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Optimal Fishing Policies That Maximize Sustainable Ecosystem Services

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The classic theory of fisheries management seeks a maximum sustainable yield from a target species. There are several variations that include uncertainty, fluctuation, and species interactions. The knowledge that sustainable fisheries do not always guarantee conservation of a diversity of species is well known. Ecosystems provide several categories of ecosystem services to human well-being: supporting, provisioning, regulating, and cultural services. Fishery yields belong to provisioning services. The existence of living marine resources may maintain these services, and certainly a much larger contribution from regulating services than that from fishery yields. Therefore, we define an optimal fishing strategy that maximizes the total ecosystem service instead of a sustainable fishery yield. We call this the fishing policy for "maximum sustainable ecosystem service" (MSES). The regulating service likely depends on the standing biomass, while the provisioning service from fisheries depends on the catch amount. We obtain fishing policies for MSES in a single species model with and without uncertainties and in multiple species models. In any case, fishing efforts are usually much smaller than those for a maximum sustainable yield (MSY). We also discuss the role of fisheries in sustaining ecosystem services, and the nature of ecosystem comanagement.

KEYWORDS maximum sustainable ecosystem service; MSES; uncertainty; ecosystem comanagement; maximum sustainable yield

1. Introduction

The theory of maximum sustainable yield (MSY) takes into account the long-term yield from living marine resources. This theory implicitly assumes that a fishing ban does not produce any benefit from marine ecosystems, although this theory explicitly assumes a negative relationship between yield and standing stock abundance. However, ecosystems give us a variety of benefits (World Resource Institute 2005). Harvests from agriculture, forestry and fisheries are just a small part of the ecosystem service (Costanza *et al*. 1997; Satake and Rudel 2007). Ecosystem services include supporting services such as soil formation, photosynthesis, and nutrient cycling, provisioning services such as food, water, timber, and fiber; regulating services that affect climate, floods, disease, waste, and water quality; and cultural services that provide recreational, aesthetic, and spiritual benefits (World Resource Institute 2005). Fishery yields belong to provisioning services. The existence of living marine resources may maintain these services and certainly make a larger contribution to the total ecosystem service than fishery yields (Costanza *et al*. 1997).

There are some criticisms of the theory of MSY (Matsuda and Abrams 2008). The MSY fishing policy does not reflect uncertainty in stock estimates (measurement errors) and in the relationship between the spawning stock and recruitment (process uncertainties). The MSY policy also ignores the complexity in ecosystem processes because the theory uses single stock dynamic models (Matsuda and Katsukawa 2002). Matsuda and Abrams (2006) analyzed the maximum sustainable yield from entire food webs with an independent fishing effort for each species. They concluded that MSY policy does not guarantee the coexistence of species and proposed the concept of "constrained MSY" that maximizes the sustainable yield under which all species coexist. However, Matsuda and Abrams (2006) did not incorporate ecosystem service into the optimal fisheries policy.

In this paper, we consider an optimal policy that maximizes the total ecosystem service under some mathematically simple assumptions. We call this the maximum sustainable ecosystem service, or MSES. We assume that an ecosystem service other than fishery yields depends on the standing biomass, while the fishery yield depends on the catch amount. Regulation of fishing efforts usually enhances the standing biomass, while it usually decreases the fishery yield. We compare the MSY and MSES policies in (1) a single deterministic stock dynamic model, (2) a stochastic model for a single species with measurement errors and process uncertainty and (3) food web models consisting of six species. We also discuss the role of comanagement in the ecosystem approach.

2. Optimal Fishing Policy That Maximizes Ecosystem Service

To take account these ecosystem services, we assume that the total ecosystem service from a target fish resource, denoted by $V(N, C)$ is given by

$$V(N, C) = Y(C) - cE + S(N) \tag{1}$$

where $Y(C)$ is the yield from fisheries with catch amount C, cE is the cost of fisheries with fishing effort E, and $S(N)$ is the ecosystem service other than fishery yields with stock abundance N. Hereafter, we simply call $S(N)$ the utility of standing biomass.

We assume the following fish stock dynamics :

$$\begin{aligned} dN/dt &= f(N)N - C, \\ f(N) &= r - aN, \\ C &= qEN, \\ Y(C) &= pqEN \end{aligned} \tag{2}$$

where t is an arbitrary time unit, $f(N)$ is the per capita reproduction rate, r is the intrinsic rate of population increase, a is the magnitude of density effect, q is catchability, and p is the price of fish. Although we have assumed linear functions for cost (cE) and yield ($pqEN$), the nonlinear relationship for either cost or catch may produce a more complex result. The carrying capacity, denoted by K, is given by r/a.

The utility of standing biomass, $S(N)$, is likely to be a convex or sigmoid curve because the utility usually saturates when the stock is sufficiently abundant. We assume the following specific function:

$$S(N) = S^{\infty}N^2/(B^2 + N^2) \quad (3)$$

where S^{∞} is the limit of $S(N)$ when N is the positive infinity, B is the stock abundance where $S(N)$ is the half of S^{∞}. We assumed a sigmoid function of N as shown in Fig. 1.

In the first step we obtain the optimal fishing effort that maximizes the total ecosystem service $V(N, C)$ given by Eq. (1) at equilibrium stock abundance of Eq. (2). The equilibrium stock abundance, denoted by N^*, is given by

$$N^* = (r - qE)/a \quad (4)$$

The total ecosystem service at the equilibrium, denoted by V^*, is

$$V^* = pqEN^* - cE + S(N^*) \quad (5)$$

If the utility of the standing biomass is negligible ($S(N) = 0$) as is usually assumed in classical fisheries theory, V^* is a uni-modal function of E and the optimal fishing effort, denoted by E_{opt}, is well known by

$$E_{\mathrm{opt}} = (pqr - ac)/2pq^2 \quad (6)$$

If cost c is negligible, E_{opt} becomes the effort at the maximum sustainable yield (MSY). The quantity V^* when $E = E_{\mathrm{opt}}$ is known as the maximum economic yield.

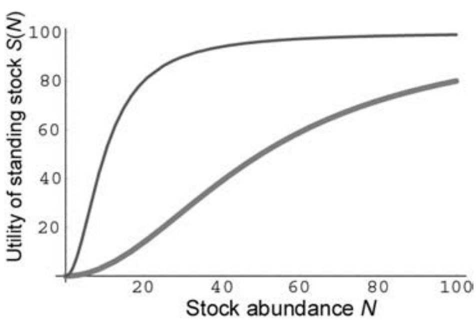

Fig. 1. The relationship between stock abundance and its utility given by Eq. (3). Bold and thin curves represent the cases where B is 50 and 10% of K, respectively.

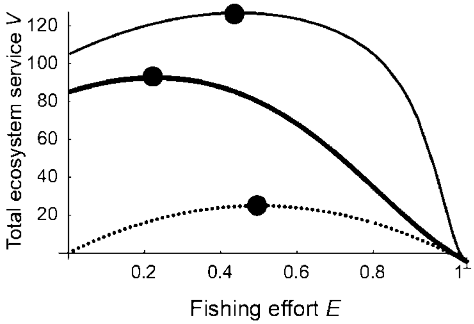

Fig. 2. The relationship between the fishing effort and the total ecosystem service for three cases of parameters. Bold, thin and dotted curves represent the cases where (S^{∞}, B) = (100, 10), (100, 50), (0, –), respectively. Other parameters are chosen as $a = 0.01$, $r = 1$, $q = 1$, $p = 1$, $c = 0.1$. The optimal effort for each case is indicated by closed circles.

If $S(N)$ is an increasing function of N, the optimal effort is always smaller than E_{opt} without the utility of standing biomass. However, the distance between E_{opt} with positive $S(N)$ and E_{opt} without $S(N)$ depends on the magnitude and curvature of $S(N)$. We numerically obtained the optimal effort for various parameters of S^{∞} and B as shown in Fig. 2. Despite the fact that the maximum $V_{\mathrm{opt}}(N^*, C^*)$, when $(S^{\infty}, B) = (100, 50)$ is larger than $V_{\mathrm{opt}}(N^*, C^*)$ when $(S^{\infty}, B) = (100, 10)$, the latter optimal effort is smaller than the former.

The derivative of V^* with E is

$$dV^*/dE = pqN^* - c - [pqE + s'(N^*)]q/a \quad (7)$$

because $dN^*/dE = -q/a$.

The optimal effort is smaller when the derivative of utility of the standing resource with respect to the stock is of a larger magnitude ($|dS/dN|$ is larger). If $S^\infty > (Ba^2 + r^2)^2(rpq - ac)/2Bqra^3$, dV^*/dE at $E = 0$ is negative and the fishing ban is optimal. For parameter values given in Fig. 2, the fishing ban is optimal when $S^\infty > 47595$ if $B = 10$ or when $S^\infty > 9995$ if $B = 50$.

3. Optimal Fishing Policy with Process Uncertainty and Measurement Errors

Equation (1), that includes the value of ecosystem service, is simple because any types of uncertainties, intrinsic instability, complexity and conflict between stakeholders are ignored. We incorporate these factors into the model to build the comprehensive theory of fisheries management. In the second step we consider a time-discrete model with the uncertainties in measurement and dynamic processes:

$$\begin{aligned} N_{t+1} &= f(N_t - C_t, \xi_t)(N_t - C_t) \\ Y(C_t) &= pC_t = pqE_tN_t \\ \tilde{N}_t &= G(N_t, \zeta_t) \\ V(N_t, C_t) &= Y(C_t) - cE_t + S(N_t - C_t) \\ E_t &= F(\tilde{N}_t)/q \end{aligned} \quad (8)$$

where N_t is the stock abundance in year t just before the fishing season, ξ_t and ζ_t are normally-distributed random variables with standard deviations σ_r and σ_m, $f(N_t - C_t, \xi_t)$ is the per capita reproduction rate with process uncertainty $\xi(t)$, $\tilde{N}_t$ is the estimate of stock abundance N_t in year t that is a function G of the true abundance N_t and measurement error $\zeta(t)$. In this case, the fishing effort E_t likely depends on time because the stock fluctuates with process errors. In the decision making, fishers just use the estimate $\tilde{N}_t$, therefore E_t is a function F of $\tilde{N}_t$.

We here assume one of the simplest ways that is similar to Reed (1979):

$$f(N_t, \xi_t) = \exp(r + \xi_t - aN_t) \quad (9)$$

We also incorporate measurement uncertainty:

$$G(N_t, \zeta_t) = N_t \exp(\zeta_t) \quad (10)$$

The optimal harvesting policy is obtained by the dynamic programming theory (Mangel and Clark 1988) although the solution is still analytically unknown if a measurement error exists (Kotani *et al.* 2008). Here we simply assume that the allowable effort rule in actual fisheries management in the USA, Japan, and other countries as shown in Fig. 3:

$$F(\tilde{N}) = \begin{cases} 0 & \text{if } \tilde{N}_t < N_{\text{ban}} \\ F_{\text{target}}\left(\tilde{N} - N_{\text{ban}}\right)/\left(N_{\text{limit}} - N_{\text{ban}}\right) & \\ & \text{if } N_{\text{ban}} \le \tilde{N}_t < N_{\text{limit}} \\ F_{\text{target}} & \text{if } \tilde{N}_t \ge N_{\text{limit}} \end{cases} \quad (11)$$

where N_{ban} and N_{limit} are the stock abundances of the upper threshold for a fishing ban and the lowest threshold that maintains a fishing effort F_{target}, F_{target} is the target fishing effort under which the maximum sustainable yield is theoretically achieved. The conventional fisheries management under the United Nations Convention on the Law of the Sea (UNCLOS) usually determines the total allowable catch (TAC) instead of the fishing effort. Therefore, the actual catch should have been the minimum of qE_tN_t and $qE_t\tilde{N}_t$.

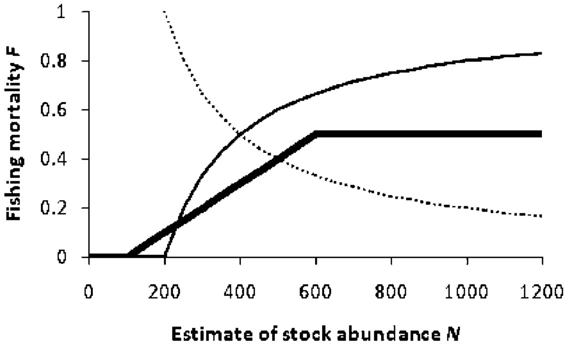

Fig. 3. Rules of fishing effort that depend on the estimate of stock abundance. Bold, thin and dotted curves represent the ABC rule given by Eq. (11), constant escapement given by Eq. (12) and constant catch given by Eq. (13), where q = 1, N_{crit} = 100, C_{c} = 200, N_{limit} = 600, F_{target} = 0.5.

Here we simply assumed that C_t is given by qE_tN_t or that the control of fishing effort depends on the estimate of stock abundance (see also Katsukawa 2004). The fishing mortality coefficient F is approximately given by qE.

If we ignore the utility of standing biomass ($S(N)$ = 0) and measurement error (σ_{m} = 0), the constant escapement policy is well known as the optimal policy that maximizes the long-term yield:

$$E_{cs}(N) = \begin{cases} 0 & \text{if } N_t < N_{\text{crit}} \\ \left(N - N_{\text{crit}}\right)/qN & \text{if } N_t \geq N_{\text{crit}} \end{cases} \tag{12}$$

where N_{crit} is a positive constant. The catch $qE_{t,\text{opt}}N_t$ is $(N_t - N_{\text{crit}})$ and the stock abundance after the fishing season is N_{crit}. The optimal escapement level depends on the magnitude of process uncertainty (σ_{r}). The fishing effort for the constant catch policy is

$$E_{\text{cc}}(N) = C_{\text{c}}/qN \tag{13}$$

where C_{c} is the constant catch amount.

If the measurement error exists ($\sigma_{\text{m}} > 0$), Katsukawa (2004) numerically obtained a suitable management policy that keeps a larger average yield and a larger minimum stock with a smaller variance. This solution implicitly incorporates the utility of standing biomass into the performance of fishing policy. Katsukawa (2004) concluded that the fishing effort that achieves the maximum sustainable yield is a bad performance. He recommends a simpler rule given by Eq. (11) that is similar to the New Management Procedure of the International Whaling Commission.

Kotani *et al.* (2008) considered the optimal control of exotic species and numerically obtained an optimal harvesting policy when the utility of standing population is negative. Even for the optimal control of exotic species, the constant escapement is optimal.

Figure 4 illustrates a resultant stock abundance, catch amount from MSY, and MSES policies. We obtained the optimal policies by grid search for F_{target}, N_{ban}, N_{limit} as 0.1, 100, 100, respectively, to maximize the average yield or ecosystem services over 100 years from 100 simulations. The fishing effort under the optimal policy for MSY was higher than that for MSES, while the stock abundance under the optimal policy for MSY was lower than that for MSES. We chose parameters (S^{∞}, B, r, a, σ_{r}, σ_{e}, p, c) = (100, 1000, 0.5, 0.001, 30%, 50%, 1, 0). We obtained

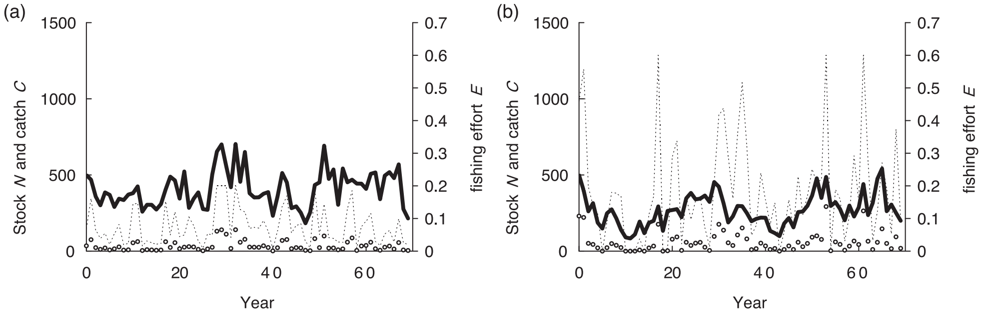

Fig. 4. An example of stock and catch fluctuation under fishing effort given in Eq. (11). Policies for MSY and MSES are shown in panels (a) and (b), respectively. Bold and dotted curves represent the stock abundance and fishing effort. Circles represent the catch.

a fishing policy characterized by (F_{target}, N_{ban}, N_{limit}) in Eq. (11). The policy for MSY was (F_{target}, N_{ban}, N_{limit}) = (0.6, 100, 700), whose average yield and average total ecosystem services were 77.6 and 165.7, respectively. The policy for MSES was (F_{target}, N_{ban}, N_{limit}) = (0.2, 100, 1000), whose average yield and average total ecosystem services were 43.3 and 216.4, respectively.

4. Optimal Policy from Food Webs

Ecosystems are characterized by uncertainty, dynamic properties and complexity. We have analyzed the effects of uncertainty and dynamic properties previously. Matsuda and Abrams (2006) used simple food-web models to investigate the nature of yield- or profit-maximizing exploitation of communities including a variety of six-species systems with as many as five trophic levels. These models show that, for most webs, relatively few species are harvested at equilibrium and that a significant fraction of the species is lost from the web. They also considered a constraint that all species must be retained in the system usually increases the number of species and trophic levels harvested at the yield-maximizing policy. The reduction in total yield caused by such a constraint is modest for most food webs.

Matsuda and Abrams (2006) did not explicitly evaluate the ecosystem service of standing biomass but added a constraint of species persistence. Here we incorporate the utility of ecosystem service from standing biomass into their model that includes s species:

$$dN_i/dt = (r_i + \sum a_{ji}N_j - q_iE_i)N_i, \quad \text{for } i = 1, 2, \ldots, s \tag{14}$$

where $N_i(t)$ is the stock abundance of species i at time t, r_i is the intrinsic growth rate of population increase of species i, a_{ji} is the magnitude of intra- and inter-specific competition from species j to i, and E_i is the fishing effort on species i.

We define the total ecosystem service from the food web:

$$\begin{aligned} Y(\mathbf{E}) &= \sum E_i(p_iq_iN_i - c_i) \\ V(\mathbf{E}) &= Y(\mathbf{E}) + \sum S_i(N_i) \end{aligned} \tag{15}$$

where p, q, c and S differ between species. Matsuda and Abrams (2006) ignored the utility of standing biomass $S_i(N_i)$. There may be some multiplicative effects of interspecific interactions on the ecosystem service. It is difficult to quantitatively evaluate the total ecosystem service from the web. Here we assumed the simple sum of the contribution of each species on the total ecosystem service.

$$S_i(N_i) = S_i^{\infty} N_i^2/(B_i^2 + N_i^2) \tag{16}$$

We evaluate the total ecosystem service at the equilibrium. The equilibrium, denoted by $\mathbf{N}^* = (N_1^*, N_2^*, ..., N_s^*)^T$, is given by

$$\mathbf{N}^* = -\mathbf{A}^{-1}(\mathbf{r} - \mathbf{qE})$$

or

$$\begin{pmatrix} N_1^* \\ N_2^* \\ N_3^* \\ \vdots \\ N_s^* \end{pmatrix} = \begin{pmatrix} a_{11} & a_{12} & a_{13} & \cdots & a_{1s} \\ a_{21} & a_{22} & a_{23} & \cdots & a_{2s} \\ a_{31} & a_{32} & a_{33} & \cdots & a_{3s} \\ \vdots & \vdots & \vdots & \ddots & \vdots \\ a_{s1} & a_{s2} & a_{s3} & \cdots & a_{ss} \end{pmatrix}^{-1} \begin{pmatrix} r_1 - q_1E_1 \\ r_2 - q_2E_2 \\ r_3 - q_3E_3 \\ \vdots \\ r_s - q_sE_s \end{pmatrix} \tag{17}$$

where $\mathbf{r} = (r_1, r_2, r_3, ..., r_s)^T$; $\mathbf{qe} = (q_1E_1, q_2E_2, q_3E_3, ..., q_sE_s)^T$ and $\mathbf{A}$ means the community matrix, the (i, j)-th elements of which is a_{ij}.

We obtained two types of solutions of fishing efforts $\mathbf{E}$ that maximize $Y(\mathbf{E})$ and $V(\mathbf{E})$ in Eq. (15) for the same food web with cost and prices, $\mathbf{A}$, $\mathbf{c}$, $\mathbf{q}$, $\mathbf{p}$, $\mathbf{r}$. We call these solutions "MSY policy" ($\mathbf{E}_{\mathrm{MSY}}$) and "maximum sustainable ecosystem service (MSES) policy" ($\mathbf{E}_{\mathrm{MSES}}$), respectively. We compare between the total ecosystem service $V(\mathbf{E})$ at MSY and $V(\mathbf{E})$ at MSES, denoted by V_{MSY} and V_{MSES}, respectively. The total yield using the MSY policy is always larger than that for the MSES solution, while the total ecosystem service for the MSES solution is always larger than that for the MSY solution.

We obtained MSY and MSES policies for 1000 randomly structured food webs of a six species food web with positive equilibrium. We assumed that the diagonal elements aii are −1 for bottom-level species (i = 1 and 2) and 0 for consumer species (i = 3 through 6). We ignored interspecific competition between species 1 and 2 ($a_{12} = a_{21} = 0$). We also assumed a_{ij} is 0 with a probability of 50% and its absolute value is between 0 and 1 with a probability of 50% for any pair of predator species i and prey species j ($j < i$). We also assumed that q_i is 1, because q and p similarly affect the optimal solution. Other parameter values are chosen by independent draws from a uniform distribution between 0 and 1 for r_i, p_1 and p_2, and between 0 and 3 for p_3 to p_6 because predators usually fetch a higher price than prey species. We ignored the cost of fishing effort, so $c_i = 0$. Finally, we assumed that S_i^{∞} and B_i are 10 and $0.2N_i^{**}$ for each species, where N_i^{**} means the equilibrium stock abundance without fisheries ($\mathbf{E} = 0$).

Figure 5 shows four typical examples of food webs. When the total ecosystem service decreases with an increasing fishing effort of any species, the MSES policy does not fish any species at all, as shown in Fig. 5(a). Among 1000 randomly connected food webs, about 85% of the webs have the same food webs with the same set of exploited species. About 15% of the webs have the same food webs with the same fishing efforts between MSY and MSES (Fig. 5(b)).

The total yield at MSES (denoted by Y_{MSES}) ranged 0–100% of the total yield at MSY (denoted by Y_{MSY}) whose average was 40%. The total ecosystem service at MSY (V_{MSY}) ranged between 21 and 100% of the total ecosystem service at MSES (V_{MSES}) whose geometric average was 69%. There was a positive relationship between $Y_{\mathrm{MSY}}/V_{\mathrm{MSES}}$ and $Y_{\mathrm{MSES}}/Y_{\mathrm{MSY}}$ (Fig. 6), while a fishing ban at MSES appeared in 1% of the 1000 food webs when the fishery yields were the smaller part of the total services ($Y_{\mathrm{MSY}}/V_{\mathrm{MSES}}$ is small) and the MSES policy was identical to the MSY policy when the fishery yield was the larger part of the total services.

All species rarely coexisted under the MSY policy for 1000 randomly structured food webs, while these six species coexist for about 91% of the webs using the MSES policy. The number of exploited species at MSES was usually much larger than the

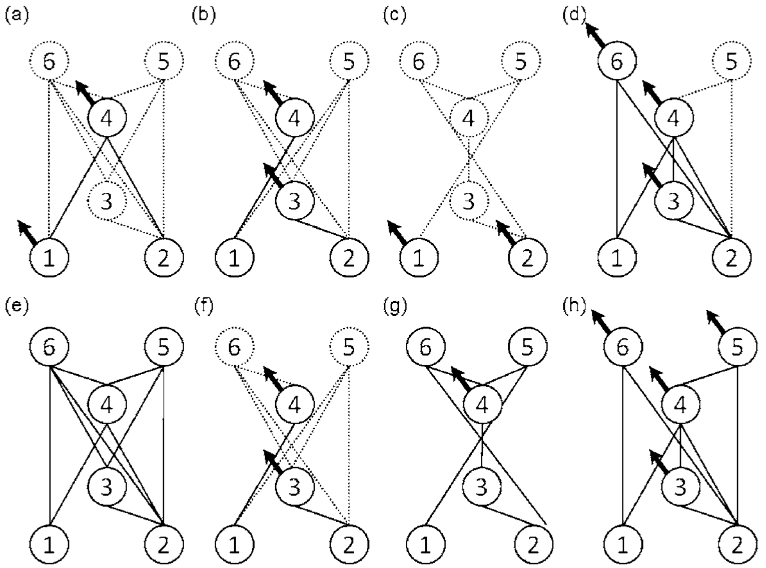

Fig. 5. Resultant food web and optimal fishing policies that maximize the total sustainable yield (panels (a)–(d)) and the total ecosystem service (panels (e)–(h)). The MSES policy for four examples of the six-species systems; circles and arrows represent species and fishery, respectively. Lines between circles represent trophic links from smaller numbered species to larger numbered species. Dotted circles mean that these species became extinct. Panels (e)–(h) are the MSES policies for four systems that are identical to panels (a)–(d), respectively.

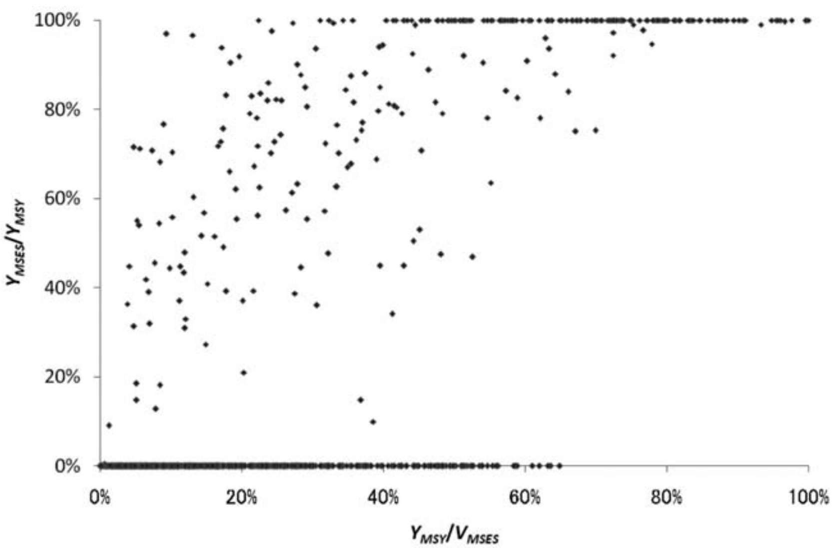

Fig. 6. The relationship between the relative ecosystem service and the relative total yield under the MSES policy from 1000 random food webs.

Table 1. Resultant food webs and fishing efforts from 1000 randomly constructed six species systems. Each column shows the frequency distribution of the numbers of extant species using MSY and MSES policies, the number of exploited species using MSY and MSES policies.

	No. of extant species at MSY	No. of exploited species at MSY	No of extant species at MSES	No of exploited species at MSES
1	31	160	0	0
2	139	828	178	201
3	48	10	431	512
4	56	0	220	256
5	18	0	24	28
6	6	0	144	0

number at MSY (Table 1). The ratio Y_{MSY}/V_{MSES} implies the ratio between the maximum sustainable yield and the maximum sustainable ecosystem service by respectively optimal policies. This ratio widely ranged between 0 and 97%. If the total ecosystem service without a fishery yield is small, the MSES policy did not guarantee the coexistence of species.

5. From Fisheries Comanagement to Ecosystem Comanagement

In order to increase the fishery yield $Y(E)$, monitoring activities of status of stock are critically important. Even more so for sustaining the utility of standing biomass $S(N)$. Government should play an important role in these monitoring activities. However, in reality, it is almost impossible for the government to monitor all the detailed ecosystems along the coast and within exclusive economic zones (EEZ). Therefore, the knowledge of fishers and data from fishery activities should be fully utilized.

As Table 1 shows, yield-maximizing fisheries are likely to take only a small number of species from one trophic level. So, under the yield-maximizing policy, fishing gear and vessel type will converge on the most efficient, and we can gain information about very limited aspects of the ecosystem. Our analyses also indicate that yield-maximizing fisheries would invoke the loss of a significant fraction of species in the web. It will inevitably lead to the degradation of $S(N)$, and easily set off the benefit from $Y(E^*)$, and ultimately reduce the total ecosystem services, $V(E)$. To avoid these situations, government has to monitor the rest of the ecosystem, and regulate the yield-maximizing fisheries in a top-down way. The reality is, again, these costs would be beyond the budget of many countries, especially developing countries. To sum up the above discussions, yield-maximizing, economically-efficient fisheries are rational for enjoying fishery rent, but not always so in sustaining total ecosystem services for society.

From the viewpoint of sustaining ecosystem services, one reasonable alternative is to conduct responsible fisheries targets for a wide range of species with a variety of gear. For example, in the Shiretoko World Natural Heritage area, local fishers have conducted various kinds of operations and caught most of the keystone species of the local ecosystem. They have accumulated the catch data of these species for over 50 years. In addition, local fishers have autonomously monitored the resource status every year, and adaptively modified the local rules for sustainable resource use. These data and knowledge is now one of the most important foundations for monitoring the changes in the ecosystem structure and functions in the heritage area. As this case shows, responsible fisheries can significantly contribute to the sustainability of ecosystem services. Like the Shiretoko case, a fisheries management approach in which government and local fishers share the responsibilities and authorities for the use of sustainable resource is called fisheries comanagement, the strongest argument against the conventional top-down approach (Makino and Matsuda 2005).

Fisheries comanagement will not always lead to the ecosystem management. As this study showed, the total ecosystem service from MSES policy is always larger than that from MSY policy, and $\mathbf{E}_{\mathrm{MSY}}$ has a tendency to be greater than $\mathbf{E}_{\mathrm{MSES}}$. In order to increase the total ecosystem services, interests from other sectors than fisheries, such as an environmental ministry or non-government organizations (NGOs), should be included in the decision-making arena (Makino 2005). Then, fisheries comanagement will evolve into an ecosystem comanagement. Under this ecosystem a comanagement framework, coexistence of small-scale, artisanal fisheries and large-scale, efficient fisheries is the rational solution for sustaining ecosystem services.

6. Discussion

In the three types of models we have analyzed in this paper, the concept of maximum sustainable ecosystem services results in a more conservative policy than MSY. A fishing ban is optimal if the ecosystem service from standing biomass is sufficiently larger than the fishery yields, despite the fact that MSES do not always prevent any species from becoming extinct. These results are intuitively understandable.

It is difficult to quantitatively evaluate the magnitude of ecosystem services. Ecosystem processes, including food web interaction, are complex and not well known. The assumptions and parameter values used in this paper should be improved in the future. Our analyses have the following problems.

First, we do not know the appropriate magnitude of ecosystem services (S_i^∞). We assumed that the ecosystem services from a standing biomass is a simple sum of the service from each species with the same magnitude of S_i^∞. Even for such simple models, we obtained a variety of MSES policies from a fishing ban of all species to the extinction of some species. The regulating services are at least 10-fold larger than fishery yield (Costanza *et al*. 1997). Our analyses suggest that a fishing ban is optimal, even when the fishery yield using the MSY policy is much more than 10% of the total ecosystem services using the MSES policy. If the regulating services do not significantly decrease through little fishing effort, a fishing ban is less likely, but a significant reduction of the standing biomass is strongly discouraged.

Second, we do not include implementation errors. This is a big problem in consensus building. Depending on the countries or areas, food supply and job creation by greater fishing efforts are considerably important factors from a social security point of view. Therefore, relative weights between ecosystem services and fishery yields might be the societal choice.

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