2'*) we have: $P^* = S - \frac{\delta_p}{\alpha_p}$

If $C^*= 0$ and $H^*, P^* \neq 0$ (steady state NPH), equation (*4*) is hold and we have:

$$N^* = S - P^* - H^* \qquad (1'')$$
$$\alpha_p N^* - \delta_p - \alpha_h H^* = 0 \qquad (2'')$$
$$\alpha_h P^* - \delta_h \qquad = 0 \qquad (3'')$$

From equation (*3''*): $P^* = \frac{\delta_h}{\alpha_h}$;
Insert P^* and N^* from (*1''*) to (*2''*) we have:

$$\alpha_p\left(S - \frac{\delta_h}{\alpha_h} - H^*\right) - \delta_p - \alpha_h H^* = 0 \leftrightarrow H^* = \frac{\alpha_p}{\alpha_p + \alpha_h}\left(S - \frac{\delta_p}{\alpha_p} - \frac{\delta_h}{\alpha_h}\right)$$

If we have $C^*, H^*, P^* \neq 0$ (steady state *NPHC*), we have:

$$
\begin{aligned}
&N^* = S - P^* - H^* - C^* && (1''') \\
&\alpha_p N^* - \delta_p - \alpha_h H^* = 0 && (2''') \\
&\alpha_h P^* - \delta_h - \alpha_c C^* = 0 && (3''') \\
&\alpha_c H^* - \delta_c = 0 && (4''')
\end{aligned}
$$

From (*4'''*): $H^* = \frac{\delta_c}{\alpha_c}$; Insert H^* and N^* from (*1'''*) to (*2'''*) and (*3'''*), we have:

$$
\begin{aligned}
&\alpha_p(S - P^* - H^* - C^*) - \delta_p - \alpha_h H^* = 0 && (2'''') \\
&\alpha_h P^* - \delta_h - \alpha_c C^* = 0 && (3'''')
\end{aligned}
$$

From (*3''''*): $C^* = \frac{\alpha_h P^* - \delta_h}{\alpha_c}$; put C^* to (*2''''*), we have:

$$
\begin{aligned}
&S - \frac{\delta_p + \alpha_h H^*}{\alpha_p} - H^* + \frac{\delta_h}{\alpha_c} = P^* + \frac{\alpha_h P^*}{\alpha_c} \\
&\leftrightarrow P^*\left(1 + \frac{\alpha_h}{\alpha_c}\right) = S - \frac{\delta_p + \alpha_h H^*}{\alpha_p} - H^* + \frac{\delta_h}{\alpha_c} \\
&\leftrightarrow P^* = \frac{\alpha_c}{\alpha_c + \alpha_h}\left(S - \frac{\delta_p}{\alpha_p} + \frac{\delta_h}{\alpha_c} - \frac{H^*(\alpha_p + \alpha_h)}{\alpha_p}\right)
\end{aligned}
$$

$$
\begin{aligned}
\rightarrow C^* &= \frac{\alpha_h \frac{\alpha_c}{\alpha_c + \alpha_h}\left(S - \frac{\delta_p}{\alpha_p} + \frac{\delta_h}{\alpha_c} - \frac{H^*(\alpha_p + \alpha_h)}{\alpha_p}\right) - \delta_h}{\alpha_c} = \frac{\alpha_h}{\alpha_c + \alpha_h}\left(S - \frac{\delta_p}{\alpha_p} + \frac{\delta_h}{\alpha_c} - \frac{H^*(\alpha_p + \alpha_h)}{\alpha_p}\right) - \frac{\delta_h}{\alpha_c} \\
&= \frac{\alpha_h}{\alpha_c + \alpha_h}\left(S - \frac{\delta_p}{\alpha_p} - \frac{H^*(\alpha_p + \alpha_h)}{\alpha_p} + \frac{\delta_h}{\alpha_c} - \frac{\delta_h}{\alpha_c} * \frac{\alpha_c + \alpha_h}{\alpha_h}\right) \\
&= \frac{\alpha_h}{\alpha_c + \alpha_h}\left(S - \frac{\delta_p}{\alpha_p} - \frac{H^*(\alpha_p + \alpha_h)}{\alpha_p} + \frac{\delta_h \alpha_h - \delta_h \alpha_c - \delta_h \alpha_h}{\alpha_c \alpha_h}\right) \\
&= \frac{\alpha_h}{\alpha_c + \alpha_h}\left(S - \frac{\delta_p}{\alpha_p} - \frac{H^*(\alpha_p + \alpha_h)}{\alpha_p} + \frac{-\delta_h \alpha_c}{\alpha_c \alpha_h}\right) = \frac{\alpha_h}{\alpha_c + \alpha_h}\left(S - \frac{\delta_p}{\alpha_p} - \frac{H^*(\alpha_p + \alpha_h)}{\alpha_p} - \frac{\delta_h}{\alpha h}\right)
\end{aligned}
$$

Annex 2. The Nutrient Cycling Model and the Logistic Equation

$$\begin{cases} \dfrac{dP}{dt} = \alpha_p PN - \delta_p P - \alpha_h PH & (1) \\ \dfrac{dH}{dt} = \alpha_h PH - \delta_h H - \alpha_c HC & (2) \\ \dfrac{dC}{dt} = \alpha_c CH - \delta_c C & (3) \\ N \;= S - P - H - C & (4) \end{cases}$$

Insert equation (4) into equations (1), (2), (3) we will have the following equations

$$\begin{cases} \dfrac{dP}{dt} = P[\alpha_p(S \quad P \quad H \quad C) \quad \delta_p \quad \alpha_h H] \\ \dfrac{dH}{dt} = H[\alpha_h(S - N - H - C) - \delta_h - \alpha_c C] \qquad (5) \\ \dfrac{dC}{dt} = C[\alpha_c(S - N - P - C) - \delta_c] \end{cases}$$

$$\leftrightarrow \begin{cases} \dfrac{dP}{dt} = [\alpha_p(S - H - C) - \delta_p - \alpha_h H]P(1 - \dfrac{P}{\dfrac{\alpha_p(S - H - C) - \delta_p - \alpha_h H}{\alpha_p}} \\ \dfrac{dH}{dt} = [\alpha_h(S - N - C) - \delta_h - \alpha_c C]H(1 - \dfrac{H}{\dfrac{\alpha_h(S - N - C) - \delta_h - \alpha_c C}{\alpha_h}} \qquad (6) \\ \dfrac{dC}{dt} = [\alpha_c(S - N - P) - \delta_c]P(1 - \dfrac{C}{\dfrac{\alpha_c(S - N - P) - \delta_c}{\alpha_c}} \end{cases}$$

Defining $r_p = \alpha_p(S - H - C) - \delta_p - \alpha_h H$, $K_p = \frac{r_p}{\alpha_p}$, $r_h = \alpha_h(S - N - C) - \delta_h - \alpha_c C$, $K_h = \frac{r_h}{\alpha_h}$, $r_c = \alpha_c(S - N - P) - \delta_c$, $K_c = \frac{r_c}{\alpha_c}$ then allows us to recognize (6) as the logistic equation

$$\begin{cases} \dfrac{dP}{dt} = r_p P \left(1 - \dfrac{P}{K_p}\right) \\ \dfrac{dH}{dt} = r_h H \left(1 - \dfrac{H}{K_h}\right) \\ \dfrac{dC}{dt} = r_c C \left(1 - \dfrac{C}{K_c}\right) \end{cases}$$

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