Environmental Conservation **36** (4): 301–311 © Foundation for Environmental Conservation 2010 doi:10.1017/S0376892910000135

A bioeconomic optimization approach for rebuilding marine communities: British Columbia case study

C. H. AINSWORTH* AND T. J. PITCHER
Fisheries Centre, University of British Columbia. 2202 Main Mall, Vancouver, British Columbia, Canada V6T 1Z4
Date submitted: 16 June 2009; Date accepted: 7 January 2010

SUMMARY

Many marine ecosystems are depleted of living resources as a result of long-term overexploitation. Restoration plans should perhaps consider the entire ecosystem as opposed to single species, yet there is currently no suitable framework available for the design and comparison of whole-ecosystem restoration trajectories. This paper presents a novel addition to Ecopath with Ecosim's policy search routine, the 'specific biomass' objective function, which allows gaming scenarios to be run using selective fishing as a tool to rebuild depleted marine ecosystems or modify them into a preferred state. In this paper, restoration scenarios aimed to restore an ecosystem in Northern British Columbia to a state similar to the historic ecosystem of 1950 AD. Restoration plans that achieve restoration quickly tend to require a large sacrifice in fishery profits, while slower plans allow for continued harvest benefits. A convex relationship between profit and recovered biodiversity suggests that there may be an optimal rate of restoration. Cost-benefit analysis demonstrates that conservative restoration plans can offer a rate of return superior to bank interest when viewed as an investment in natural capital. Increasing the selectivity of fishing gear improves the economic outlook.

Keywords: British Columbia, Ecopath with Ecosim, ecosystem approach to fisheries (EAF), ecosystem restoration, ecosystem-based management (EBM), Maxdex fleet, optimal restorable biomass (ORB), trophic models

INTRODUCTION

Overfishing has had severe impacts on target fish populations (Pauly *et al.* 1998; Christensen *et al.* 2003; Myers & Worm 2003, 2005; Halpern *et al.* 2008). In many systems, historical reconstructions conclude that contemporary fisheries achieve only a fraction of their former yields (for example see Rosenberg *et al.* 2005; Cadigan & Hutchings 2001). Although formal fisheries management plans may mandate the restoration of depleted species to some specified level, for example a level based on maximum sustainable yield, some fraction of estimated virgin biomass or other criterion, any set of goal biomasses developed by single species methods may be incommensurate in an ecosystem context (Larkin 1977, 1996; Walters *et al.* 2005). A responsible management objective might be to restore entire ecosystems using a holistic approach to management, rather than through a piecemeal application of single-species biomass goals (Campbell *et al.* 2009; Pitcher 2001).

Historic ecosystems have been suggested as potential whole-ecosystem restoration goals (Pitcher 2001), not least because (1) as past systems existed, we know their structure was compatible with thermodynamic laws, whereas an invented ecosystem structure may not be; and (2) past abundances act as cognitive guides as to what human benefits it might be possible to recover in an almost universally depleted marine realm (Pitcher 2001, 2005; Pitcher *et al.* 1999). The past may thus serve as an analogue for a restored future, which sets expectations as to the production potential, biodiversity and resilience potentially present in a healthy ecosystem. More sophisticated goals have also been evaluated; for example, ecosystems containing an optimized set of species biomasses as defined by the optimal restorable biomass concept (ORB: Ainsworth & Pitcher 2008; Pitcher *et al.* 2005). Careful consideration and debate should be given to selecting the restoration goal, which is largely a question of social priorities (Campbell *et al.* 2009). However, even if suitable goals could be established for whole-ecosystem restoration, it is doubtful whether the ecosystem can be manipulated into the desired state (Mace 2001). The work presented here offers a first step in that direction.

We introduce a new tool for ecosystem-based fisheries management (EBFM) to help identify fishing strategies that will promote a shift in the ecosystem towards a more desired state. It is a modified version of Ecopath with Ecosim's (EwE) policy search routine, which includes updated algorithms and new user interfaces (for EwE see Polovina 1984; Christensen & Pauly 1992, 1993; Walters *et al.* 1997, 2000; for the policy search routine see Christensen & Walters 2004). The principal advance we have achieved is the development of the specific biomass (SB) criterion, a flexible ecological objective function that can be used to direct the policy search routine to achieve whole ecosystem restoration, while providing users with precise control over the qualities of the restoration plan. The SB criterion replaces a much simpler

*Correspondence: Dr Cameron Ainsworth, Northwest Fisheries Science Center, 2725 Montlake Boulevard, East Seattle, WA 98113, USA Tel: +1 202 860 3289 e-mail: cameron.ainsworth@noaa.gov

ecological objective function that is present in the release version of EwE software, and can work in concert with previously described economic and social objective functions to generate a wide range of restoration scenarios.

We conducted the analysis in two parts. First, we demonstrate the effectiveness of the SB criterion by attempting to restore the historic 1950 AD ecosystem of Northern British Columbia (BC) in dynamic simulations commencing with the 2000 EwE model (Ainsworth *et al.* 2008*a*). Second, we evaluate candidate restoration scenarios with a more practical ecosystem goal, based on the ORB concept, which resembles a goal that society may wish to consider. The biomass configuration of this example goal ecosystem supports profitable fisheries (i.e. fisheries where revenues exceed costs) and maximizes an objective function based on long-term economic returns from fisheries, while prohibiting extinctions (see economic ORB in Ainsworth & Pitcher 2008). Because we structured the ORB ecosystem to provide economic returns, it is designed, once restoration is achieved, to generate more sustainable benefits than either the 2000 ecosystem or the historic 1950 ecosystem; this benefit may be used to defray the costs of restoration.

METHODS

Ecopath with Ecosim

Ecopath with Ecosim is a trophodynamic marine ecosystem simulator that partitions living and non-living components of the ecosystem into sets of similar species (functional groups). It acts as a thermodynamic accounting system, tracking the flows of biomass between groups according to a diet matrix, deducting energy spent in respiration and lost as unassimilated food (Ecopath: see Polovina 1984; Christensen & Pauly 1992; Ecosim: Walters *et al.* 1997; Appendix 1, see supplementary material at URL http://www.ncl.ac.uk/icef/EC_Supplement.htm).

Ecosim's policy search routine (Christensen & Walters 2004) permits exploration of ecological, economic and social trade-offs inherent in fisheries management policies, and optimization of fleet design to achieve policy objectives. It uses a non-linear optimization algorithm (Fletcher & Powell 1963) and iteratively adjusts fishing mortalities until the fleet-effort combination is discovered that maximizes fishery benefits (Christensen *et al.* 2005). The objective function used for the task of rebuilding here includes economic and ecosystem rebuilding criteria (Eq. 1).

$$\mathrm{OBJ} = \mathrm{W}_{\mathrm{ECON}} \cdot \frac{\left(\sum \mathrm{NPV}_{\mathrm{ij}}\right)_{current}}{\left(\sum \mathrm{NPV}_{\mathrm{ij}}\right)_{baseline}} + \mathrm{W}_{\mathrm{REB}} \cdot \frac{\left(\sum SB_{\mathrm{it}}\right)_{current}}{\left(\sum SB_{\mathrm{it}}\right)_{baseline}} \tag{1}$$

W_{ECON} and W_{REB} are relative weighting factors for economic and ecosystem rebuilding. The summed terms represent net present value (NPV) of the harvest plan for each functional group (i) and gear type (j) and the ecosystem rebuilding benefits measured using the SB index for each functional group and simulation time step (t). NPV summarizes the expected stream of profits, discounting benefits far off in the future using an intergenerational discounting term (Sumaila 2001, 2004; Sumaila & Walters 2005) at conservative discount rates for the current generation ($\delta = 4\%$) and future generations ($\delta_{\mathrm{fg}} = 10\%$). We used the intergenerational discounting approach because it is more suitable for long-term fisheries conservation than the conventional method of discounting (Ainsworth & Sumaila 2005).

Northern British Columbia models

This paper employs previously published EwE models for Northern British Columbia (BC) for the years 1950 and 2000 (for full details of these see Ainsworth 2006 and Ainsworth *et al.* 2008*a*). Briefly, the 1950 and 2000 models use a common structure. Fifty-three functional groups describe the ecosystem, and 11 juvenile/adult split pools represent trophic ontogeny in commercially important fish species. Basal species tend to be aggregated, while fishery targets and ecologically important species are modelled in more detail. Biomass data for 1950 and 2000 comes from Department of Fisheries and Oceans (DFO) reports, literature sources and community interviews (Ainsworth 2006). Catch data comes mainly from historical catch records and includes estimates of illegal, unreported and unregulated catch (Ainsworth & Pitcher 2005). Model basic parameters were evaluated using a sensitivity analysis, and the 1950 simulation was fitted to time series. Trophic vulnerabilities (species interaction rate parameters) were calibrated for the 1950 model and adapted to the 2000 model using a novel procedure that assumes stationarity in species foraging tactics (Ainsworth *et al.* 2008*a*).

Fishing fleets

Larkin (1996) said, 'Existing fleets are a blunt instrument for fine tuning the relative abundance of species'. Where gear types catch multiple species, this limits the ability to structure the ecosystem for human benefit. For example, one depleted species cannot be selected for rebuilding while maintaining high levels of catch on sympatric species using unselective gear because high levels of bycatch could sabotage rebuilding efforts. Here, we refer to a hypothetical fishing fleet that pursues one species group per sector as the maximum dexterity or *maxdex* fishing fleet. The maxdex fleet is used to project a 'best case' restoration scenario, where precise manipulation of species biomasses is possible and the goal ecosystem configuration is not limited by the selectivity of fishing gear, only by the ecological interplay of species. This sets the benchmark for restoration and helps evaluation of the effectiveness of various candidate fleets to be used for the task of rebuilding.

We use two other fleets in the rebuilding scenarios here. The first is a representation of the Northern BC fleet *c.* 2000 (described in Ainsworth *et al.* 2008*a*); the second is a 'next-generation' fishing fleet designed with ecologically and socially responsible criteria in mind. We refer to this as the *lost valley* fishing fleet (after Pitcher *et al.* 2004; described for Northern BC by Ainsworth *et al.* 2004). The lost valley fleet reduces bycatch and collateral habitat damage to within technologically achievable limits, and adheres, unlike most world fisheries today (Pitcher *et al.* 2009), to the FAO (Food and Agriculture Organization of the United Nations) Code of Conduct for Responsible Fisheries (FAO 1995).

A new restoration objective function

The distributed version of EwE V5.1 contains several objective functions representing economic, social and ecological benefits. It contains another standard objective function to rebuild biomass of depleted species, which is called 'mandated rebuilding' (Christensen *et al.* 2005). However, that objective function is suitable only when a single species group is mandated for recovery and is not adequate when multiple species groups are to be restored simultaneously. It recognizes improvement only through biomass increases of groups, whereas selective declines may be needed to facilitate growth in species through direct or indirect trophic interactions. The 'mandated rebuilding' routine also assumes that restoration is equally desirable in all groups, which makes prioritization impossible.

The specific biomass (SB) criterion is an objective function designed to make any prioritization between species groups explicit, and to give the user more control over the trade-offs inherent in ecosystem restoration. The algorithm used to compute the new SB function essentially governs the trade-off between restoring biomass of a large number of easily recoverable groups, versus a smaller number of critical 'problem' groups. The latter are groups that are slow to increase in biomass (such as slow-growing groups) or where biomass reductions are difficult to achieve (such as groups not subject to direct harvests). The SB function uses two main variables to define the desired trade-off: the unit of ecosystem 'improvement' towards goal biomass and the choice of model used to calculate the value of marginal improvement towards the goal.

The SB criterion evaluates the biomass of species groups flagged by the user for rebuilding. The difference between starting biomass, when $t = 0$, and the goal biomass for restoration defines the potential scope for improvement. At each time step, group biomass is employed to calculate an internal term (θ) representing the proximity of the group's biomass to the rebuilding goal, where θ is a unitless multiple of the initial biomass differential between the initial and goal biomass. Initially, $\theta = 0$ if starting biomass is less than the goal and $\theta = 2$ if starting biomass is greater than the goal. As group biomass increases or decreases towards the goal, θ will approach 1. If starting biomass equals goal biomass, then $\theta_{start} = 1$. At each simulation time step, proximity to goal (θ) is calculated (Eq. 2):

$$\theta = \begin{cases} \dfrac{B_{current} - B_{start}}{B_{goal} - B_{start}} & \text{if } B_{start} < B_{goal} \\ 2 - \left[\dfrac{B_{current} - B_{start}}{B_{goal} - B_{start}}\right] & \text{if } B_{start} > B_{goal} \end{cases} \tag{2}$$

$B_{current}$ is the functional group's biomass at time step t, B_{goal} is the goal biomass and B_{start} is baseline biomass when $t = 0$. Each group's contribution to the SB index is calculated based on θ; groups will contribute their maximum to the objective function when $\theta = 1$.

If we define the unit of ecosystem 'improvement' towards the goal strictly as a change in biomass (hereafter called the biomass criterion), then in depleted ecosystems the search will tend to advocate fishing strategies that greatly reduce fishing mortality from baseline levels. If we define improvement in terms of a per cent change towards target (hereafter called the per cent criterion), which will cause a larger number of groups to approach their targets, but less overall change in system biomass. Under the per cent criterion, proximity to target (θ) is passed directly to the marginal improvement model as $\theta_{per\ cent}$ for each functional group. If the unit of improvement is biomass, then $\theta_{biomass}$ is first calculated (Eq. 3). A combined term ($\theta_{combined}$) can also be used for mid-range solutions (Eq. 4).

$$\theta_{biomass} = \theta \cdot \left|B_{goal} - B_{start}\right| \tag{3}$$

$$\theta_{combined} = X \cdot \theta_{biomass} + (1 - X) \cdot \theta_{percent} \tag{4}$$

X is a weighting factor between 0 and 1.

The marginal improvement valuation model allows the user to weigh the relative contribution of a functional group to the objective function SB according to the group's current distance from the goal biomass. Under the linear valuation model (Fig. 1*a*; Eq. 5), all species groups are weighted equally in the calculation regardless of their distance to target.

$$SB_{lin} = \begin{cases} \theta & \text{if } B_{start} < B_{goal} \\ -\theta + 2 & \text{if } B_{start} > B_{goal} \end{cases} \tag{5}$$

The linear model will make whatever trade-offs are necessary to reduce biomass residuals versus the desired ecosystem configuration, minimizing $\sum_{it} |\theta_{it} - 1|$ for each functional group i and time step t.

Under the quadratic model option (Fig. 1*b*; Eq. 6), the greatest marginal increase in the objective function occurs when groups first begin to move towards their goal biomass. More groups will improve in the optimal fishing policies than under the linear model, but the average improvement in the proximity function θ will be lower. The quadratic model is precautionary because the objective function decreases rapidly as group biomasses drift away from their goals in either positive or negative direction.

$$SB_{quad} = -\theta^2 + 2\theta \tag{6}$$

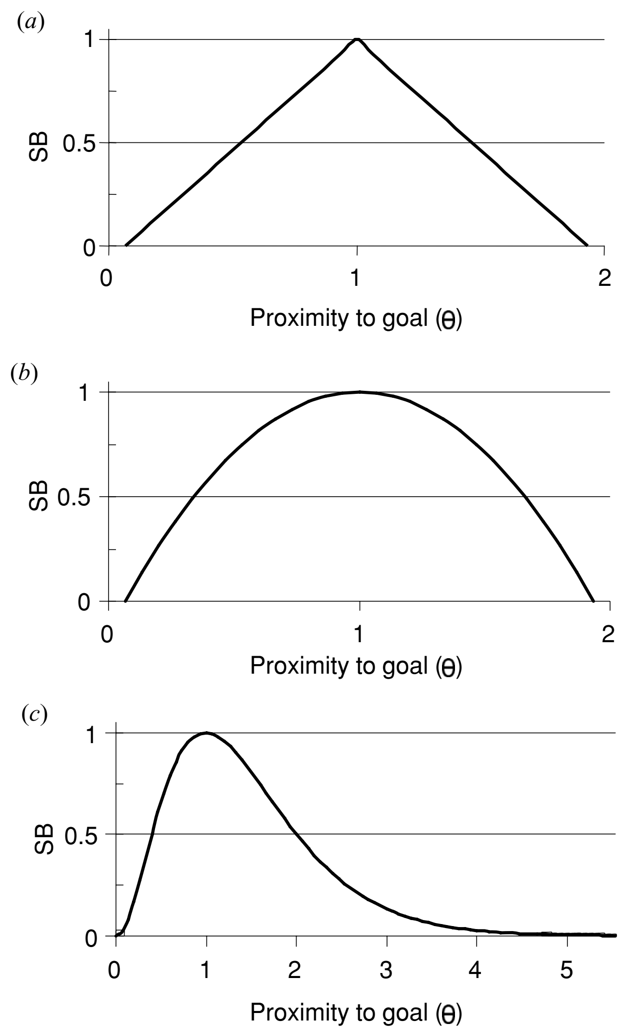

Figure 1 Three models describing marginal improvement in SB function. The initial ecosystem condition is taken as $\theta = 0$ if start biomass is less than goal, or $\theta = 2$ if start biomass is greater than goal. As the functional group approaches its goal biomass, θ approaches 1. (*a*) Linear model weighs a unit of improvement the same, regardless of the current biomass differential versus goal. (*b*) Quadratic model gives the greatest improvement to objective function when groups first begin to move towards target. (*c*) Asymmetric gamma model is precautionary; biomass increase towards target is more valuable to the objective function than biomass decrease.

The gamma valuation option determines marginal improvement based on a gamma function (Fig. 1*c*; Eq. 7). Because it is asymmetric, improvement from functional group biomass growth contributes more to the objective function than improvement from group biomass decreases; it is therefore the most precautionary option. Optimal policies will more often overshoot target biomasses than fall short.

$$SB_{gam} = \phi \cdot \frac{\left(\frac{\theta}{\beta}\right)^{\gamma-1} \cdot e^{-\left(\frac{\theta}{\beta}\right)}}{\beta\Gamma(\gamma)}$$

$$\text{where} : \Gamma(\gamma) = \int_0^\infty t^{\gamma-1} e^{-t} dt \qquad (7)$$

Phi (ϕ) is a scaling term for the Y-axis, (γ) is the shape parameter and (β) is a scaling term for the gamma function (Γ)). The gamma model provides excellent flexibility to define the shape of the valuation curve. The model response is standardized according to arbitrary but interpretable values. Parameters are set so that the gamma objective function is worth 0.5 when proximity to goal (θ) is equal to 0.5 or 2.0. In other words, a functional group will contribute the same to the SB objective function when it is 50% short of its goal biomass as when it is 100% in excess of its goal biomass, and that value corresponds to 50% of the maximum possible contribution for that group towards the objective function. Parameters used to set this relationship were $\phi = 1.74$, $\gamma = 3.21$ and $\beta = 0.45$.

We developed this new policy search tool as a compiled executable replacing Ecopath.exe, available from us on request. Some auxiliary parameters are accessible on the new interface, such as an 'extinction threshold', which allows the user to specify a minimum acceptable functional group biomass for the restoration plan, as a per cent of initial biomass. All harvest plans we evaluated employed a 5% depletion threshold, the default value in the new interface.

Restoration case study

Using the new SB objective function, we designed a restoration scenario that would convert the Northern BC ecosystem (*c.* 2000) to a more productive state. The goal ecosystem configuration was based on the 1950 ecosystem, optimized for long-term harvest benefits following the optimal restorable biomass (ORB) concept (Ainsworth & Pitcher 2008; Pitcher *et al.* 2005). Briefly, ORB is the equilibrium ecosystem condition that would result from fishing a historic ecosystem responsibly for a long period of time. The species composition in the ecosystem is adjusted, through application of an optimal fishery programme (Ainsworth 2006), to support the largest sustainable fishery profits at equilibrium while disallowing extinctions. We restricted the analysis to restoration plans targeting this economic ORB ecosystem since it proved to be a more competitive economic goal than the historical 1950 ecosystem and should provide a better return on the investment of restoration. Each restoration scenario is directed according to a multi-criterion objective that contains varying weights on economic and rebuilding (SB) objectives (Eq. 4).

Cost-benefit analysis

The cost-benefit analysis of the whole restoration plan can be divided into three stages (Fig. 2). In the restoration phase (α), optimal fishing mortalities estimated by the SB algorithm reduce harvest rates from the status quo level and allow the ecosystem to rebuild biomass of targeted species. The profit sacrificed from the baseline level (status quo fisheries) is considered the 'cost' of restoration, as these yields would

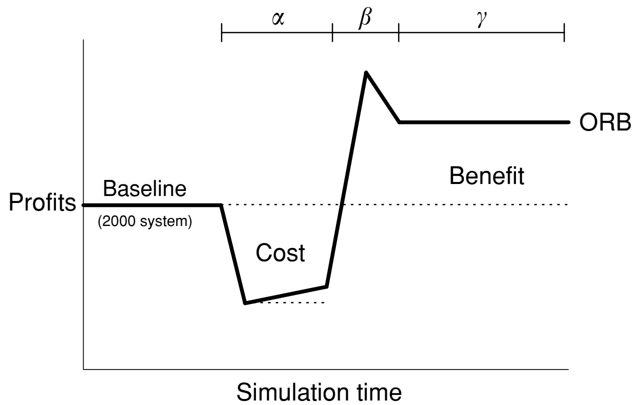

Figure 2 Conceptual diagram showing cost-benefit analysis. Restoration plan consists of three phases: Optimal fishing phase for ecosystem rebuilding (α); readjustment of fishing effort to establish new equilibrium and cancel biomass accumulations (β); sustainable fishing at new profit equilibrium (γ). Costs and benefits of restoration are taken in relation to forecasted profits from the initial ecosystem, assuming status quo profit.

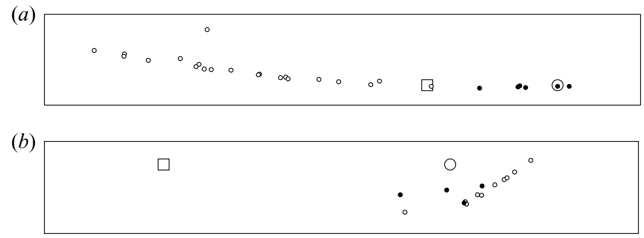

Figure 3 Principal component analysis showing ecosystem configurations after restoration. Large square represents the initial ecosystem configuration (2000); large circle represents the goal configuration (historic 1950 ecosystem); small closed circles show harvest simulations optimized for biomass recovery (unit of improvement is biomass); small open circles show additional simulations (per cent and mixed unit of improvement). (*a*) All groups mandated for restoration. (*b*) Commercial groups mandated only. Plots have been rotated to reveal restoration success along the x-axis.

otherwise have been available to resource users. The second phase is transitional (β); fishing effort is adjusted to cancel any remaining biomass accumulations. In the third phase (γ), final equilibrium harvest levels are maintained until the end of the evaluated time horizon.

Exploitable biomass increases with restoration so fisheries can draw more profit sustainably from the restored ecosystem than from the original ecosystem. The difference is called the economic ‘benefit’ of restoration. We have assumed completely malleable fishing capital; there is no penalty associated with fleet restructuring, although such costs could be included where they can be estimated. Fixed and variable costs of fishing and catch value are specified by gear type (Ainsworth 2006).

Here, the restoration process begins in 2000. At the end of the restoration phase (α), which lasts 5, 7.5, 10, 15, 20 or 30 years, a new static model based on a particular time step in the simulation (in this case, the last year of the restoration phase) is created using the EII export/import procedure in Ecosim (Christensen *et al*. 2005). We call this the transition model. In a few cases, minor changes were made to the transition models to recreate mass-balance (for example we cancelled residual biomass accumulations and adjusted predation mortality) but these had minimal effects on the species biomass values at equilibrium. During the transition phase (β), we cancelled biomass accumulations out of the transition model using another optimal fishing policy directed by the SB algorithm, which sought to hold biomasses at constant levels corresponding to the end of the restoration phase. The transition phase lasts 20 years. Harvesting then continues at the restored profit equilibrium throughout the equilibrium harvest phase (γ), which we extended to 100 years to evaluate benefits over several human generations.

RESULTS

Algorithm diagnostics

We tested the performance of the SB algorithm in achieving an ecosystem target, the historic 1950 ecosystem. The biomass criterion for unit of improvement allows a greater overall change in functional group biomass towards the target configuration than either the ‘mixed’ or ‘per cent’ unit of improvement. It allows rebuilding of heavily depleted species groups. The per cent criterion results in a more even improvement among species groups.

The linear marginal improvement valuation model produced the greatest overall change in the ecosystem, indicated by a low sum of squares versus the target configuration. The quadratic and gamma valuation models were unable to restructure the ecosystem to match the target configuration as precisely. The average per cent improvement across species groups was approximately the same for the linear and quadratic valuation models; however, the gamma model allowed more groups to exceed their target biomasses, and therefore errs on the side of caution.

The SB algorithm was able to reconfigure the ecosystem to resemble a specific goal, in this case the historic 1950 ecosystem (Fig. 3). Principal component analysis (PCA) summarized the similarity of the end-state functional group biomass vector with the target ecosystem configuration after a 50-year restoration plan. Effective ecosystem reconfiguration was possible when the biomasses of all species groups were mandated for adjustment (Fig. 3*a*). The PCA recognized changes in biomass so only plans optimized under the biomass unit of improvement achieved results. When fewer groups were mandated for restoration, the distinction between the biomass and per cent units of improvement became less important since the mode of action was similar for optimal fishing policies using both metrics. Fishing solutions therefore converged (Fig. 3*b*).

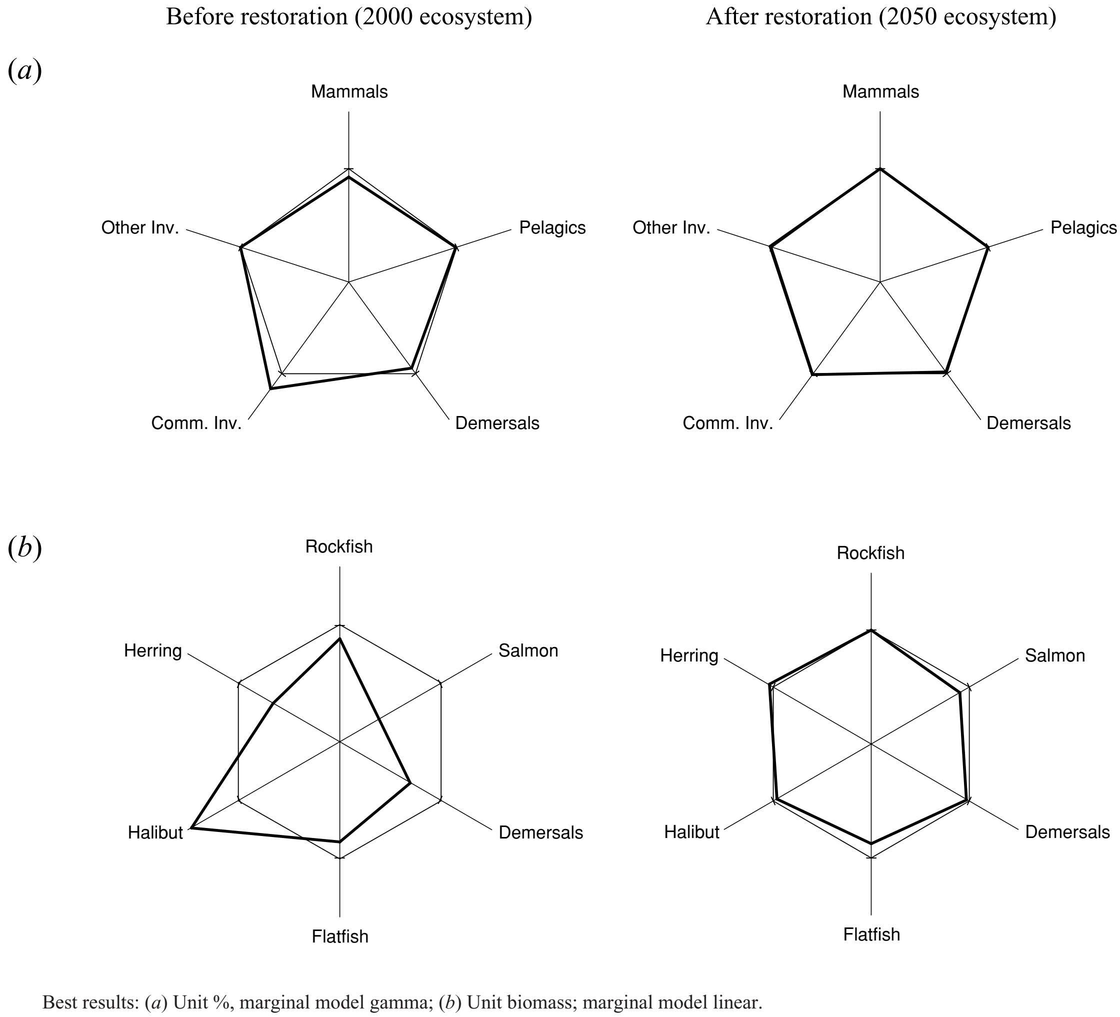

Figure 4 End-state group biomass after rebuilding relative to target 1950 goal biomass. Thin line shows 1950 goal biomass (defined as 1); thick line shows initial 2000 biomass (left) and restoration end-state biomass (right). (*a*) All species groups mandated for restoration. (*b*) Commercial species groups mandated only. Biomass values are aggregated across species groups. Radial scale is linear. Comm. Inv. = commercial invertebrates, Other Inv. = other invertebrates.

The end-state functional group biomasses following restoration lay very close to their target levels of the historical 1950 ecosystem (Fig. 4). When only commercial groups were mandated for recovery, biomass closely approached the target biomass. Fewer conflicting dynamics needed to be resolved in an optimization concerning fewer groups. Of the eleven commercial functional groups mandated for recovery, the restoration process brought biomass levels to within 15% of target, on average. When all groups were mandated for recovery, groups approached within 25% of target biomass on average.

Cost-benefit analysis of restoration

Restoration scenarios that were directed entirely by the SB objective ($W_{REB} = 1$, $W_{ECON} = 0$; Eq. 4) achieved the best restoration possible without concern for economic costs and benefits. However, by incrementing the relative weight of the economic objective, we produced a spread of restoration plans that placed increasing emphasis on the profitability of fisheries throughout the restoration process (Fig. 5). All of the fishing policies achieved a prescribed level of restoration measured as a reduction in model residuals versus the goal configuration (Fig. 5). Plans with a strong economic directive concentrated rebuilding efforts on the most valuable species groups (Fig. 5, top left), while plans with less concern for harvest benefits affected a greater number of groups resulting in higher end-state biodiversity. Biodiversity was not an explicit objective of this policy search, although that facility exists (see Kempton's Q index modified for EwE; Ainsworth & Pitcher 2006). Nevertheless, biodiversity increased as the ecosystem approached the goal. There was a convex relationship between profit and biodiversity, so restoration scenarios that left the ecosystem in the middle of this range achieved an advantageous compromise.

NPV of restoration to the economic ORB ecosystem using the lost valley fleet and the BC fleet rebuilt the

https://doi.org/10.1017/S0376892910000135 Published online by Cambridge University Press

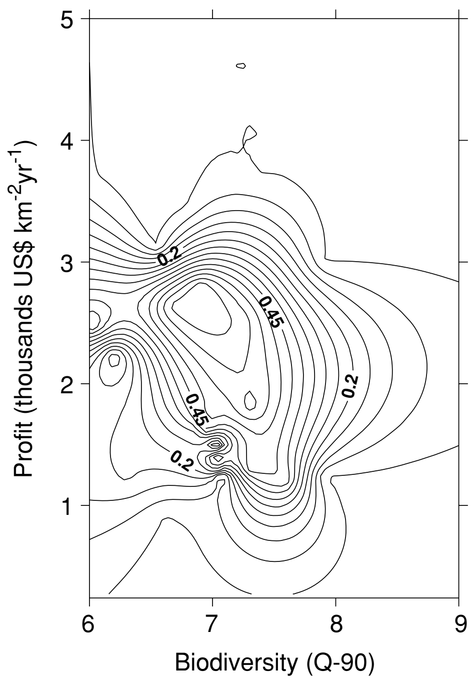

Figure 5 End-state profit and biodiversity of restoration plans ($n = 39$) targeting the historic 1950 ecosystem based on random initialization of fishing effort. The optimal equilibriums were calculated using various weightings in the objective function for ecosystem restoration and economic performance. All species groups are considered in the restoration objective; plans use biomass as the unit of improvement, employ a linear marginal improvement valuation model and apply the maxdex fishing fleet. Contours show reduction in sum of squares; peak represents 65% reduction. Initial ecosystem values: profit \$480·km^{-2}·yr^{-1}; biodiversity 7.4.

Figure 6 NPV of restoration plans achieving a minimum reduction in residuals versus goal ecosystem. Each scenario includes a restoration and harvest phase. Discount rate ($\delta = 5\%$. Target ecosystem is the 1950 economic ORB. Fleets used are: BC fleet (closed circles) and the lost valley fleet (open circles). Criterion for successful restoration is a 20% reduction in the sum of squares versus commercial species groups; this level was chosen to provide a maximum spread in points along the x-axis. Plans optimized for restoration achieve the SS reduction criterion quickly, but plans that include an economic objective achieve restoration more slowly and maintain higher profit during the rebuilding phase.

ecosystem to a minimum specified degree measured as a reduction in residuals versus target (Fig. 6). Scenarios used a spread of weightings on the economic and SB rebuilding objectives. When an economic objective was included, the rebuilding took longer to achieve the specified reduction in residuals but produced more fishery profits annually (Fig. 6). Costs of restoration were spread out over time. NPV was calculated assuming a standard discount rate of $\delta = 5\%$, similar to long-term bank interest (Government of Canada benchmark bond yielded 4.29% in April 2007, see URL http://www.bankofcanada.ca; Fig. 6). When NPV = 0, scenarios involving the lost valley fleet, which has less bycatch than the BC fleet and is able to fine-tune biomass proportions more precisely and with less waste, outperformed bank interest as an investment in natural capital (Fig. 6).

Varying restoration time

The NPV of restoration plans that targeted the economic ORB ecosystem using the BC fleet and the lost valley fleet included various weightings on the economic objective and the SB rebuilding objective, but all achieved some degree of improvement versus the target ecosystem according to a least squares criterion (Fig. 7). Longer restorations plans were able to achieve a closer match to the target ORB system, and so could deliver a greater sustained profit once the new equilibrium is reached. Using these simple equilibrium level optimal fishing mortalities to drive the simulations, most of the restoration benefit occurred within the first 15–20 years; the system then reached 'restored' biomass equilibrium according to the imposed optimal fleet-effort pattern, and there was little benefit in extending the restoration phase (α). Even with the unmodified BC fleet, conservative restoration plans outperformed status quo profit, but the lost valley fleet, which had less bycatch, offered the most cost-effective restoration scenarios.

DISCUSSION

Ecological limits to restoration

We have demonstrated application of the new restoration tool using restoration goals based on historic ecosystems. Historical states may be suitable objectives for ecological and social reasons, but many additional factors must be considered before any real-world application. Chronic impacts from overfishing may have compromised the ability of the ecosystem to recover in the short term. If trophic energy flows up the food web have become more linear and simplified because of fishing (Pauly *et al*. 1998, 2002), then the energy budget of coastal marine ecosystems may no longer support a broad diversity of specialized predators. In addition, if directional climate

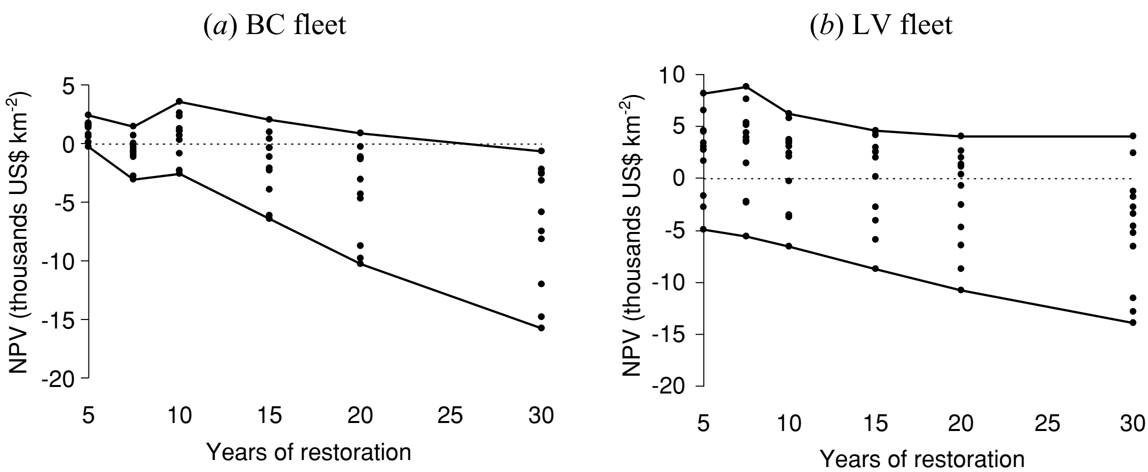

Figure 7 Net present value of restoration scenarios for (*a*) BC fleet and (*b*) lost valley fleet. Goal ecosystem is economic ORB. Restoration scenarios last 100 years and include a rebuilding phase (α) of 5, 7.5, 10, 15, 20 or 30 years and an equilibrium harvest phase lasting until year 100. The profit stream is evaluated using a 5% discount rate. Broken line indicates status quo NPV (profits in year 2000 extended to year 100). Optimizations use biomass as the unit of improvement, and employ a linear marginal improvement valuation model, which provides the best conditions for profitability. Various weightings on economic and SB criteria provide a spread of possible restoration plans. The vertical spread of points along the Y-axis consists of runs optimized for economic returns at top, and runs optimized for ecosystem rebuilding at bottom.

change has occurred since historic times, then restoration policies directed towards historic states will be fighting a natural shift in the assemblage. Concerns have also been raised regarding the reduced fitness of populations owing to founder effects, evolutionary change in response to fishing, irreversible species introductions and persistent changes in environmental regimes (such as nutrients or hydrography) (Pitcher 2005). Extinction problems are also recognized, as is the loss of locally adapted populations and keystone species (Pitcher 2005). These issues present tangible ecological and environmental limits to restoration, some insurmountable, thus we have restricted the present study to scenarios targeting restoration goals based on relatively recent historical conditions. A trophic ecosystem modelling approach may evaluate many of these issues, but additional modelling tools are needed.

At the time of this study, the EwE policy search routine was limited to determining only equilibrium-level optimal fishing mortalities owing to an error in the way that parameters were scaled when estimating the numerical derivatives in the Fletcher-Powell optimization routine (C. Walters, UBC Fisheries Centre, personal communication 2006). All the optimizations we conducted therefore used a 'single block' optimization, namely a single fishing mortality assigned to each gear type and held constant throughout all simulation years. Future work could explore the use of complex multi-staged restoration plans. Non-linear and hysteretic change may prevent a complex marine system from reverting to its wilderness state once fishing pressure is removed (Scheffer *et al*. 2001; Hughes *et al*. 2005; Duarte *et al*. 2009). However, the SB algorithm may identify complex restoration schemes, and advance an alternative stable state desirable from a policy perspective.

If fisheries activity or food web dynamics maintain a current persistent state, then we might develop recovery scenarios that use direct and indirect trophic effects to promote changes in the assemblage; for example by suppressing predators whose compensatory feeding habits help maintain current ecosystem structure. However, changes in nutrification, flow patterns or other external environmental factors (as may be induced by humans or as the result of natural variation) could sabotage restoration efforts through separate and synergistic activity (Duarte *et al*. 2009). Such environmental factors may be incorporated into EwE through use of simple time-forcing functions, but calibrating the system response against climate models (as per the Intergovernmental Panel on Climate Change scenarios) would require special care. A preliminary effort made with the current model confirmed non-linear dynamics in fisheries value and species biomass with decreasing nutrient availability. However, model-coupling facilities in EwE (version 6) will greatly increase our ability to address these non-linear dynamics: to place confidence limits on predictions of ecosystem structure following restoration, or even incorporate climate effects explicitly in the policy optimization yielding more precautionary management targets.

With these new tools, it may be possible to test regime-shift hypotheses that explain the recovery failure of Atlantic Northern cod (for example relating to environmental changes, mammal predation or poaching/bycatch: Rice & Rivard 2003; or depensation effects: Walters & Kitchell 2001). In the case of northern BC, a pre-industrial assemblage may only be achieved through careful alteration of keystone groups and maintenance of key trophic interactions, such as the sea otter-kelp-urchin triad (Estes & Duggins 1995; Steneck *et al*. 2002). We can expect idiosyncrasies specific to each ecosystem (Duarte *et al*. 2009) and there are clearly large uncertainties involved, some of which can be evaluated in the EwE framework. In the BC example, uncertainties surrounding the density dependent feeding rates of sea otters and urchins can be evaluated (after Mackinson *et al*. 2003), as can dependence of juvenile fish on biogenic structure (see for example Okey *et al*. 2004; Ainsworth *et al*. 2008*b*). However, from a policy perspective, scientific uncertainty may not be sufficient grounds to delay restoration attempts; successful reintroductions of sea otters have already occurred in BC and elsewhere in the north-east Pacific (Riedman & Estes 1998; Lance *et al*. 2004).

Cost-benefit analysis

There is potential to improve the bioeconomic model used by the policy search routine in this paper; for instance, the malleability of fishing capital should be carefully considered. Some fisheries scientists view overcapacity and overcapitalization as the single greatest threat to the long-term viability of fish stocks (Mace 1997; Gréboval & Munro 1999; Ward *et al*. 2001). However, here we assume that there is no cost or penalty associated with fleet restructuring or decommissioning. Ongoing developments in the EwE framework, such as a recent effort to integrate a new fleet buy-back scheme into the optimization procedure (Cheung & Sumaila 2008), may permit better assessments as to the economic viability of whole ecosystem restoration.

In the economic analysis here, costs and benefits in each year were considered in relation to the status quo profit, which was the estimated fisheries profit for 2000. The profit, we assume, will remain constant over the next 50 years. The implicit assumption is that the levels of fishing mortality in northern BC are sustainable. There may be reason to doubt this where declining species are concerned, notably rockfish and other valuable demersal fish. We are therefore using a generous estimate of baseline profit. If the current level of profit actually declines over the next 50 years, the benefits of restoration would have been underestimated here, and consequently the NPV of the restoration plan.

The rebuilding plans (Fig. 6) allow some level of fishing to occur throughout the rebuilding process. This important feature could make restoration more socially acceptable to stakeholders than an approach in which fishing pressure is completely eliminated to allow quick rebuilding (see Clark & Munro 1975) since we are not strictly eliminating fishing effort in these rebuilding plans. We are instead using selective fishing as a tool, applying fishing effort for example to remove competitors of depressed species and facilitate the growth of prey. Strategic application of fishing effort should out-perform a blanket policy of fishing cessation, and achieve faster and more economical growth of desired species (Ainsworth 2006). Moreover, economic externalities may slow down the optimal rate of rebuilding in real applications, as would the encumbering species interactions that can now be explicitly managed using this new EBFM tool.

CONCLUSIONS

Development of tools to predict marine ecosystem dynamics continues to advance the science behind restoration ecology, but the level of coordination that must be achieved among all stakeholders, and even among marine scientists, may prove to be a greater challenge to the restoration agenda than technical requirements. Ecosystem restoration is comprehensive by definition, and restoration policies will necessarily enlist many industrial and scientific partners into the process of restoring the natural system. Scientists from diverse disciplines such as ecology, sociology and economics will need to work together to breakdown barriers that have so far divided marine science into disjointed sectors (Rosenberg & McLeod 2005). Industry especially must be committed to the task, since any tenable restoration plan is likely to impact the short-term profitability of fisheries in favour of long-term sustainability. In this respect, scenarios that allow profit during restoration might more politically palatable.

ACKNOWLEDGEMENTS

We gratefully acknowledge the following people for helpful comments on the specific biomass objective function and implementation in EwE: Villy Christensen, Carl Walters, Daniel Pauly, Rashid Sumaila (UBC Fisheries Centre), Les Lavkulich (UBC Institute for Resources, Environment and Sustainability) and Alan Sinclair (Department of Fisheries and Oceans).

References

Ainsworth, C.H. (2006) Strategic marine ecosystem restoration in northern British Columbia. Ph.D. thesis. Resource Management and Environmental Studies, University of British Columbia, Canada: 423 pp.

Ainsworth, C.H. & Pitcher, T.J. (2005) Estimating illegal, unreported and unregulated catch in British Columbia's marine fisheries. *Fisheries Research* **75**: 40–55.

Ainsworth, C.H. & Pitcher, T.J. (2006) Modifying Kempton's species diversity index for use with ecosystem simulation models. *Ecological Indicators* **6**(3): 623–630.

Ainsworth, C.H. & Pitcher, T.J. (2008) Back to the future in northern British Columbia: evaluating historic marine ecosystems and optimal restorable biomass as restoration goals for the future. In: *Reconciling Fisheries with Conservation: Proceedings of the Fourth World Fisheries Congress*, ed. J.L. Nielsen, J.J. Dodson, K. Friedland, T.R. Hamon, J. Musick & E. Verspoor, pp. 317–329. Bethesda, Maryland, USA: American Fisheries Society, Symposium 49: 1946 pp.

Ainsworth, C.H. & Sumaila, U.R. (2005) Intergenerational valuation of fisheries resources can justify long-term conservation: a case study in Atlantic cod (*Gadus morhua*). *Canadian Journal of Fisheries and Aquatic Sciences* **62**: 1104–1110.

Ainsworth, C.H., Heymans, J.J. & Pitcher, T.J. (2004) Policy search methods for back to the future. In: *Back to the Future: Advances in Methodology for Modeling and Evaluating Past Ecosystems. Fisheries Centre Research Reports 12(1)*, ed. T.J. Pitcher, pp. 48–63. Vancouver, Canada: Fisheries Center, University of British Columbia: 158 pp.

Ainsworth, C.H., Pitcher, T.J., Heymans, J.J. & Vasconcellos, M. (2008*a*) Reconstructing historical marine ecosystems using food web models: Northern British Columbia from pre-European contact to present. *Ecological Modelling* **216**: 354–368.

Ainsworth, C.H., Varkey, D. & Pitcher, T.J. (2008*b*) Ecosystem simulations supporting ecosystem-based fisheries management in the Coral Triangle, Indonesia. *Ecological Modelling* **214**(2–4): 361–374.

Cadigan, S.T. & Hutchings, J.A. (2001) Nineteenth-century expansion of the Newfoundland fishery for Atlantic cod: an exploration of underlying causes. In: *The Exploited Seas: New Directions for Marine Environmental History*, ed. P. Holm, T.D. Smith & D.J. Starkey, pp. 31–65. St John's, Newfoundland, Canada: International Maritime Economic History Association and Census of Marine Life.

Campbell, L.M., Gray, N.J., Hazen, E.L. & Shackeroff, J.M. (2009) Beyond baselines: rethinking priorities for ocean conservation. *Ecology and Society* **14**(1): 1–14.

Cheung, W.W.L. & Sumaila, U.R. (2008) Trade-offs between conservation and socio-economic objectives in managing a tropical marine ecosystem. *Ecological Economics* **66**(1): 193–210.

Christensen, V. & Pauly, D. (1992) ECOPATH II. A software for balancing steady-state models and calculating network characteristics. *Ecological Modeling* **61**: 169–185.

Christensen, V. & Pauly, D., eds. (1993) Flow characteristics of aquatic ecosystems. In: *Trophic Models of Aquatic Ecosystems. ICLARM Conference Proceedings 26*, ed. V. Christensen & D. Pauly, pp. 338–352. Manila, Philippines: International Center for Living Aquatic Resource Management: 390 pp.

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

Christensen, V., Walters, C.J. & Pauly, D. (2005) *Ecopath with Ecosim: a User's Guide*. November 2005 edition. Vancouver, Canada: Fisheries Centre, University of British Columbia: 154 pp. [www document]. URL http://ecopath.org/modules/Support/Helpfile/EweUserGuide51.pdf

Christensen, V., Guénette, S., Heymans, J.J., Walters, C.J., Watson, R., Zeller, D. & Pauly, D. (2003) Hundred-year decline of North Atlantic predatory fishes. *Fish and Fisheries* **4**: 1–24.

Clark, W.G. & Munro, G.R. (1975) Economics of fishing and modern capital theory: a simplified approach. *Journal of Environmental Economics and Management* **2**: 92–106.

Duarte, C.M., Conley, D.J., Carstensen, J. & Sánchez-Camacho, M. (2009) Return to Neverland: shifting baselines affect eutrophication restoration targets. *Estuaries and Coasts* **32**: 29–36.

Estes, J.A. & Duggins, D.O. (1995) Sea otters and kelp forests in Alaska: generality and variation in a community ecological paradigm. *Ecological Monographs* **65**(1): 75–100.

FAO (1995) The code of conduct for responsible fisheries. Food and Agriculture Organization of the United Nations. Adopted by the Twenty-eighth session of the FAO Conference in October 1995 [www document]. URL http://www.fao.org/DOCREP/005/v9878e/v9878e00.htm

Fletcher, R. & Powell, M.J.D. (1963) A rapidly convergent descent method for minimization. *The Computer Journal* **6**: 163–168.

Gréboval, D. & Munro, G.R. (1999) Overcapitalization and excess capacity in world fisheries: underlying economics and methods of control. In: *Managing Fishing Capacity: Selected Papers on Underlying Concepts and Issues*, pp. 1–48. Food and Agriculture Organization of the UN, FAO Fisheries Technical Paper No. 286. Rome, Italy: FAO.

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R. & Watson, R. (2008) A global map of human impact on marine ecosystems. *Science* **15**(5865): 948–952.

Hughes, T.P., Bellwood, D.R., Folke, C., Steneck, R. & Wilson, J. (2005) New paradigms for supporting the resilience of marine ecosystems. *Trends in Ecology and Evolution* **20**(7): 380–386.

Lance, M., Richardson, S. & Allen, H. (2004) Washington state recovery plan for the sea otter. Washington Department of Fish and Wildlife, Olympia, USA: 91 pp. [www document]. URL http://wdfw.wa.gov/wlm/diversty/soc/recovery/seaotter/final_seaotter_recovery_plan_dec2004.pdf

Larkin, P.A. (1977) An epitaph for the concept of maximum sustained yield. *Transactions of the American Fisheries Society* **106**(1): 1–11.

Larkin, P.A. (1996) Concepts and issues in marine ecosystem management. *Reviews in Fish Biology and Fisheries* **6**: 139–164.

Mace, P.M. (1997) Developing and sustaining world fisheries resources: the state of science and management (keynote presentation). In: *Developing and Sustaining World Fisheries Resources: the State of Science and Management. Proceedings of the Second World Fisheries Congress*, ed. D.A. Hancock, D.C. Smith, A. Grant & J.P. Beumer, pp. 1–22. Melbourne, Australia: CSIRO Publishing.

Mace, P.M. (2001) A new role for MSY in single-species and ecosystem approaches to fisheries stock assessment and management. *Fish and Fisheries* **2**: 2–32.

Mackinson, S., Blanchard, J.L., Pinnegar, J.K. & Scott, R. (2003) Consequences of alternative functional response formulations in models exploring whale-fishery interactions. *Marine Mammal Science* **19**(4): 661–681.

Mackay, A. (1981) The generalized inverse. *Practical Computing* (September): 108–110.

Myers, R.A. & Worm, B. (2003) Rapid worldwide depletion of predatory fish communities. *Nature* **423**: 280–283.

Myers, R.A. & Worm, B. (2005) Extinction, survival, or recovery of large predatory fishes. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences* **360**: 13–20.

Okey, T.A., Vargo, G.A., Mackinson, S., Vasconcellos, M., Mahmoudi, B. & Meyer, C.A. (2004) Simulating community effects of sea floor shading by plankton blooms over the West Florida Shelf. *Ecological Modelling* **172**(2–4): 339–359.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R. & Torres, F. (1998) Fishing down marine foods webs. *Science* **279**: 860–863.

Pauly, D., Christensen, V., Guénette, S., Pitcher, T.J., Sumaila, U.R., Walters, C.J., Watson, R. & Zeller, D. (2002) Towards sustainability in world fisheries. *Nature* **418**(6898): 689–695.

Pitcher, T.J. (2001) Fisheries managed to rebuild ecosystems: reconstructing the past to salvage the future. *Ecological Applications* **11**(2): 601–617.

Pitcher, T.J. (2005) 'Back to the future': a fresh policy initiative for fisheries and a restoration ecology for ocean ecosystems. *Philosophical Transactions of the Royal Society, Series B. Biological Sciences* **360**(1453): 107–121.

Pitcher, T.J., Haggan, N., Preikshot, D. & Pauly, D. (1999) 'Back to the future': a method employing ecosystem modeling to maximize the sustainable benefits from fisheries. In: *Ecosystem Approaches for Fisheries Management. Proceedings of the 16th Lowell Wakefield Fisheries Symposium AK-SG-99-01*, pp. 447–466. Sea Grant Fairbanks, Alaska, USA: University of Alaska.

Pitcher, T.J., Heymans, J.J., Ainsworth, C.H., Buchary, E.A., Sumaila, U.R. & Christensen, V. (2004) Opening the lost valley: implementing a 'back to the future' restoration policy for marine ecosystems and their fisheries. In: *Sustainable Management of North American Fisheries*, ed. E.E. Knudsen, D.D. MacDonald & J.K. Muirhead, pp. 173–201. Bethesda, Maryland, USA: American Fisheries Society, Symposium 43.

Pitcher, T.J., Ainsworth, C.H., Buchary, E.A., Cheung, W., Forrest, R., Haggan, N., Lozano, H., Morato, T. & Morissette, L. (2005) Strategic management of marine ecosystems using whole-ecosystem simulation modeling: the 'back to the future' policy approach. In: *Strategic Management of Marine Ecosystems*, (NATO Science Series IV. Earth and Environmental Sciences, 50), ed. E. Levner, I. Linkov & J.M. Proth, pp. 199–258. Dordrecht, the Netherlands: Springer: 313 pp.

Pitcher, T.J., Kalikoski, D., Pramod, G. & Short, K. (2009) Not honouring the code. *Nature* **457**: 658–659.

Polovina, J.J. (1984) Model of a coral reef ecosystem. I. The Ecopath model and its application to French frigate shoals. *Coral Reefs* **3**(1): 1–11.

Riedman, M.L. & Estes, J.A. (1998) A review of the history, distribution and foraging ecology of sea otters. In: *The Community Ecology of Sea Otters*, ed. G.R. Van Blarian & J.A. Estes, pp. 4–21. Germany: Springer Verlag.

Rice, J. & Rivard, D. (2003) Proceedings of the zonal assessment meeting: Atlantic cod. Halifax, Nova Scotia. February 17–26, 2003 [www document]. URL http://www.dfo-mpo.gc.ca/csas/Csas/proceedings/2003/PRO2003_021_E.pdf

Rosenberg, A. & McLeod, K.O. (2005) Implementing ecosystem-based approaches to management for the conservation of ecosystem services. *Marine Ecology Progress Series* **300**: 270–274.

Rosenberg, A., Bolster, W.J., Alexander, K.E., Leavenworth, W.B., Cooper, A.B. & McKenzie, M.G. (2005) The history of ocean resources: modeling cod biomass using historical records. *Frontiers in Ecology and the Environment* **2**(3): 83–90.

Scheffer, M., Carpenter, S., Foley, J.A., Folke, C. & Walker, B. (2001) Catastrophic shifts in ecosystems. *Nature* **413**: 591–596.

Sumaila, U.R. (2001) Generational cost benefit analysis for the evaluation of marine ecosystem restoration. In: Fisheries Impacts on North Atlantic Ecosystems: Evaluations and Policy Exploration, ed. T.J. Pitcher & U.R. Sumaila, pp. 3–9. *Fisheries Centre Research Reports* **9**(5): 94 pp [www document]. URL http://www.seaaroundus.org/report/impactpolicyF.htm

Sumaila, U.R. (2004) Intergenerational Cost Benefit Analysis and Marine Ecosystem Restoration. *Fish and Fisheries* **5**(4): 329–343.

Sumaila, U.R. & Walters, C.J. (2005) Intergenerational discounting. *Ecological Economics* **52**: 135–142.

Steneck, R.S., Graham, M.H., Bourque, B.J., Corbette, D., Erlandson, J.M., Estes, J.A. & Tegner, M.J. (2002) Kelp forest ecosystems: biodiversity, stability, resilience and future. *Environmental Conservation* **29**(4): 436–489.

Walters, C.J. & Kitchell, J.F. (2001) Cultivation/depensation effects on juvenile survival and recruitment: implications for the theory of fishing. *Canadian Journal of Fisheries and Aquatic Sciences* **58**(1): 39–50.

Walters, C., Christensen, V. & Pauly, D. (1997) Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. *Reviews in Fish Biology and Fisheries* **7**: 139–172.

Walters, C.J., Pauly, D., Christensen, V. & Kitchell, J.F. (2000) Representing density dependent consequences of life history strategies in aquatic ecosystems: Ecosim II. *Ecosystems* **3**: 70–83.

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

Ward, J.M., Brainerd, T., Freese, S., Mace, P., Milazzo, M., Squires, D., Terry, J., Thunberd, E., Travis, M. & Walden, J. (2001) Report of the National Task Force for Measuring Fishing Capacity. NOAA Technical Memorandum, NOAA, USA.