

Biodiversity Conservation in a Game Theoretic Model of a Fishery

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Abstract. This paper proposes a game theoretic modeling framework for the assessment of the trade-off between economic efficiency gains and biodiversity conservation in a fishery. It introduces a biodiversity index, and develops an application of the method couched on Namibian hake fisheries. Results from the illustrative example show that the opportunity cost of achieving a “near perfect” level of biodiversity conservation is high. On a more positive note, however, the results also indicate that it is possible (through appropriate management action) to achieve reasonably high biodiversity levels without losing most of the potential economic gains.

Key words: Cooperative, Noncooperative, Wet fish, Freezer, Trawlers, Namibia

1. Introduction

This paper is about the interaction between biodiversity and human society. It seeks to address questions such as how does human action and activities impact biodiversity? How does the institutional environment under which this interaction takes place affect the potential outcomes both in terms of economic efficiency and biodiversity conservation? Thus, the paper squarely addresses important concerns as expressed by the global community at the 1992 United Nations Conference on Environment and Development (UNCED), and by the Convention on Biological Diversity (UNEP 1995).

Biodiversity refers to the variety of life forms: the different plants, animals and microorganisms, the genes they contain, and the ecosystems they form (Anon. 1992). The concept emphasizes the interrelated nature of the living world and its processes. Biodiversity is important because human beings rely on biological systems and processes for their sustenance and health. Biotic resources¹ also serve recreation and tourism, and underpin the ecosystems, which provide us with many services. Biodiversity also has important social and cultural values. Thus, biological diversity provides so many different types of benefits that studies of this type are necessary to help decision making in situations where biodiversity conservation is an important objective.

Previous economic studies on the issue of biodiversity can be grouped into three. First, there are those that look at diversity indices to assess the marginal contribution of individual species to biodiversity (e.g. Eiswerth and

Hanley 1992, Solow and Polasky 1994, Solow, Polasky and Broadus 1993, Weitzman 1992). Second, we have papers that deal with the valuation of biodiversity or aspects thereof (e.g. Loomis and White 1996). Third, we have analysis that attempt to determine the opportunity cost of achieving targets of survival likelihood for individual species (e.g. Haight 1995, Montgomery, Brown and Adams 1994), or aggregations of viabilities of a group of species (e.g. Bevers, Holf, Kent Raphael 1995). The current paper belongs to the latter two. It focuses on genetic diversity concerns in fisheries, and how this can be incorporated into a bioeconomic model. The paper provides a framework that can be used to value aspects of biodiversity, and help determine the opportunity cost of achieving different levels of biological diversity. It extends the current literature by developing a computational game theoretic model, which explicitly incorporates biodiversity concerns.

One can divide society’s goal for the use of fishery resources into two broad categories, namely, conserving the biological diversity of the resource, and ensuring economic efficiency in its use. It is easy to see that in most situations these goals conflict with each other. To maximize biodiversity fully implies zero harvesting of fish, in most instances. Also, to pursue the absolute maximization of economic benefits will most likely imply the depletion of biodiversity (see for instance, Clark 1973). The challenge for this paper is to develop a modeling framework that would allow us to explore the nature of the trade-offs between economic benefits on the one hand, and biodiversity conservation, on the other.

The next section specifies how biodiversity is modeled. In section 3, we present the game theoretic modeling framework. Section 4 presents a case study of the Namibian hake fisheries in order to demonstrate the

¹ Biota refers to all organisms including animals, plants, fungi and microorganisms in a given area.

potential of the proposed model. Finally, section 5 concludes the paper.

2. Modeling biodiversity

Biological diversity is usually considered at three different levels: genetic diversity, species diversity and ecosystem diversity. Genetic diversity refers to diversity *between* and *within* populations (Norse, Rosenbaum, Wilcove, Wilcox, Romme, Johnston and Stout 1986). Species diversity, refers to the variety of species, it seeks to capture the number of different species occurring in a taxonomic grouping, or the number of different species occurring in a geographic area (Magurran 1988). On the other hand, ecosystem diversity refers to diversity *between* and *within* ecosystems (Norse 1993). The modeling framework developed in this paper is more applicable to biodiversity of the genetic and species types. The focus in the present paper is on genetic biodiversity, as this will allow us to present the workings of the model in a simple way. We specifically study the economics of conserving biodiversity within a population of fish.

Diversity measures generally take into account two factors, namely, *species richness* (that is, number of species) and *evenness*, sometimes known as equitability (Magurran 1988). The present paper looks at evenness within a fish population. It is concerned with maintaining diversity between, say the male and female parts of a population; the juvenile and mature parts of a population; or even the proportion of the different age groups of fish in the population. The index of genetic biological diversity in this paper is therefore the proportion of numbers of male to female, juvenile to mature; the different age groups in a population, or any such measures. The benchmark index is the ratio of say, the standing juvenile biomass to the standing adult biomass when there is no harvesting. This ratio is given a value of 1 or 100%. The benchmark index captures the natural biological diversity of the population, and therefore the “perfect” biodiversity level. The index for any other scenario, I , is then given by the following equation:

(1)

$$I = 1 - \frac{|y - x|}{x}$$

where $x > 0$ denotes the ratio of the juvenile to the mature parts of the biomass when there is no harvesting, and y represents the ratio when there is harvesting. It should be noted that the smaller the value of I the less biodiversity there is in the population, since that signifies either too much or too little juveniles in relation to the ‘no harvest’ situation.

3. Modeling management scenarios

It is assumed that the regulator of a given fishery, say a government ministry, has devolved or delegated the

authority to manage the fishery to the fishers. Hence, they decide how much of the fish to harvest in each period. In such a decentralized setting, the participants can pursue their fishery goals in a cooperative or non-cooperative game situation². Hence, to address the issues raised by this paper, we set up cooperative and non-cooperative game theoretic models³. Two versions of the cooperative model are presented: one in which a biodiversity conservation objective is explicitly incorporated, and one in which it is not. In the case of the latter, we look at cooperative “with” and “without” side payments (Munro 1979, Sumaila 1997).

3.1 The cooperative model

Consider a fishery with two groups of participants⁴, say w and f . Let the net private economic benefits to these groups be B_w and B_f , respectively. These benefits depend positively on the quantity of fish harvested by the two users, that is, H_w and H_f , which in turn depend on the stock size, N , and the amount of fishing effort, e , that each player takes out fishing. On the other hand, B_w depends negatively on the amount of fish harvested by f , and B_f depends negatively on the harvest of w .

Formally, we have,

(2)

$$B_w(H_w) := B_w(N, e_w);$$

$$B_f(H_f) := B_f(N, e_f);$$

Where

$$\partial B_w / \partial H_w > 0; \partial B_w / \partial N > 0;$$

$$\partial B_w / \partial e_w > 0; \partial B_w / \partial e_f < 0;$$

$$\partial B_f / \partial H_f > 0; \partial B_f / \partial N > 0;$$

$$\partial B_f / \partial e_f > 0; \partial B_f / \partial e_w < 0;$$

Under cooperation, it is assumed that the objective is the optimization of long-term *joint* benefits from their use of the resource. The problem of the cooperating agents is therefore to maximize total net joint benefits, B_t , by choosing harvest levels ($H_{w,t}$ and $H_{f,t}$ $t=1..T$) or fishing

² By a game we mean any activity involving two or more participants, each of whom recognizes that the outcome for himself depends not only on his own actions, but also those of other participants.

³ A non-cooperative game is one in which there is no “good” communication between the players in the game; no binding contracts can be entered into; and players take the actions of the others in the game as given, and then decide their own actions unilaterally. A cooperative game is the opposite of the non-cooperative (see Nash 1953).

⁴ For mathematical and computational convenience, we develop what is termed in game theoretic terminology as “two-person” games. In principle, however, the qualitative results of the paper should be valid also for n -person games ($n \geq 2$).

effort levels ($e_{w,t}$ and $e_{f,t}$ $t=1..T$), where T is the last (terminal) period:

$$(3) \quad \max_{e_{w,t}, e_{f,t}} \sum_{t=1}^T [B_t] p_{t-1}$$

subject to

$$N_{a,t+1} = N_{a,t} - H_{a,t}^w - H_{a,t}^f; \text{ for } t=1..T,$$

$$a = 0, 1, \dots, a^m, \dots, A, N_{0,t} = R_t; N_{a,0} \text{ given}$$

and the obvious non-negativity constraints.

In equation 3,

$$B_t = \omega B_{w,t} + (1 - \omega) B_{f,t} \text{ and}$$

R_t is the recruitment function.

The parameter ρ is the discount factor, and r is the discount rate; $N_{a,t}$ is the biomass of age a fish in period t . a^m is age at maturity and A is the last age group; ω denotes the weight put on the preferences of w , and $1 - \omega$ refers to the weight placed on the preferences of f . Note that the above formulation assumes that the stock of fish in the previous period determines the availability of fish for current use. It should be noted that the biological model specified is multi-cohort age structured, this can in principle be easily extended into a multi-species model.

Returning to the biodiversity index, we note that from the stock constraint in equation 3, the total juvenile stock size, N_j , is given by $\sum_{a=0}^{a^m-1} N_{a,t}$. Similarly, the total mature

stock size, N_m , is $\sum_{a=a^m}^A N_{a,t}$. Hence, x , the ratio of the juvenile to the mature parts of the biomass when there is no harvesting, is obtained by computing N_j divided by N_m when the model is run without harvesting. The variable y , the ratio when there is harvesting, is calculated similarly but this time using results from a run of the model when there is harvesting.

The first stage in the analysis involves the computation of the optimal solution to the problem expressed by equation 3 for different values of the preference parameter ω . The next stage entails inspecting the outcomes for the different ω values in order to identify the best outcomes under the different versions of the cooperative model. In the cooperative with an explicit biodiversity conservation objective, the optimal ω is the one that produces the *highest biodiversity index, I*. In the cooperative with side payments management scenario, the optimal ω is the one that *maximizes joint discounted economic benefits*, even if that implies pensioning out one of the players. Finally, under the cooperative without side payments regime, the optimal ω is the one that *maximizes the product of the*

distance between the cooperative outcomes and the threat point outcomes of the players [see 14]⁵.

3.2 The non-cooperative model

This model attempts to capture a situation in which devolving the responsibility to manage the resource leads to non-cooperative behavior. This is likely to happen when there is no communication between the participants; the parties can enter into no binding agreements, and each of them goes about exploiting the resource unilaterally (see Sumaila 1999) and the references therein]. There are a number of good reasons why it is important to develop this model. First, it is well known that most of the time interaction between agents exploiting open access, and common property resources usually degenerate into something close to the kind of outcomes predicted by non-cooperative models. Evidence of this is to be found in the many collapses of fish stocks around the world over time⁶. Second, it is important to keep reminding stakeholders, regulators and the general public of the potential losses from non-cooperative behavior, with the hope that this will eventually help bring about more cooperative management regimes in the world's shared fishery and other common property natural resources.

Under non-cooperation, it is assumed that the objective of each player is to maximize *own private* benefits, B_p , $p=w,f$, from the use of the resource. The problem of the non-cooperating agents is therefore to choose effort levels in each fishing period so as to achieve their own objective, without due regard to the consequences of their action on other participants.

There are two key differences between the cooperative with an explicit biodiversity objective and the non-cooperative model. First, the users in the non-cooperative setting do not incorporate biodiversity concerns into their reaction functions – they care only about the private benefit that accrues to them. Second, users race for the fish, as they unilaterally decide how much to take. It should be noted that the latter is also the main difference between the non-cooperative and the other versions of the cooperative model.

The algorithm developed in (Flåm 1993) is applied to compute the solutions to the above models. Appropriate modified Lagrange multipliers are written for this problem as in (Sumaila 1997) and solved numerically using the simulation package known as Powersim.

⁵ A threat point in this analysis refers to the noncooperative payoffs to the players.

⁶ Examples are Norwegian spring spawning herring (see Bjørndal, Hole, Slinde, Asche and Reithe); Atlantic cod fisheries off Newfoundland (see Walter and Maquire 1996); and Peruvian anchovies (see Idyll 1973).

4. A case study: Namibian hake fishery

The aim in this section is to use the Namibian hake fisheries to illustrate the possible outcomes under the cooperative and non-cooperative versions of the models presented above. Hake is one of the three most important fish species of the highly productive Namibian Exclusive Economic Zone (EEZ). The others are horse mackerel and pilchard. Among the species of hakes inhabiting the Namibian EEZ, that is, *Merluccius capensis* (also known as cape hake), *Merluccius paradoxus* (deep-water hake) and *Merluccius pollis*, only the former two are of major importance to the fishery. Both species are relatively long-lived, reaching ages of up to and over 12 years. Hakes are usually found close to the bottom of the water column during daytime but rise to intermediate levels during nighttime, probably following their prey.

A variety of fishing vessels are used to harvest hake; differing in their gross registered tonnage (GRT), engine horse power (HP), processing equipment, and freezing capacity. However, mainly wet fish and freezer trawlers land the hake stocks. For instance, in 1994, out of a total of 108213 tons of hake landed, 99152 tons were by wet fish and freezer trawlers. This is well over 90% of the total landings of hake for that year. The rest is landed using monk/sole trawlers, longlines, and mid-water trawlers [see Moorsom 1994]. 1996 data show that this picture has not changed in any significant way (Anon. 1997). The paper therefore focuses attention on these vessels and organizes the wet fish and freezer trawlers into two separate entities denoted w and f , respectively. An important feature of these two vessel groups is that while the wet fish trawlers fish closer to shore and therefore harvest mainly juvenile hake, the freezer trawlers fish further off shore and consequently land mostly mature hake (Les Clark, Ministry of Fisheries and Marine Resources, pers. Comm.). These different features are captured through the catchability coefficients of the vessels [see Sumaila 1997].

4.1 The biological specifics of the model

The biological foundation for the Namibian hake fisheries is represented by the equations given below. The spawning biomass, B_t^s :

$$(5) \quad B_t^s = \sum_{a=0}^A p_a w_a n_{a,t}$$

where $a=0,1,\dots,A$, denotes age group a hake; A is the last age group. The variable w_a stands for weight of hake of age a at the start of the year. $t=1,2,\dots,T$ is fishing years, with T denoting the terminal period, p_a stands for the

proportion of age a hake that is mature, and $n_{a,t}$ represents the number of age a hake in year t .

The stock-recruit relationship, R_t :

$$(6) \quad R_t = n_{0,t} = \frac{\alpha B_t^s}{(\alpha\beta + B_t^s)^{\gamma}}$$

where $n_{0,t}$ is the number of recruits in year t ; and α, β, γ are parameters of the extended Beverton Holt stock-recruit relationship (Punt 1988).

From the above, the basic stock dynamics can be represented by the equations below:

$$(7) \quad \begin{aligned} n_{a,t} &= \phi n_{a-1,t-1} - h_{a,t}, \text{ for } 0 < a < A \\ n_{A,t} &= \phi n_{A,t} - \phi n_{A-1,t-1} - h_{A,t}, n_{a,0} \text{ given} \end{aligned}$$

The function $h_{a,t} = \sum_p q_{p,a} n_{a,t} e_t$ denotes the total harvest

by both players of age group a hake in fishing period t ; ϕ is the age independent natural survival rate; e_t is the fishing effort exerted on hake in period t , while q stands for the catchability coefficient of the hake harvesting vessels. It should be noted that the stock dynamics of the last age group of hake is given special treatment. This is meant to capture the fact that all age A hake do not die at the end of a given period.

4.2 The economic specifics of the model

For the two players, $p=w, f$, let the harvest cost function in period t , $C(p,t)$, be defined as

$$(8) \quad C(e_{p,t}) = \frac{k_p e_{p,t}^{1+b}}{1+b}$$

where $b = 0.01$, and $k_p/(1+b) \approx k_p$ is the cost of engaging one fishing fleet for one year.

This formulation is chosen to give the cost function a ‘‘slight’’ convexity to help convergence in the model [see 7]. The single period profit of player p is given by:

$$(9) \quad \pi_p = \sum_{a=0}^A v_p w_a q_{p,a} n_{a,t} e_{p,t} - \frac{k_p e_{p,t}^{1+b}}{1+b}$$

where $n_{a,t}$ is the age- and period-dependent stock size in number of fish; w_a is the mean weight of fish of age a ; and $q_{p,a}$ is the age and player dependent catchability coefficient, that is, the share of age group a hake being caught by one unit of fishing effort of player p .

The discounted private economic rent or net benefit to player, p , is denoted:
(10)

$$B_p(n, e_p) = \sum_{t=1}^T \delta_p^t \pi_p(n_t, e_{p,t})$$

where $\delta_p = (1 + r_p)^{-1}$ is the discount factor. The variable n (n_t) is the post-catch stock matrix (vector) in number of fish; and r_p denotes the interest rate of player p .

Age (a) Yrs	Selec- tivity Sa	Catchability W f	Propor- tion mature	Wei- ght Kg	Initial nos. million s
0	0	0 0	0.0	0.001	2.00
1	0.007	0.0062 0.0060	0.0	0.035	1.30
2	0.032	0.0031 0.0162	0.0	0.094	0.64
3	0.216	0.0207 0.0162	0.0	0.187	0.40
4	0.426	0.0384 0.0162	0.5	0.319	0.28
5	0.972	0.0576 0.0004	1.0	0.550	0.18
6	1.028	0.0580 0.0060	1.0	0.929	0.13
7	1.000	0.0000 0.0162	1.0	1.445	0.10
8	1.000	0.0000 0.0162	1.0	2.108	0.04
9	1.000	0.0000 0.0162	1.0	2.542	0.03

Table 1: Values of the parameters used in the model. Maximum age, total selectivity and weight are taken from (Punt and Butterworth 1991). Catchability coefficients derived. Initial stock size and proportion mature estimated.

4.3 Model data

The biological, economic and technological data employed for the simulations are mostly taken from (Punt and Butterworth 1991, Punt 1988, Sumaila 2000 *in press*) and the Ministry of Fisheries and Marine Resources, Windhoek, Namibia (MFMR). Table 1 presents (i) the proportion mature of each age group, p_a , (ii) the average weight, w_a , (iii) the total selectivity for each age group, S_a , (iv) the initial numbers of each age group of fish, and (v) the catchability coefficients for each vessel type. The latter are calculated by splitting the total selectivity presented in (Punt and Butterworth 1991) according to the observed targeting patterns of juvenile and mature hake by the two vessels, and using the framework in Appendix 1 of Sumaila 1997.

The rest of the model parameters are given the values: $\alpha=6300$ (million) $\beta = 0.16$; $\gamma=1.0$ [18]; $A= 9$ (Punt and Butterworth 1991). Natural survival rate, θ , is assumed to be 0.81 per year. Price per kilogram for the landings of the wet fish ($v_1=N\$ 8.18$) and freezer ($v_2=N\$ 7.38$) trawlers are taken from (Sumaila 2000). The cost of engaging the wet fish and freezer trawlers for one year are determined from fishing industry data to be N\$12.29 and N\$ 39.90 million, respectively. A discount factor of 0.952 (equivalent to a real interest rate of 5%) is assumed.

Table II: Reports the biodiversity index (in percentage) and the discounted economic rent (in billion N\$) for different ω -values in the cooperative scenarios, and for the non-cooperative model. The best ω -values under the different management regimes are depicted in bold/italics.

ω	Rent to w	Rent to f	Total rent	Bio-diversity index
0.0	0	7.52	7.52	93
0.1	0.18	5.01	5.19	96
0.2	1.73	3.2	4.93	98
0.3	2.96	2.14	5.1	91
0.4	4.07	1.58	5.65	86
0.5	5.12	1.24	6.36	82
0.6	6.18	0.96	7.14	78
0.7	7.09	0.66	7.75	74
0.8	7.96	0.41	8.37	70
0.9	9.01	0.16	9.17	67
<i>1.0</i>	<i>10.23</i>	<i>0.00</i>	<i>10.23</i>	<i>65</i>
Non-coop	4.63	0.50	5.13	56

4.4 The results

The general results of the study are presented in Table II. The table displays the biodiversity index and the discounted economic rent for the different ω values. We see from this table that in the case of the cooperative with

an explicit biodiversity conservation objective, the optimal ω value is 0.2, as this is the ω value that gives the highest biodiversity index. At $\omega = 0.2$, the index is 98% of the ideal “no harvest” scenario. When it comes to the cooperative with side payments scenario, the optimal ω is 1, as this gives the highest possible joint discounted economic rent. On the other hand, the optimal ω for the cooperative without side payments turned out to be 0.6. This gives the Nash solution for a cooperative without side payments management scenario [15]. The result for the non-cooperative scenario is presented in the last row of the table.

To discuss the trade-off involved between biodiversity conservation and economic benefits from the use of hake, Table III and Figure 1 are presented below. From this table and figure, we see that the highest biodiversity index is achieved under the cooperative with an explicit biodiversity conservation objective scenario. With an index of 98%, this version of our model conserves biodiversity best. On the other hand, the non-cooperative model delivers the worst biodiversity conservation, with only 56% of the ‘no harvest’ biological diversity preserved. In between these two extremes lie the outcome of 65% in the case of cooperative with side payments, and 78% in the case of the cooperative without side payments.

Table III: Presents the total discounted rent (in billion N\$), the biodiversity index (in percentage) and the average total annual harvest (in thousand tonnes) for the optimal ω under the different scenarios.

	Rent	Biod. index	Harvest
Coop. with biodiversity	4.94	98	58
Coop. with side payments	10.23	65	129
Coop. without side payments	7.14	78	84.6
Noocoperative	5.13	56	86.9

Turning to the economic results, we see that the cooperative with side payments outcome produces the highest joint discounted economic rent, thus achieving what [14] calls the *optimum optimorum* (N\$10.23 billion). The next best economic result is achieved under the

cooperative without side payments scenario (N\$7.14 billion). This is followed by the non-cooperative outcome (N\$5.14 billion), and the cooperative with an explicit biodiversity objective produces an economic result of N\$4.94 billion, respectively. These economic results are reflected in the harvest levels under the different scenarios. The only exception here is in the case of the non-cooperative scenario, where even though the harvest level is slightly more than that in the cooperative without side payments scenario, and about 50% higher than in the case of the explicit biodiversity objective scenario, the economic rent is much lower relative to that in the case of the former, and nearly the same in the case of the latter.

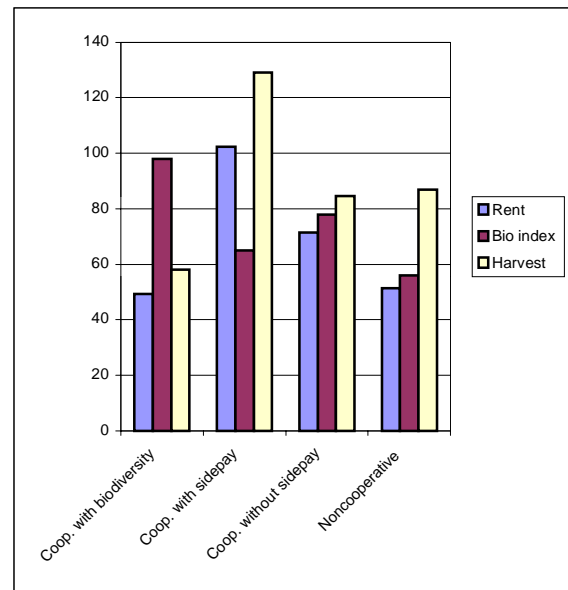


Figure 1: Plots the biodiversity index (in percentage), the discounted economic rent (in 100 million N\$) and the average total annual harvest (in thousand tonnes) for the optimal ω under the different management regimes.

5. Concluding remarks

A number of interpretations, observations and deductions can be made from the results presented above. First, it is clear that the opportunity cost of achieving “near perfect” level of biodiversity conservation is high. This turned out to be over 50% of the potential discounted economic rent in our model. Second, a cooperative without side payments management scenario appears to do reasonably well on both economic efficiency and biodiversity conservation criteria. This scenario produces an economic result, which is 70% of the potential discounted economic rent, and about 78% of the ‘no harvest’ biodiversity level. Hence, it is possible to attain reasonably high biodiversity level, and reasonably high economic rent at the same time. Third, non-cooperative behavior is (as has been shown again and again by previous studies) bad on all

counts – it delivers only 56% of the potential biodiversity level, and just over 60% of the potential economic rent. Fourth, the bargaining powers of the players, w and f , varies with the type of management scenario. For instance, in the scenario with an explicit biodiversity objective, the freezer trawlers do most of the harvesting and therefore make the most economic rent. On the other hand, the wet fish trawlers do most of the harvesting in the cooperative without side payments scenario.

This paper has presented a modeling framework that can be used to analyze and study the impact of different management scenarios on biodiversity conservation in a bioeconomic model. The modeling approach can help us examine the trade-offs between biodiversity conservation and economic considerations. It can, in this way, inform policy on what appropriate management scenarios to put in place.

A simple application of the approach has been presented to show the potential of the approach. More complex applications in multi-species fisheries are possible. In principle, the main limitation of the modeling approach is the “curse of dimensionality”, that is, limitations imposed by a lack of adequate computational power.

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