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Abstract

The Economic Value of Rebuilding Fisheries

Christopher Costello[1], Brian P. Kinlan[2], Sarah E. Lester[3],
and Steven D. Gaines[1]

The global demand for protein from seafood – whether wild, caught or cultured, whether for direct consumption or as feed for livestock – is high and projected to continue growing. However, the ocean's ability to meet this demand is uncertain due to either mismanagement or, in some cases, lack of management of marine fish stocks. Efforts to rebuild and recover the world's fisheries will benefit from an improved understanding of the long-term economic benefits of recovering collapsed stocks, the trajectory and duration of different rebuilding approaches, variation in the value and timing of recovery for fisheries with different economic, biological, and regulatory characteristics, including identifying which fisheries are likely to benefit most from recovery, and the benefits of avoiding collapse in the first place. These questions are addressed using a dynamic bioeconomic optimisation model that explicitly accounts for economics, management, and ecology of size-structured exploited fish populations. Within this model framework, different management options (effort controls on small-, medium-, and large-sized fish) including management that optimises economic returns over a specified planning horizon are simulated and the consequences compared. The results show considerable economic gains from rebuilding fisheries, with magnitudes varying across fisheries.

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The Economic Value of Rebuilding Fisheries

Introduction

The global demand for protein from seafood – whether wild caught or cultured, whether for direct consumption or as feed for livestock – is high and projected to continue growing (Duarte *et al.* 2009). However, the ocean's ability to meet this demand has been called into question. Recent estimates suggest that 28% of global fisheries are overexploited, depleted or recovering from depletion, with an additional 52% at or close to their maximum sustainable limits (FAO 2009). Fishing fleets have responded by expanding their reach into new and deeper waters (Roberts 2002, Pauly *et al.* 2005). But living marine resources are not inexhaustible, as evidenced by global catches of wild caught seafood peaking in the late 1980s and declining slightly since (Watson and Pauly 2001). Continuing catch declines are predicted as more and more fisheries collapse (Worm *et al.* 2006), accompanied by dramatic changes to the trophic structure and functioning of marine ecosystems (Pauly *et al.* 1998, Jackson *et al.* 2001, Myers and Worm 2003). Furthermore, many capture fisheries are reliant on destructive fishing practices that further impact ecosystem health. This growing body of evidence suggests that global fisheries as a whole are increasingly unsustainable (UNEP 2006, Worm *et al.* 2006), which poses a serious threat to the social and economic welfare of fishing communities and those reliant on seafood.

Unsustainable harvests of marine fish stocks can largely be attributed to mismanagement and in some cases, lack of management. Most fish stocks are managed as common pool or open access resources, models of management that lead to the "Tragedy of the Commons" (Hardin 1968) where fishermen race against each other to catch as large a fraction of the yield as they can. The race for fish is in the best interest of individual fishermen since they lack security of ownership, but this race ultimately leads to the economic and often also the ecological collapse of the fishery. This failure of open access management suggests a solution; property-rights based approaches to fisheries management are gaining increasing prominence as a means for correctly aligning incentives to insure the long-term prosperity of fisheries and fishing communities (Costello *et al.* 2008, Festa *et al.* 2008). Indeed, it is now widely agreed that we have the necessary management toolbox to rebuild collapsed fisheries (Worm *et al.* 2009), though the benefits from doing so have not been carefully explored.

In addition to degrading ocean ecosystems and limiting our future access to wild caught seafood, over-exploitation of the world's fisheries has been shown to have severe economic consequences. Using highly simplified steady state models, the World Bank report "The Sunken Billions" has estimated a global loss of approximately USD 50 billion annually associated with mismanagement (Arnason *et al.* 2009). Over the past three decades, this would imply a cumulative global loss of USD 2 trillion comparing actual versus potential net economic benefits. Even more alarming, this may be a conservative estimate as it excludes potential economic costs of current fishery management to the recreational fishing sector, ocean-related tourism sector, and seafood processors and distributors. Although the Sunken Billions analysis draws much needed attention to the potential economic value of rebuilding global fisheries by reducing effort, its methodology and results have been criticised for being overly generalised. Regardless, it is but a starting point as it does not account for the heterogeneity that exists across fisheries, countries, and ecosystems, and thus provides little in the way of concrete guidance for how to design rebuilding plans for particular fish stocks or how to prioritise recovery efforts among fisheries based on expected returns. Moreover, it does not explicitly examine the dynamic paths of fishery recovery, for example, addressing the questions of whether optimal rebuilding plans include a temporary fishery closure, and how and when fishing should resume.

Recovering ailing fisheries is likely to incur short-term costs, including job losses and lost revenues, during the transition to more sustainable catch levels. As a result, rebuilding efforts are often hindered by the real or perceived short-term costs of rebuilding, with little acknowledgement of potential long-term benefits. Balancing economic gains against social and political resistance to the implementation of recovery plans requires evidence of the magnitude and time course of likely longer-term benefits. Sumaila and Suatoni (2005) partially address this need for 17 federally managed fish stocks in the United States that have been declared over-fished (i.e. their abundance is below a specified biomass threshold). The authors determine the net present value (NPV) that can be derived from rebuilding these stocks by comparing a current catch scenario that assumes recent catch patterns will continue unchanged, to a rebuilding scenario that projects catches based on the adopted rebuilding plan for that fishery. They found that rebuilding scenarios yielded three times the NPV of current catch scenarios. This analysis illustrates the utility of a dynamic bioeconomic approach, but it does not investigate how the adopted recovery plans (determined by regulatory processes) compare to rebuilding plans that are economically optimal or to strategies that optimise for other goals. As such, it does not provide the more general insights that are needed for recovering fisheries worldwide.

Efforts to rebuild and recover the world’s fisheries, whether collapsed or heading towards collapse, will benefit greatly from an improved understanding of:

- the long-term economic benefits of recovering collapsed stocks,
- the trajectory and duration of different rebuilding approaches,
- variation in the value and timing of recovery for fisheries with different economic, biological, and regulatory characteristics, including identifying which fisheries are likely to benefit most from recovery, and
- the benefits of avoiding collapse in the first place.

We address these questions by developing and implementing a method for estimating the value of rebuilding collapsed fisheries with different ecological and economic characteristics and the value of avoiding collapse of those fisheries to begin with. We use a dynamic bioeconomic optimisation model that explicitly accounts for economics, management, and ecology of size-structured exploited fish populations. Within this model framework, we simulate and compare the consequences of different management options (effort controls on small, medium, and large-sised fish) including management that optimises economic return over a specified planning horizon. By examining the net present values of fisheries under management policies that rebuild fisheries or maintain them in a rebuilt state, compared to those that maintain or cause collapse, we estimate the value of rebuilding and the benefit of avoiding collapse. We implement this procedure for 18 model fisheries which span a spectrum of different life history, management, and economic characteristics. These hypothetical fisheries are representative of a large class of global fisheries, enabling us to assess the likely benefits of rebuilding fisheries world-wide. We compare an economically optimal rebuilding strategy to more aggressive and less aggressive (with respect to the rapidity of stock recovery) rebuilding tactics. Our approach explicitly includes dynamics of exploited fisheries, management options for improving the economic value of fisheries, and life-history traits of the species being recovered.

The resulting general assessment of the value of rebuilding collapsed stocks, the timing of stock recovery, and the value of avoiding collapse suggests appropriate management strategies for simultaneously rebuilding and enhancing the sustainable economic value of fisheries. We find that there are substantial economic gains from rebuilding collapsed fisheries or from avoiding collapse in the first place. However, realizing these economic benefits requires adopting a rational rebuilding policy. Relying entirely on full fishery closures can erode some of the potential economic value of rebuilding, although also allows fish stocks to rebound more quickly than an economically optimal rebuilding strategy. On the other extreme, allowing continued high levels of harvest to achieve short-term fishery yields can further extend rebuilding times and results in a net loss of economic value. Finally, the value of rebuilding (or of avoiding collapse) and the trajectories of change in stock biomass, harvest yields, and fishery profits are related to underlying life history characteristics of the fish stock, as well as to economic and regulatory parameters of the fishery (e.g. minimum size limits). This analysis suggests the possibility for the development of general guidelines for rebuilding strategies catered to the biological, economic, and regulatory characteristics of the species requiring recovery.

Methods

Model description

To assess the value of rebuilding, we employ a bioeconomic modeling framework with three linked components. (1) A biological stock model, which represents the biological dynamics of the fishery (growth, natural mortality, and recruitment). It is a discrete time, non-spatial, size-structured population model with three size classes and Beverton-Holt type recruitment. (2) A harvest model, which represents fishing as a linear equation relating catch biomass to stock biomass for each size class via a catchability parameter. (3) A profit model, which represents the economic value of harvest, determining annual net profit and, by analyzing profits over time, calculating the net present value of the fish resource given a fishing policy, a discount rate, and a time horizon. Using the coupled stock-harvest-profit model, we then apply a numerical method (Value Function Iteration, implemented as described in Adda and Cooper, 2003) to estimate the size-targeted fishing effort policy function that maximises the net present value of the fishery at any given time (i.e. the "optimal policy").

We parameterised the model to a set of 18 hypothetical fisheries spanning a wide range of basic biological, harvest and economic traits. Biological features of hypothetical fish stocks are reported in Table 1, along with the type of species on which the parameters are based. The example real-world fisheries and data sources that guided development of each hypothetical fishery are listed in the Annex (Tables A1-A3), along with an explanation of the biological parameters. We used these parameters to develop a simple size-structured biological population model of each hypothetical species (Annex). We also obtained basic harvest parameters (catchability) and economic information (prices) from real fisheries and used them as a guide to parameterise harvest, prices and costs of hypothetical fisheries (Table 2).

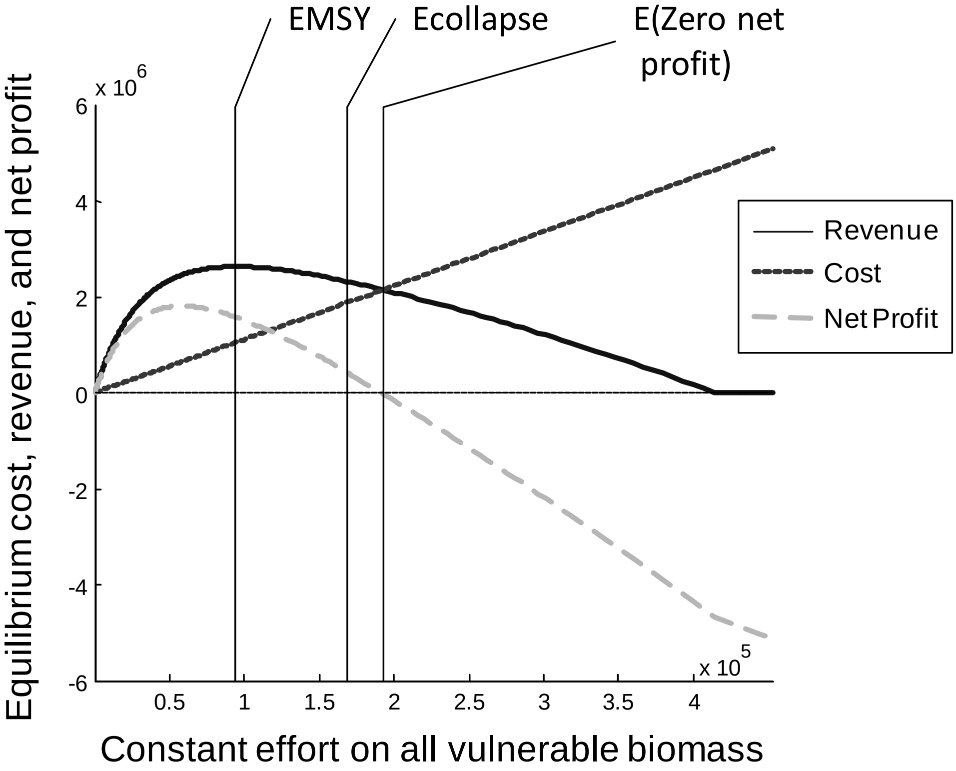
Equilibrium reference points and defining collapse

Using the stock-harvest model for each hypothetical fishery, it is possible to define reference levels of effort, catch, and stock biomass relative to the long-run equilibrium state of the fishery (Figure 1). In order to produce a consistent set of reference points (Table 3), we calculate equilibria under constant effort with knife-edge selection of fish above the legal size (Hilborn and Walters 1992). An important reference point that arises from this analysis is the maximum sustainable yield (MSY), which is the maximum annual harvest the fishery can sustainably produce, given the knife-edge selectivity assumption. The equilibrium, fully recruited, vulnerable stock biomass at MSY is referred to as BMSY. We then select a definition of fishery collapse that is relative to BMSY (Figure 1). We define collapse as a state in which the vulnerable stock biomass is reduced to 50% of its value at MSY (50% of BMSY), or to as close to that level as possible under maximum fishing effort. However, for certain fisheries with relatively high minimum size limits, it may not be possible to reduce the equilibrium vulnerable biomass to 50% BMSY even if all vulnerable fish are removed in each year. This is based upon the guidelines for definition of an "overfished" state currently used by the United States National Marine Fisheries Service (NOAA 1998: United States Code of Federal Regulations 50 Part 600(d)(2)(ii)). The constant, knife-edge selective effort that leads to this equilibrium state is defined as that which brings the fishery into a collapsed state, and once collapsed, maintains it in that state.

Fishing costs

We define cost per unit effort for each fishery relative to the long-run equilibria of the fishery under constant effort with knife-edge selectivity, as described above. Specifically, we use the biological model to determine the effort level that reduces the equilibrium vulnerable biomass of the stock to 40% of BMSY, and chose the cost per unit effort to ensure that the net profit of the fishery is zero at this effort level (note: this is possible because we have data on price received per unit fish biomass harvested). This assumption may not be appropriate for some real fisheries, but ensures that all of our hypothetical fisheries are at least slightly profitable when they are collapsed. For generality, we assume that fixed costs are zero for all hypothetical fisheries. As an extension, for one of our hypothetical fisheries, we explore other cost scenarios, including higher and lower per unit effort costs and two non-zero fixed cost scenarios.

Figure 1. Example of the definition of maximum sustainable yield (MSY), collapse, and zero net profit reference points



Note: Example of the definition of maximum sustainable yield (MSY), collapse, and zero net profit reference points relative to equilibrium revenue, cost and net profit curves. Equilibrium curves are drawn with respect to constant effort on all vulnerable biomass (i.e. knife-edge selectivity). The collapse effort policy is defined as the constant effort resulting in an equilibrium vulnerable biomass below 50% of its equilibrium value at MSY (i.e. <50% BMSY), or as close to this as possible under extinguishing effort. Cost per unit effort is chosen such that stock biomass at the zero net profit reference point corresponds to 40% BMSY, thus ensuring that the fishery is slightly profitable even at collapse. The resulting cost estimates are broadly representative of fisheries that do not become unprofitable before collapse.

Table 1. Biological characteristics of hypothetical fisheries

ID	Name	M[2]	k	Linf	To	Lo	a	b	L repro	L legal	L medium[3]	L large[3]	Length measure[4]
1	Subtropical Small Pelagic	0.80	0.38	25.8	-0.3	.	0.009	3.1	15.9	15	18.4	22.7	cm TL
2	Subtropical Shrimp	2.18	2.04	15.9	-0.08	.	0.046	2.944	11	5.7	11.5	14.0	cm AL
3	Subtropical Grouper	0.18	0.19	118.1	-0.425	.	0.015	2.943	64.8	61	83.1	103.1	cm TL
4	Cold Temperate Scallop	0.11	0.38	14.7	.	0.4	0.023	2.94	4	7.1	10.1	12.7	cm SH
5	Cold Temperate Flounder	0.23	0.29	66.9	-1.146	.	0.005	3.22	26.7	35.6	50.7	59.8	cm TL
6	Subtropical Wrasse	0.12	0.24	43.8	-1.002	.	0.028	2.909	16.6	30.5	32.2	38.7	cm FL
7	Subtropical Snapper	0.18	0.53	44.7	-0.630	.	0.023	2.739	20.9	30.5	33.4	39.5	cm TL
8	Subtropical Jack	0.21	0.34	119.4	-0.45	.	0.033	2.808	82	91.4	85.4	104.5	cm FL
9	Temperate Hake	0.28	0.3	60.9	0.03	.	0.018	2.733	37	30	41.2	52.5	cm FL
10	Tropical/Subtrop. Lobster	0.39	0.34	19	0	.	1.264	2.803	7.25	7.6	13.0	16.4	cm CL
11	Temperate Rockfish	0.04	0.05	49.5	-3.425	.	0.012	3.042	34	28	36.0	43.7	cm TL
12	Subtropical Sparid	0.21	0.21	51.0	-1.32	.	0.021	2.894	28.1	35.56	37.8	45.1	cm TL
13	Warm Temperate Snapper	0.11	0.25	89.4	-0.01	.	0.009	3.104	30.1	50.8	61.8	78.0	cm TL
14	Cold Temperate Sole	0.13	0.15	35.8	0.47	.	0.010	3.056	28.3	10	24.1	31.0	cm TL
15	Temperate Monkfish	0.24	0.13	130	1.510	.	0.019	2.947	39.5	43	83.2	111.2	cm TL
16	Temperate Tilefish	0.10	0.13	107.9	0.494	.	0.005	3.26	48	40	73.4	94.0	cm FL
17	Subtropical Clam	1.52	0.60	8.2	0	.	0.146	3.164	5	6	4	7	cm ShL
18	Temperate Small Pelagic	0.20	0.26	47.3	-1.34	.	0.006	3.154	28.9	24.6	36.0	42.3	cm TL

* Except as noted, all parameters were taken directly from the literature, and sources are given in Annex A. Column headings are explained in Note 1.

1. ID, fishery identification code used in subsequent tables; M, natural mortality (model fit, averaged across size classes); k, Von Bertalanffy growth rate; Linf, Von Bertalanffy asymptotic maximum length; t0, Von Bertalanffy t0 parameter; L0, Von Bertalanffy L0 parameter; a, coefficient of power-law allometric length-weight relationship; b, exponent of power-law allometric length-weight relationship; Lrepro, median or typical length at reproductive maturity; Llegal, median or typical length at legal size or first vulnerability to capture; Lmedium, length at transition from small to medium size class; Llarge, length at transition from medium to large size class; Length measure, units and type of lengths reported for each species.
2. The natural mortality rate reported here is the arithmetic average of the model-fit values of M for each size class, which are reported in Annex Table A5.
3. Lengths defining the transition from small to medium (Lmedium) and medium to large (Llarge) were derived from the growth-in-weight curve calculated by combining the length-weight and Von Bertalanffy growth relationships as described in Methods.
4. TL, total length; AL, abdominal length; SH, shell height; FL, fork length; CW, carapace width; CL, carapace length; ShL, shell length.

Table 2. Economic and harvest characteristics of hypothetical fisheries

ID	Effort cost (2008 USD)	Average price per kg (2008 USD)	q	Extinguishing effort (1/q)	Cost of extinguishing effort (1/q * effort cost)	Effort unit
1	48 596	0.99	7.383E-05	13544	658 178 171	Trips
2	49 584	4.98	2.138E-03	468	23 187 624	boats
3	653	8.15	1.311E-04	7627	4 983 807	Nominal effort units
4	4 344	15.36	1.799E-05	55597	241 493 106	Days at sea
5	114	4.92	2.183E-06	458029	52 306 985	Trips
6	2	5.65	4.336E-06	230645	520 088	Trips
7	112	5.58	3.399E-06	294209	32 898 911	Trips
8	34	2.56	6.564E-06	152350	5 159 419	Nominal effort units
9	3 041	0.13	1.150E-05	86989	264 575 358	Nominal effort units
10	425	12.47	1.506E-05	66412	28 206 428	Trips
11	34 898	2.47	2.059E-02	49	1 694 860	Nominal effort units
12	364	3.13	1.154E-04	8663	3 153 225	Nominal effort units
13	11	6.48	2.206E-06	453226	5 114 282	Hook-hours
14	11 938	0.21	2.582E-04	3874	46 243 547	Nominal effort units
15	19 388	1.96	1.073E-04	9322	180 731 647	Nominal effort units
16	1 333	4.74	8.501E-05	11763	15 684 602	Days at sea
17	691	0.25	7.776E-03	129	88 872	Fishing days
18	89 153	0.33	1.788E-04	5593	498 676 489	Nominal effort units

Table 3. Unfished ("0"), maximum sustainable yield (MSY) and collapse (clpse) reference points*

	Reference Biomasses (B), Harvests (H), and Efforts (E)						Collapse relative to reference points			
ID	**B0 (mt)**	**Bmsy (mt)**	**Hmsy (mt)**	**EMSY**	**Bclpse (mt)**	**Eclpse**	**Bclpse/B0**	**Bclpse/ Bmsy**	**Hclse/ Hmax[1]**	**Hclpse/ Hmsy**
1	3321701	1665105	280587	2282	817296	3439	0.25	0.49	0.06	0.74
2	15604	9559	7708	377	4734	468	0.30	0.50	0.30	0.61
3	7447	1529	651	3246	764	6057	0.10	0.50	0.08	0.93
4	174716	39318	8839	12499	19472	22919	0.11	0.50	0.05	0.91
5	111191	26061	12414	218179	12758	411985	0.11	0.49	0.10	0.92
6	1049	230	61	61385	113	113081	0.11	0.49	0.05	0.90
7	41221	14734	2218	44288	7189	72961	0.17	0.49	0.04	0.80
8	20170	4309	2622	92713	2268	152350	0.11	0.53	0.11	0.87
9	20753749	4699084	2727943	50500	2560375	86989	0.12	0.54	0.12	0.94
10	19005	4579	3091	44825	2977	66412	0.16	0.65[2]	0.16	0.96
11	4924	1712	58	2	847	3	0.17	0.49	0.01	0.82
12	7180	2520	437	1503	1237	2493	0.17	0.49	0.05	0.81
13	7610	1974	410	94237	974	168707	0.13	0.49	0.05	0.88
14	2319808	547078	124386	881	271938	1634	0.12	0.50	0.05	0.92
15	926261	230776	94359	3811	113277	6793	0.12	0.49	0.09	0.87
16	33240	8267	1479	2104	4101	3841	0.12	0.50	0.04	0.91
17	1192	580	508	113	498	129	0.42	0.86[2]	0.42	0.98
18	8445213	3827747	210349	307	1871072	476	0.22	0.49	0.02	0.76
						Mean	**0.17**	**0.53**	**0.10**	**0.86**
						Standard deviation	**0.08**	**0.09**	**0.10**	**0.09**

* For hypothetical fisheries, and metrics for the collapsed state relative to reference points.

1. Hmax is the maximum possible annual catch in the history of a fishery, calculated as described in *Methods*.

2 For these two fisheries the maximum possible fishing effort (1/q) still results in a stock > 50% Bmsy the next year, due to annual recruitment or growth of sub-legal size classes. The "collapsed" state is defined by the equilibrium stock biomass under maximum effort, rather than the 50% Bmsy criterion.

Economic model

We model the economic value of fishery harvest assuming a constant price per kilogram of harvested fish, *p*, and a constant cost per unit of effort applied to harvest those fish, *c*. The first component of the economic model is the annual net profit equation:

$$\pi_t = 1000 \cdot \sum_i pH_{i,t} - \frac{1}{3} \sum_i cE_{i,t}$$

where π_t denotes net profit in year *t*, $H_{i,t}$ is the harvest of size class *i* in year *t* (in metric Tonnes), and $E_{i,t}$ is effort directed at size class *i* in year *t*. Net profit is thus the difference between gross revenues and costs. $H_{i,t}$ is determined from the stock-harvest model defined for each hypothetical fishery (Annex). The factor of 1/3 arises because we express variable costs in terms of unselective effort (equal on all size classes), but the model uses size-specific efforts. We assume that cost structure is constant (the fleet is not dynamic) and price structure is constant (the market is not dynamic; fishermen are “price takers” with respect to a fixed, constant, market price).

Net profit in one year does not fully quantify the economic value of the fishery in that year because it fails to account for expected future profits, reduced by a discount rate reflecting how much one would prefer to have the money now versus later. This more complete valuation, or net present value (NPV), is calculated at time t=0 by summing the profit function from the present time (t=0) over some specified time horizon (t=τ), discounting future profits at rate r:

$$NPV = \sum_{t=0}^{\tau} \frac{\pi_t}{(1+r)^t}$$

In our value analyses, we use a low discount rate of 1%, consistent with the inter-generational social goal of rebuilding (Larkin *et al*, 2000). We also explored the sensitivity of our results to higher discount rates (3% and 7%). To demonstrate how quickly the long-term benefits of rebuilding accrue, we report the time series of annual profits adding up to produce NPV. Because time horizons vary from fishery to fishery depending on the amount of time it takes for rebuilding and collapse to occur, all NPV calculations are reported in annualised form that is independent of time span (equivalent annuity value, Adda and Cooper 2003). Annualised NPV is calculated using the equation:

$$Annualized\,NPV = \frac{NPV}{[1-(1+\mathrm{r})^{-\tau}]/r}$$

Definition of rebuilding

A key feature of our analysis is calculating the value of rebuilding a collapsed fishery. There are many possible definitions of “rebuilding” a fish stock, reflected in a wide variety of real-world rebuilding policies and criteria (e.g. Sumaila and Suatoni 2005). Here we chose a rebuilding definition that is based on the long-run equilibrium stock biomass under an economically optimal fishing policy. As can be seen from the example in

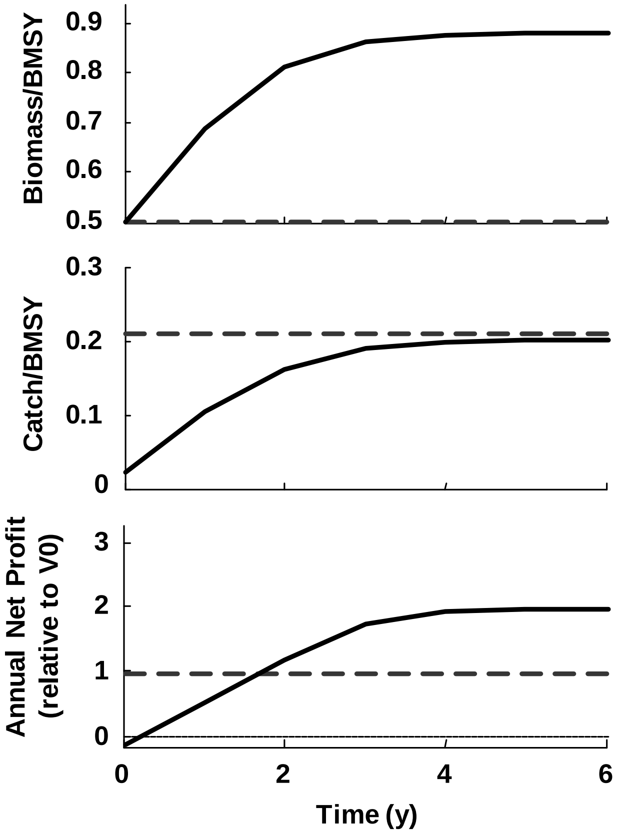
Figure 1, the economically optimal (i.e. maximum net present value) policy will always occur at a lower level of fishing effort, and result in a higher equilibrium stock biomass, than the maximum biological yield. Thus, our definition of rebuilding in terms of the economic optimum meets or exceeds most operational definitions of rebuilding, which are frequently based on the stock biomass that maximises biological yield.

To determine the optimum economic policy, we use a numerical method, Value Function Iteration (VFI), to optimise the rebuilding trajectory by maximizing the long-run NPV (100y, 1% discount rate) of each hypothetical fishery, given the stock, harvest, and economic model parameters. At this low discount rate and long time horizon, rebuilding from collapse was the optimal (i.e. profit-maximizing) decision for all of our hypothetical fisheries. At higher discount rates and shorter time horizons, rebuilding does not maximise net present value for the slowest-growing and latest-maturing species, an issue that has been explored elsewhere in more detail (Larkin *et al.* 2000) and is demonstrated by our discount rate sensitivity analysis. The optimal policy function calculated by VFI can be used to forward-simulate stock dynamics from different initial conditions under economically optimal harvest, allowing assessment of the dynamics of rebuilding. The economically optimal policy function assumes that effort on small, medium, and large size classes can be independently controlled (“size targeting”), and is calculated on a 7x8x9 (504 cell) grid of stock biomasses in the three size classes. This grid size was found to be sufficient to capture differences in value among fisheries for purposes of our analysis, although in some cases a finer grid may be desirable for precise numerical characterisation of optimal policy trajectories for management purposes. The “rebuilt” state is defined as the stock biomass at equilibrium under the optimum economic fishing policy. Under this definition, the target stock biomass for rebuilding can vary based on both economic (cost, price) and biological characteristics of the fishery. The rebuilding goal is defined as following an economically optimal trajectory to the economically optimal long-run equilibrium. In real-world applications, it may be useful (and is straightforward to implement in the VFI optimisation framework) to consider other constraints on the rebuilding target, such as conservation and ecosystem goals.

Scenario description

We explore two sets of scenarios for each of the 18 hypothetical species. Each scenario is generated by starting at a particular state (collapsed or rebuilt, as defined above), and simulating the annual stock, harvest, and net profit using the bioeconomic model. Net present values are calculated over the scenario time horizons, using a discount rate of 1%. The first scenario set compares the net present value of maintaining a fishery in a collapsed state (V0) to rebuilding from collapse (V1; Figure 2). This allows estimation of the economic value of rebuilding from collapse (V1-V0), and the relative value of rebuilding compared to collapse (V1-V0)/V0, or rebuilding value response ratio. We also explore how these results change if the discount rate is 3% or 7%.

Figure 2. Scenario description for rebuilding fisheries



Note: Calculation of the value of rebuilding from collapse. Dynamics of stock biomass (top panel), catch (center panel), and net profit (bottom panel) are illustrated for two scenarios, V0 and $V1^{opt}$. In scenario $V1^{opt}$ (solid black lines), the fishery is rebuilt from an initial collapsed state by fishing according to the optimal economic policy, as defined in *Methods*. In scenario V0 (dashed red lines), the fishery is harvested such that it remains in a constant collapsed state over the same time frame. The value of rebuilding from collapse is calculated as the net present value (NPV) of profit generated under scenario $V1^{opt}$ minus that generated under scenario V0. NPV is evaluated at time 0 and considers all profit generated over the illustrated time period

Three ways of reaching a “rebuilt” state from a “collapsed” state are considered: “optimal” (V1opt), and two suboptimal scenarios, “fast” (V1fast) and “slow” (V1slow). In all three, the fishery begins in the collapsed state. In the optimal scenario, the fishery is rebuilt by fishing according to the economic optimum policy until the stock biomass exceeds 99% of the rebuilt state. In the fast scenario, the fishery is closed (i.e. effort=0) until the stock biomass exceeds 99% of the rebuilt state, and thereafter fishing proceeds according to the optimal policy. In the slow scenario, fishing effort exceeds the optimal policy by 20% for the time period it would have taken to rebuild, preventing the rebuilding threshold from being reached in the same time as it would have been under V1opt; the policy then reverts to the optimum until the biomass is within 99% of the rebuilt threshold. Values of all scenarios in this scenario set (V0, V1opt, V1fast, and

V1slow) are calculated over the time horizon defined by the slowest rebuilding time (V1slow).

The second scenario set compares the value of collapsing a fishery and then rebuilding it (V2), to the value of maintaining it in a rebuilt state without collapse (V3; Figure 7). This allows estimation of the dollar value of avoiding collapse in the first place (V3-V2), and the relative value of fishing optimally without collapse, compared to collapsing and rebuilding (V3-V2)/V2. In both V2 and V3, the fishery begins in the rebuilt state. In V2, the fishery is collapsed by fishing according to the collapse effort policy until the stock biomass is within 1% of the collapsed state, and then allowed to return within 1% of the rebuilt state by fishing according to the optimal policy. To calculate V3, the fishery is exploited according to the optimal policy for the entire planning horizon. Values of the two scenarios are calculated over the time horizon it takes to collapse and then rebuild the fishery in scenario V2. Table 4 summarises the different net present value scenarios we calculated.

Table 4. Definitions and characteristics of the net present value scenarios used in the analyses

NPV scenario	Definition	Effort	Initial stock	Time horizon
V0	NPV of maintaining fishery in collapsed state	Effort to maintain collapse	Collapsed	Time to recover stock using rebuilding scenario in V1slow
V1slow	NPV of rebuilding fishery using “slow” recovery trajectory	20% higher effort than economically optimal effort	Collapsed	Time until recovery
V1opt	NPV of rebuilding fishery using economically optimal rebuilding trajectory	Calculated as dynamical economic optimum	Collapsed	Time to recover stock using rebuilding scenario in V1slow
V1fast	NPV of rebuilding fishery using “fast” recovery trajectory	Zero until recovery is reached.	Collapsed	Time to recover stock using rebuilding scenario in V1slow
V2	NPV of collapsing, then recovering fishery	Effort to maintain collapse, then recover according to economic optimum	Economic optimum	Time to collapse and recover stock.
V3	NPV of optimally managing fishery	Defined as economically optimal effort policy	Economic optimum	Time to collapse and recover stock, as in scenario V2

Regression analyses

A second goal of this analysis was to explore the bioeconomic features of a fishery that lead to large (or small) values of rebuilding. We conducted exploratory analysis of the relationship between biological features of fisheries and two response variables: rebuilding time and the relative value of rebuilding (measured as the log rebuilding response ratio). We used a stepwise multiple linear regression procedure (Sokal and Rohlf 1994) to screen a panel of biological metrics (Annex Tables A6 and A7) for relationships to rebuilding time or value. The stepwise procedure in JMP 8.0 (SAS Institute, Inc.) was used with a probability to enter the model of 0.1 and a probability to leave of 0.25. We then constructed preliminary joint models including all identified regressions, and successively eliminated effects that were not significant at an alpha=0.05 level in post-hoc t-tests of effect magnitude, resulting in final regression models (Table 6). Although p-values of effect tests were not corrected for multiple testing, those that were significant after applying a sequential Bonferroni procedure (Rice 1989) are indicated in the table. We excluded fishery N°17 as an outlier from these analyses because it has an exceptionally low relative value of rebuilding; with the maximum possible fishing effort, the equilibrium biomass for this fishery was still well above the criterion for collapse (Bclpse/Bmsy = 0.86, Table 3). However, final qualitative regression results were not changed by including this outlier. All regression analyses were carried out using JMP 8.0 statistical software (SAS Institute).

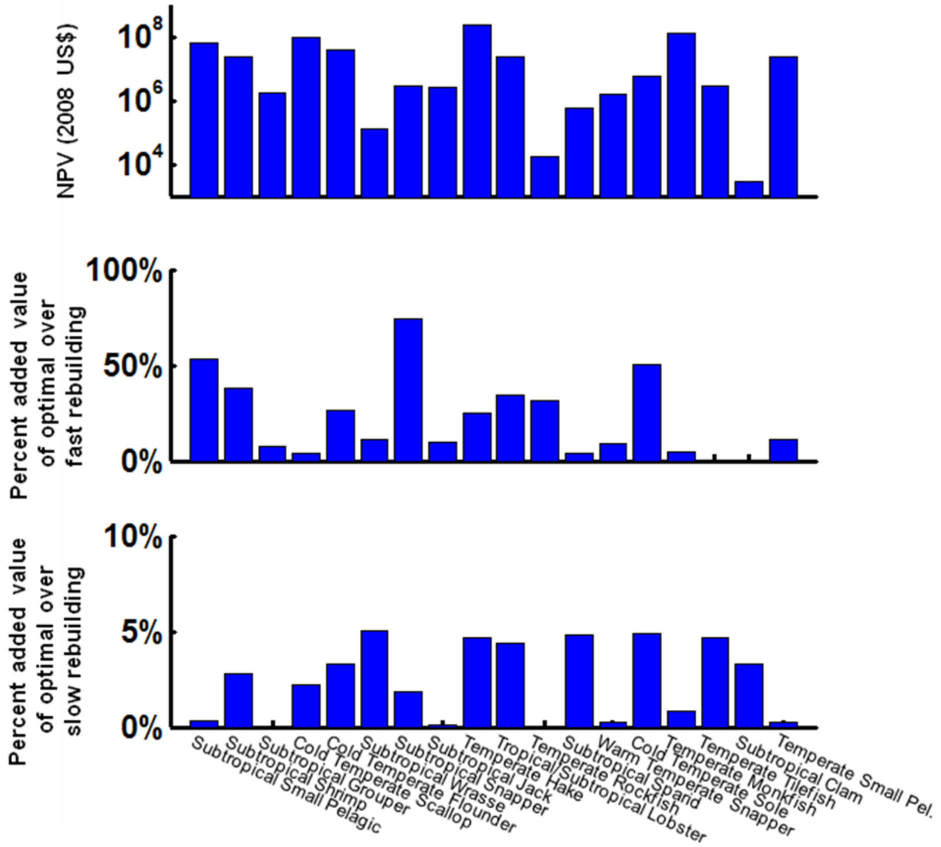
Results and discussion

Value of recovery

The value of rebuilding a fishery from a collapsed state, (V1-V0), is positive for all 18 hypothetical fisheries that we examined when an economically optimal rebuilding strategy is employed (Figure 3; Table 5). Even the two suboptimal rebuilding strategies that we examined yielded positive values of rebuilding with only one exception (Table 5). Value increases (V1-V0) ranged from almost USD 3,000 to over USD 200 million in net present value, depending on the fishery. It is important to note that positive rebuilding values require adequate time for recovery. If we were to examine shorter time horizons, rebuilding strategies would not always result in a net economic benefit relative to maintaining high harvest levels, arguing for the merging of short- and long- term perspectives in fisheries management. Furthermore, the rebuilt state in terms of stock biomass (B/Bmsy) is defined by the economically optimal policy rather than by an externally specified biological reference point and therefore is different for each fishery. For all of our hypothetical fisheries, stock biomass in the rebuilt state exceeded the common Bmsy biological reference point (Table 5), indicating that economically optimal rebuilding policies may also help to advance conservation and ecosystem-based management goals.

Absolute values of a recovered fishery (V1) or of a recovered compared to collapsed fishery (V1-V0) are dependent o . n the scale of the fishery, and thus are not particularly useful metrics for comparing across species. Relative values of rebuilding ([V1-V0]/V0), or rebuilding value response ratios, demonstrate the value of rebuilding fisheries across a wide range of fishery caricatures (Table 5). For the 18 hypothetical fisheries examined here, we found a 580% increase in value resulting from rebuilding the stock from a collapsed state using an economically optimal strategy. Removing an outlier with a particularly high relative rebuilding value, subtropical shrimp, still yields a relative increase in value of 255%, emphasizing the long-term benefits of rebuilding collapsed stocks.

Figure 3. Value of rebuilding for 18 hypothetical fisheries



Note: Top panel: annualised net present value of optimal rebuilding from collapse ($V1^{opt}$-V0; NPV, 2008 US dollars). Center panel: percent increase in value under optimal rebuilding policy compared to fast rebuilding policy (100%*[$V1^{opt}$-$V1^{fast}$]/$V1^{fast}$). Bottom panel: percent increase in value under optimal rebuilding policy compared to slow rebuilding policy (100%*[$V1^{opt}$-$V1^{slow}$]/$V1^{slow}$).

If we are to compare an economically optimal rebuilding scenario to either a more aggressive rebuilding scenario ("fast" rebuilding, in which the fishery is closed until the stock biomass exceeds the rebuilding threshold and then uses the optimal policy) or a less aggressive scenario ("slow" rebuilding, in which harvest levels are 20% higher than the optimal policy during the period of optimal rebuilding and then switch to the optimal policy), we can see the value added of the economically optimal recovery approach (Figure 3). The optimal rebuilding strategy has on average a 23% greater value added of rebuilding than the fast strategy, and a 3% greater value added than the slow strategy. Of course, there are tradeoffs among long-term economic return, short-term harvest or profits, and the biological goals of rebuilding (recovery of stock biomass), and alternate rebuilding tactics perform differently along these dimensions. Stock recovery can be achieved more quickly by using the “fast” scenario, but the long-term economic cost of lost value relative to the optimal approach can be quite high. Similarly, continuing to harvest at a level

Table 5. Values of rebuilding from collapse (V1) relative to maintaining a collapsed state (V0), rebuilding times, and rebuilding biomass, for optimal (opt), fast and slow rebuilding scenarios

	Annualised Net Present Values (2008 US$)				Rebuilding response ratios		Rebuilding time (years)			Rebuilding biomass (B/Bmsy)
ID	**V0**	**V1opt-V0**	**V1fast-V0**	**V1slow-V0**	**(V1-V0)/ V0**	**ln([V1-V0]/V0)**	**Opt**	**Fast**	**Slow**	
1	38 704 717	64 236 417	41 953 382	64 024 926	1.66	0.51	8	7	9	1.00
2	390 746	23 908 315	17 283 197	23 262 145	61.19	4.11	4	2	4	1.02
3	997 437	1 779 450	1 655 280	1 787 564	1.78	0.58	5	3	5	1.20
4	23 942 863	96 498 802	92 620 999	94 382 464	4.03	1.39	15	5	16	1.26
5	9 561 258	37 305 600	29 508 191	36 125 897	3.90	1.36	6	3	7	1.13
6	58 486	130 513	117 489	124 295	2.23	0.80	10	4	10	1.20
7	1 811 839	2 887 293	1 655 837	2 835 464	1.59	0.47	8	7	8	1.15
8	650 188	2 525 534	2 307 504	2 523 360	3.88	1.36	8	4	8	1.92
9	56 999 438	228 427 325	182 697 544	218 225 939	4.01	1.39	7	2	7	1.01
10	8 999 896	24 601 720	18 257 258	23 564 629	2.73	1.01	6	2	6	1.29
11	22 693	17 083	13 008	17 539	0.75	-0.28	26	19	29	1.00
12	208 256	600 906	578 502	573 331	2.89	1.06	22	6	29	1.08
13	448 823	1 580 493	1 453 304	1 576 237	3.52	1.26	17	6	18	1.46
14	4 783 181	5 652 499	3 758 293	5 387 656	1.18	0.17	5	4	6	1.13
15	30 219 020	134 928 576	128 858 557	133 814 943	4.47	1.50	19	3	28	1.00
16	1 242 409	2 815 289	2 811 990	2 688 903	2.27	0.82	12	4	18	1.00
17	36 050	2 883	-6 625	2 792	0.08	-2.53	4	4	5	1.33
18	9 653 670	22 282 116	20 010 048	22 223 329	2.31	0.84	24	14	25	1.06

higher than the optimal, the "slow" scenario, also has drawbacks, delaying rebuilding action and reducing long-term economic value. It is important to note that these are but three strategies for recovering a collapsed stock. There are other potential approaches, and the most feasible approach will depend not only on the stated goals of the rebuilding plan, but also the management tools available for effectively reducing exploitation rates. Designing an appropriate rebuilding policy requires a careful balancing of conservation and fisheries goals over the short- and long-term.

Changes in the size (or age) distribution of the stock are a critical part of rebuilding, and thus size selectivity of fishing effort should be an important consideration in rebuilding strategies. However, the degree to which size targeting can be achieved with existing technologies is often limited in some fisheries. The optimal rebuilding policies calculated by our method assume that effort can be targeted on small, medium, and large size classes (although only biomass that exceeds the minimum size at vulnerability will be catchable). This coarse size-targeting is intended to be broadly representative of reasonably achievable size selectivities in most fisheries, although the ability to control size targeting in particular fisheries will obviously be case-dependent. Table 6 shows size targeting of effort (exploitation rate), size structure of catch, and size structure of the stock relative to its pristine state in the long-run equilibrium of the economic optimum policy (i.e. the rebuilt state).

We can also use our analysis of the value of recovery from collapse to examine sources of value increase, and specifically to assess what characteristics of a fishery are related to a higher relative value from rebuilding. The 18 hypothetical species have a broad range of ecological and life history characteristics, allowing us to examine correlations between population parameters and the value of recovery through a regression analysis approach. Given that many of these parameters are highly correlated with each other and our relatively small species sample size, this analysis cannot provide a rigorous explanatory model (Sokal and Rohlf 1994), but is a useful exploratory approach that can help to identify some of the primary biological features that are related to the value of recovery. We first examined biological parameters that are correlated with optimal rebuilding time (Table 7a). We found that two main parameters can help to predict rebuilding time. The longer it takes a species to reach the legal or vulnerable size, the longer the rebuilding time, which makes intuitive sense, as slow growing species will be slower to recover. Second, we found that stocks with a steeper stock-recruitment relationship are correlated with faster rebuilding times, an unsurprising result given that this steepness parameter essentially measures, in recruitment terms, a stock's ability to recover from very low levels. Finally, reproductive length as a fraction of maximum length was a marginally significant parameter, possibly indicating a tendency of faster rebuilding times for species that grow quickly to a smaller size and thus are able to devote more energy in reproduction relative to growth.

Table 6. Size targeting at equilibrium under the optimal economic rebuilding policy, V1opt.

ID	Effort			Catch			Biomass		
	Equilibrium exploitation rate (% vulnerable biomass caught)			Equilibrium size structure of catch (% of total catch by weight)			Equilibrium size structure of stock biomass (% of unfished by weight)		
	Small	Medium	Large	Small	Medium	Large	Small	Medium	Large
1	0%	25%	0%	0%	100%	0%	58%	54%	40%
2	100%	0%	41%	88%	0%	12%	94%	35%	33%
3	0%	29%	39%	0%	75%	25%	100%	46%	6%
4	0%	0%	50%	0%	0%	100%	99%	99%	17%
5	0%	100%	0%	0%	100%	0%	100%	47%	0%
6	0%	4%	74%	0%	9%	91%	92%	86%	11%
7	0%	0%	22%	0%	0%	100%	76%	76%	34%
8	0%	0%	63%	0%	0%	100%	98%	98%	26%
9	0%	100%	0%	0%	100%	0%	100%	52%	0%
10	0%	100%	0%	0%	100%	0%	100%	60%	0%
11	0%	3%	6%	0%	35%	65%	72%	56%	23%
12	0%	0%	58%	0%	0%	100%	73%	73%	19%
13	0%	0%	44%	0%	0%	100%	95%	95%	21%
14	17%	20%	48%	14%	44%	41%	76%	45%	11%
15	0%	66%	35%	0%	93%	7%	94%	37%	6%
16	7%	26%	51%	5%	56%	39%	84%	48%	8%
17	0%	72%	50%	0%	51%	49%	80%	79%	57%
18	0%	0%	17%	0%	0%	100%	60%	60%	38%

Table 7. Results of final multiple linear regression of model results on biological features of fisheries[1]

Term[2]	Estimate	Std Error	t Ratio	Prob>|t|	Signif.[3]
a) Optimal rebuilding time (whole model: R2=0.66, R2adj=0.58, N=17, p=0.0022)					
Time to reach legal or vulnerable length	1.558	0.456	3.418	0.00458	*
Steepness (*h*) of the Beverton-Holt stock-recruitment function	-16.437	5.254	-3.128	0.00800	*
Length at reproductive maturity as a fraction of asymptotic maximum length	-16.773	7.819	-2.145	0.05141	
b) Log (Rebuilding value response ratio) (whole model: R2=0.71, R2adj=0.67, N=17, p=0.0002)					
Intrinsic growth rate of medium to large biomass	1.524	0.286	5.322	0.00011	*
Steepness (*h*) of the Beverton-Holt stock-recruitment function	1.413	0.569	2.485	0.02622	*

Note: Responses were (a) optimal rebuilding time, and (b) relative value of rebuilding. Final models were selected by a stepwise procedure described in *Methods*. Intercepts were included in models but are omitted from this table.

1. Regression excluded Species #17, which was an outlier, in part because of its non-standard definition of collapse (see Table 3 and Methods).

2. Regressors are sorted in ascending order of p-values.

3. Effects for which p-values were significant at alpha=0.05 after a sequential Bonferroni procedure to correct for multiple testing (Rice 1989) are indicated by an asterisk.

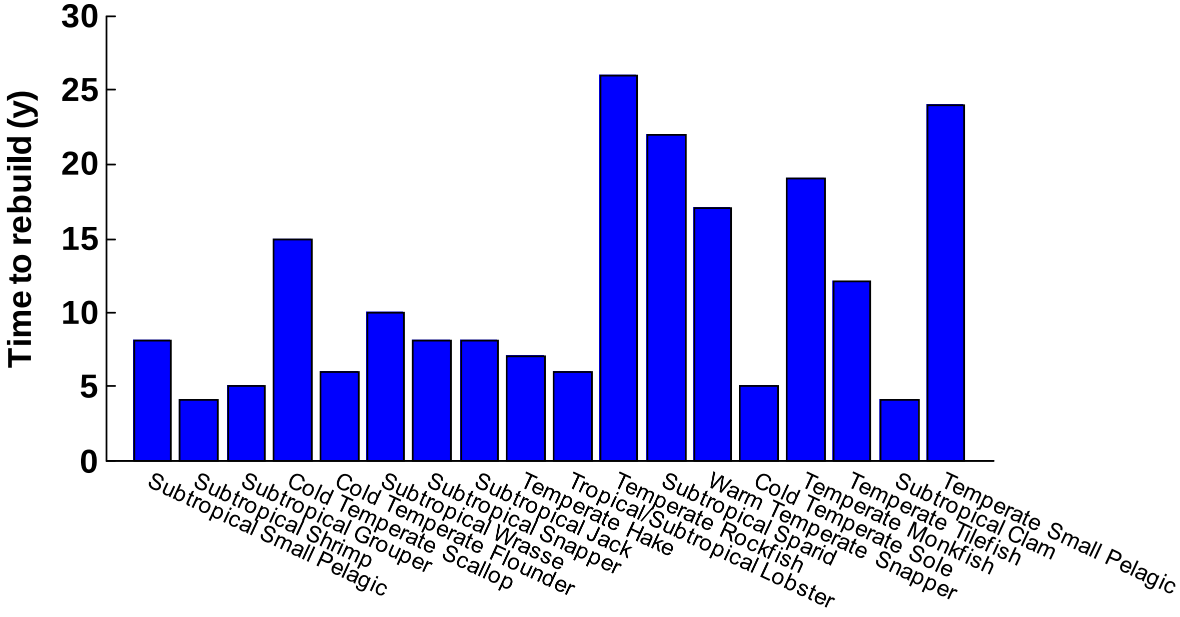
We also examined how biological parameters of the fisheries were related to relative rebuilding value (Table 7b). Species with a higher growth rate in terms of biomass have a higher relative value of recovery in the regression model, matching typical expectations that faster growing species can recover more quickly and thus reach higher yields faster. The steepness of the stock recruitment function is positively related to rebuilding value, matching results from the first regression analysis. Viewed collectively, the results of the regression models provide a starting point for formulating testable hypotheses of the mechanisms underlying the temporal patterns and value of stock recovery and for building a framework for customizing rebuilding plans to species characteristics.

Although the regression analyses found that much of the variability in rebuilding times and values are correlated with biological characteristics of the fisheries, the rebuilding values and times do not arise simply from the biological characteristics of those species. Fishery regulations (minimum size at vulnerability), harvest parameters (catchability), and economics (costs, prices) interact with (and are confounded with) the biological features of the fishery to determine the nature and value of the optimal rebuilding policy. The regulatory choice of minimum size at vulnerability is a good example. Since collapse of the fishery is defined relative to vulnerable stock biomass, a very conservative (high) size limit would generally make collapse easier to reach, and easier to recover from, than if the same species were fished with a lower size limit in place. Thus, one must take into account all features of the hypothetical fisheries, rather than simply biological characteristics, in determining if they may apply to any particular fishery.

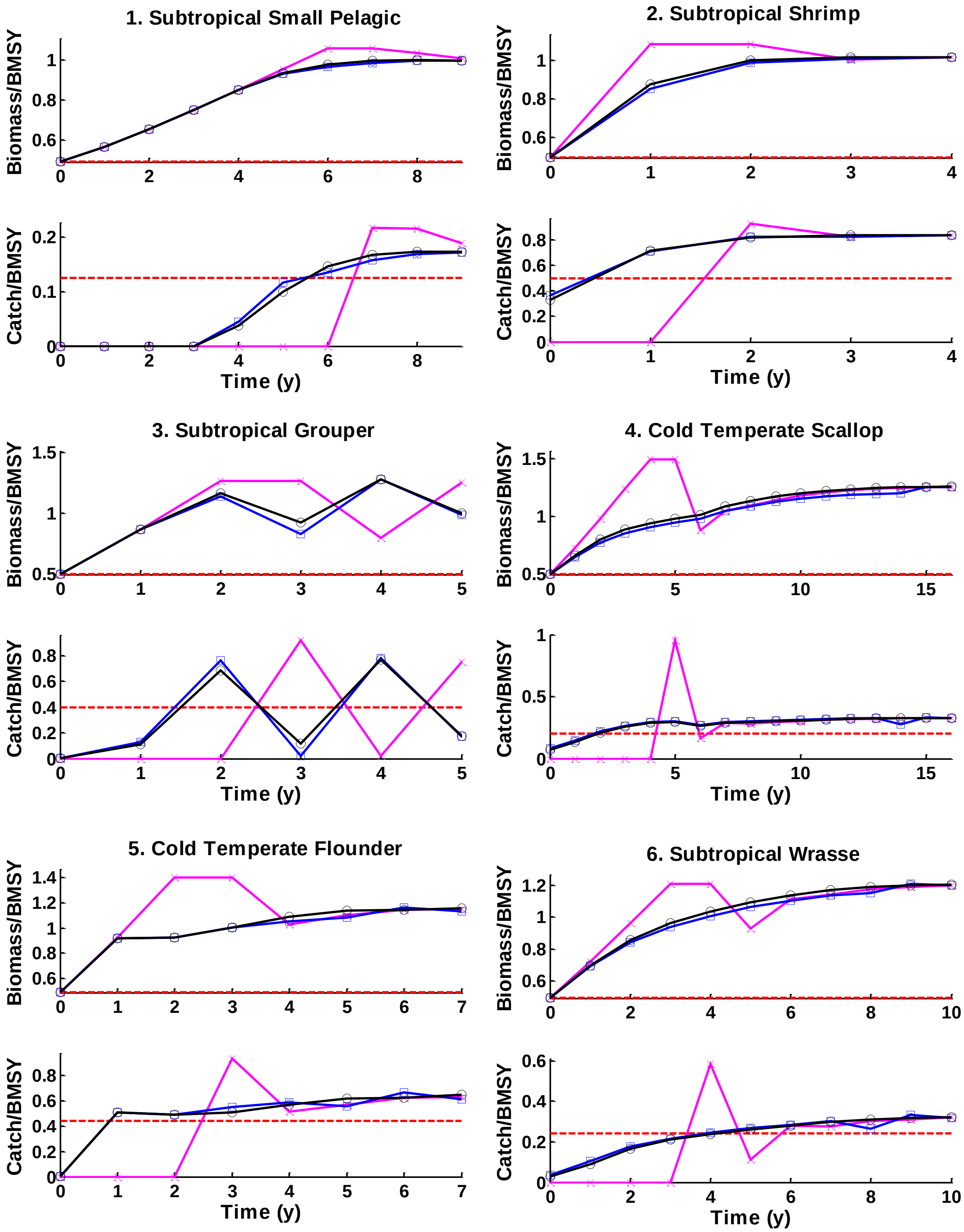
Dynamics of recovery

The economic returns from rebuilding for the 18 fisheries examined here are achieved over a broad range of time frames. Under an optimal rebuilding strategy, stock recovery requires between 4 and 26 years (with a mean of 11 years), depending on the fishery (Figure 4). The slowest species to recover is a temperate rockfish, with subtropical shrimp and clams recovering faster than the other species. These results can help guide expectations for the time course of recovery and economic benefits. When we examine the trajectories of recovery, focusing on temporal patterns in stock biomass and catch and comparing optimal, fast and slow rebuilding strategies, we find a broad range of dynamics across species and rebuilding tactics (Figures 5.1-18). Predicting rebuilding trajectories for specific fisheries will require parameterizing a model like this to the characteristics of individual fisheries, but the patterns we document for these 18 fishery caricatures provide a starting point for understanding how different types of species are likely to respond to rebuilding plans and for characterizing the dynamics of the three rebuilding strategies. For the latter, we find that for the slow rebuilding strategy, there is commonly a minimal effect on biomass and catch relative to the optimal strategy, despite the reduction in value. In other words, fishing substantially harder may have a relatively low benefit for yields while reducing the economic value of the fishery and slowing the recovery rate relative to the optimum because increased fishing effort is more expensive and reduces the dynamic productivity of the resource. Examining the fast rebuilding strategy, in which the fishery is closed until it recovers, we find that stock biomass can reach substantially higher levels over the short-term, which is some cases may represent a higher priority than economic value, but this approach does come at an economic cost as well as possibly social costs of lost fishing opportunities.

Figure 4. Time to rebuild from collapse under optimal rebuilding policy ($V1^{opt}$) for 18 hypothetical fisheries



Figures 5.1 to 5.6. Simulated rebuilding dynamics for hypothetical fisheries



1. Subtropical Small Pelagic

2. Subtropical Shrimp

3. Subtropical Grouper

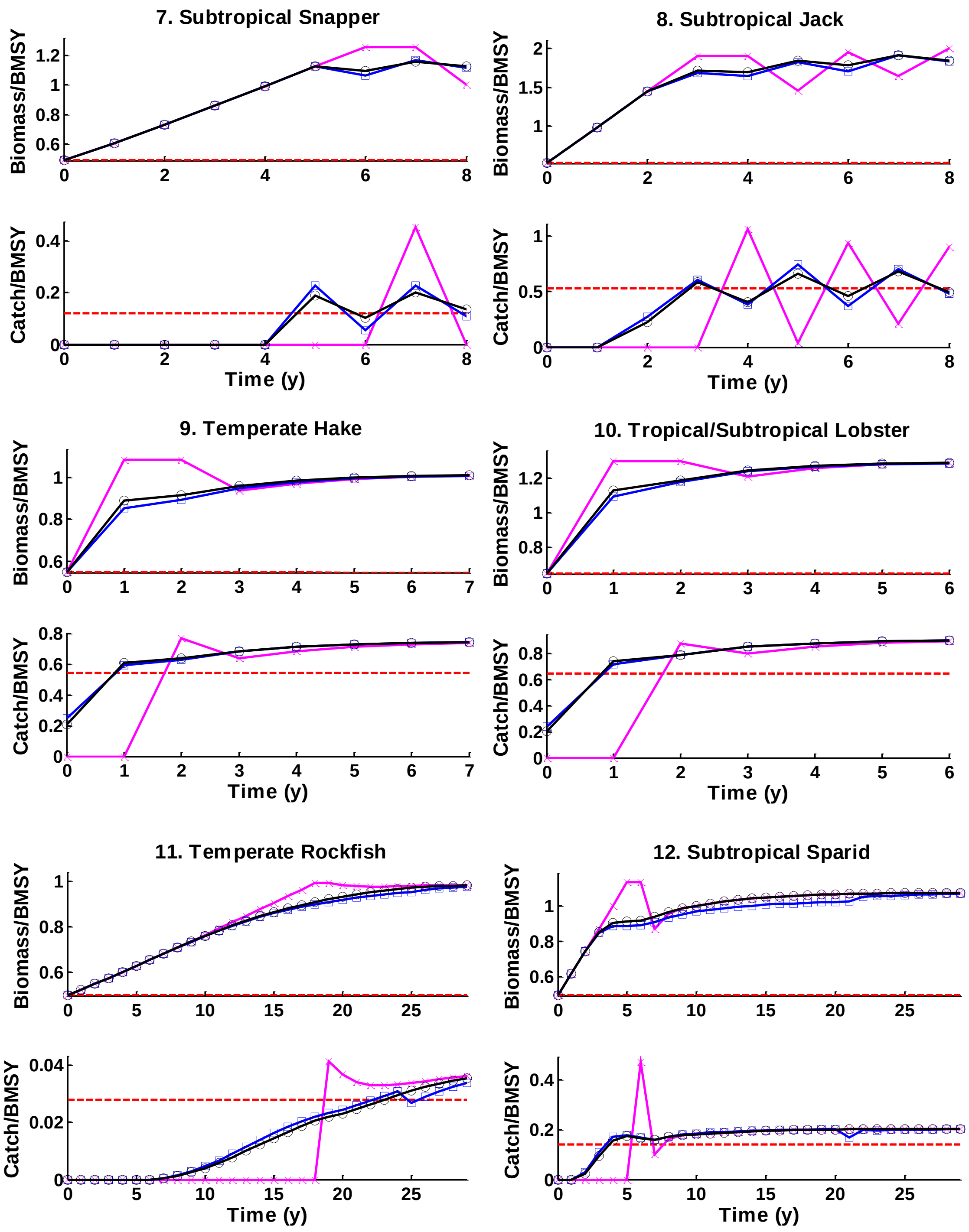
4. Cold Temperate Scallop

5. Cold Temperate Flounder

6. Subtropical Wrasse

Notes: For: a) stock biomass relative to vulnerable biomass at MSY (BMSY) and b) catch biomass relative to BMSY. Solid lines show results for the three rebuilding scenarios considered: “optimal” (V1^{opt}, black with o’s), “fast” (V1^{fast}, magenta with x’s), and “slow” (V1^{slow}, blue with squares). Dashed red lines show the collapse scenario (V0).

Figures 5.7 to 5.12. Simulated rebuilding dynamics for hypothetical fisheries



7. Subtropical Snapper

8. Subtropical Jack

9. Temperate Hake

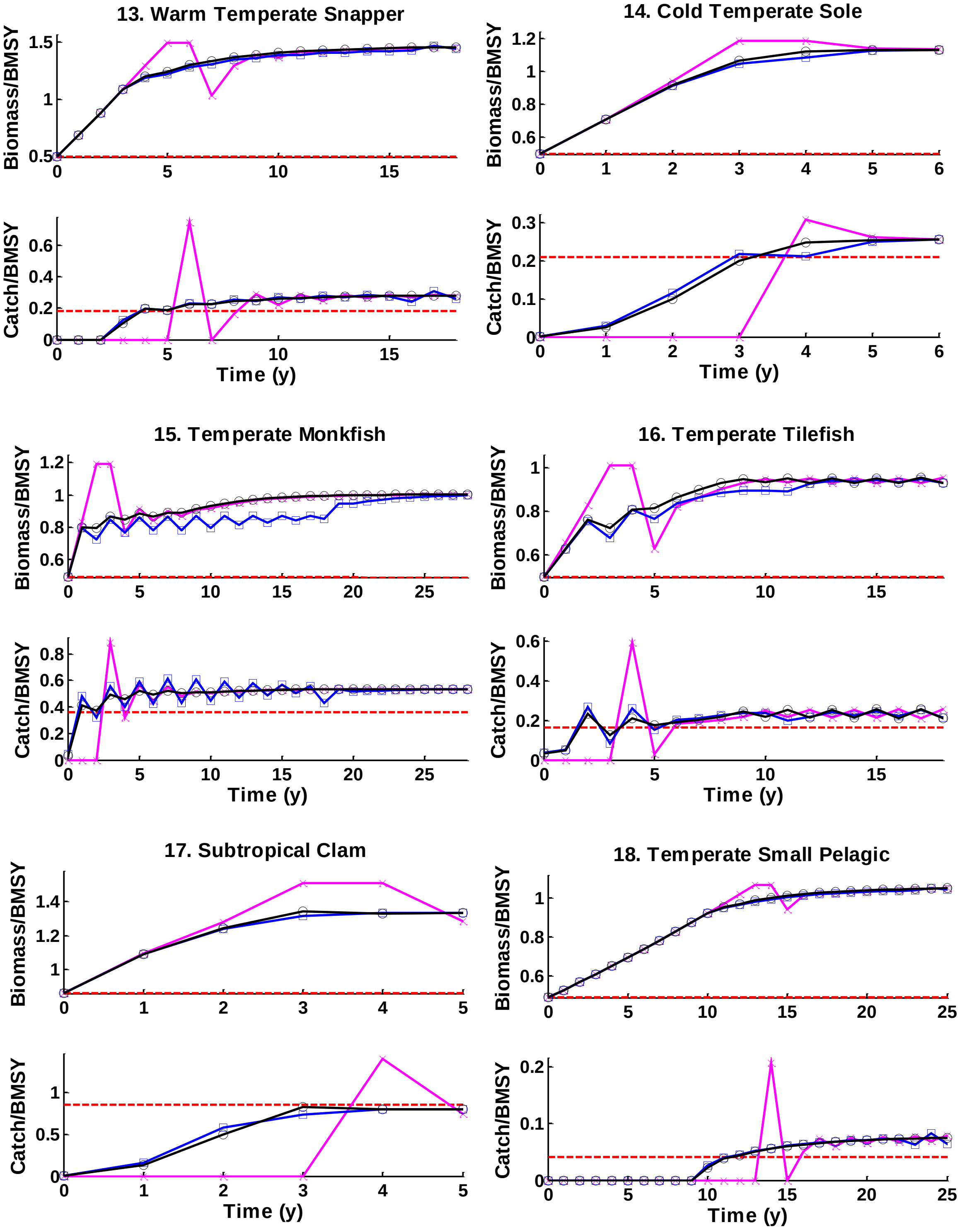
10. Tropical/Subtropical Lobster

11. Temperate Rockfish

12. Subtropical Sparid

Notes: Lines and symbols as in Figure 5.1.

Figures 5.13 to 5.18. Simulated rebuilding dynamics for hypothetical fisheries



Note: Lines and symbols as in Figure 5.1.

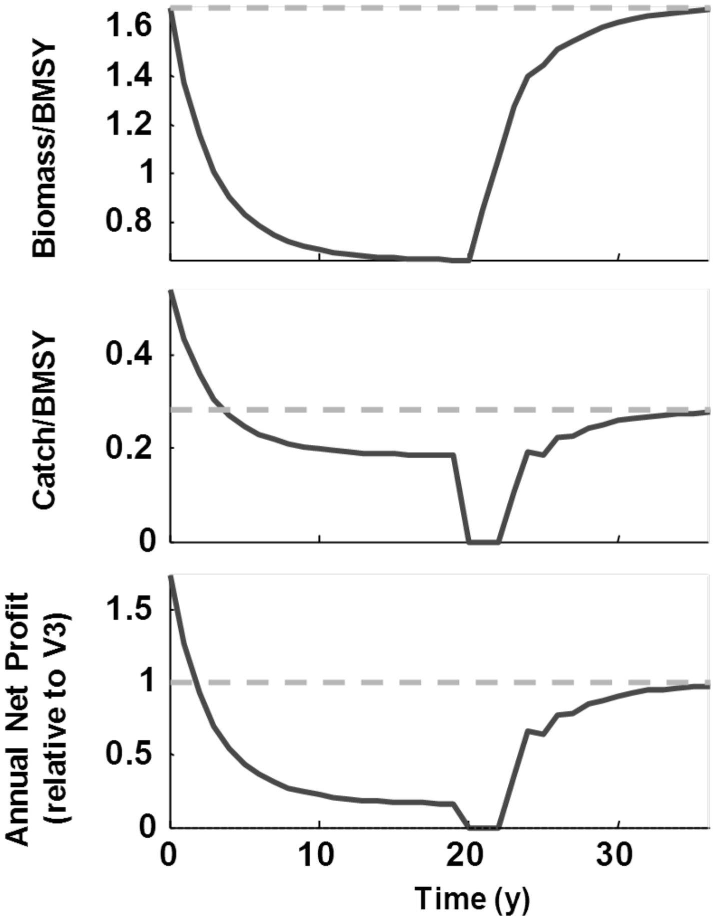
We find that in only a small number of the fisheries (subtropical and temperate small pelagics, subtropical and warm temperate snappers, temperate rockfish) does the optimum policy require a temporary fishery closure as part of the rebuilding plan. These fisheries represent two ends of a spectrum in which slow recovery necessitates a closure – very slow growing species like rockfish and snapper require a long time for biomass growth to rebuild the stock, while small pelagics collapse only under very high fishing mortality and thus require considerable population growth to recover. Comparing the optimal rebuilding policy to the slow and fast policies demonstrates that the economically optimal strategy does not rebuild the fishery as quickly as is possible. Rebuilding goals can include long-term economic value, biological sustainability, and short-term and long-term fishing opportunities. The appropriate rebuilding strategy for a given context will depend on the relative importance of these different goals. The optimal policy modeled here suggests one potential path to recovery in which both socioeconomic and biological goals are addressed, but other strategies may be preferable depending on stated rebuilding priorities. However, we argue that the optimal policy is a good starting point for policy discussions given the tendency in fisheries management to overlook the long-term dynamic value of fisheries.

Avoiding collapse

Thus far we have emphasised the long-term economic benefits of rebuilding collapsed fish stocks. While many fisheries are collapsed, a large fraction of stocks are not currently collapsed. For these, it will be preferable to avoid collapse. There may be significant financial costs associated with implementing and enforcing rebuilding plans, in addition to social and political opposition to reducing fishing pressure (Arnason *et al.* 2009). In particular, effectively reducing fishing mortality from unsustainable levels requires either top-down management or the realignment of incentives. Therefore, in order to achieve sustainable fisheries worldwide we must couple approaches that seek to recover collapsed stocks with those that attempt to avoid collapse in the first place.

We examine the value of avoiding collapse by comparing the value of maintaining an economically optimal fishery (V3) versus collapsing and then rebuilding a fishery back to the optimal state (V2; Figure 6). We find that the value of avoiding collapse is positive for all 18 fisheries (Table 8; Figure 7), with the value attributable to avoiding collapse (V3-V2) ranging from USD 13 000 to approximately USD 150 million in net present value. The relative value of avoiding collapse compared to collapsing (and subsequently recovering) a fishery from an optimal state is on average a 92% increase, with a range of 27% to 176% across the 18 fisheries (Table 8). While these values are all positive, we are examining a much longer time period compared to the time frames of the first analysis. The trajectory of collapsing from an optimal state requires, on average, 32 years for the 18 hypothetical fisheries, with a couple of fisheries taking over a century to collapse (Table 8). In considering these times to collapse, it is important to note that more aggressive collapse effort policies than the one defined here would collapse fisheries more rapidly. Adding the rebuilding times (Table 5), the full collapse-and-rebuild cycle requires an average of 43 years in our set of hypothetical species. Regardless of the time frames, these results support the contention that there are greater long-term economic benefits if collapse is avoided, compared to allowing intense fishing pressure for a short period and then initiating a stock recovery (Figure 7). Given the number of fish stocks worldwide that are collapsed or approaching collapse, this clearly suggests a significant departure from business as usual if we are to maximise the value of global fisheries.

Figure 6. Calculation of the value of avoiding collapse

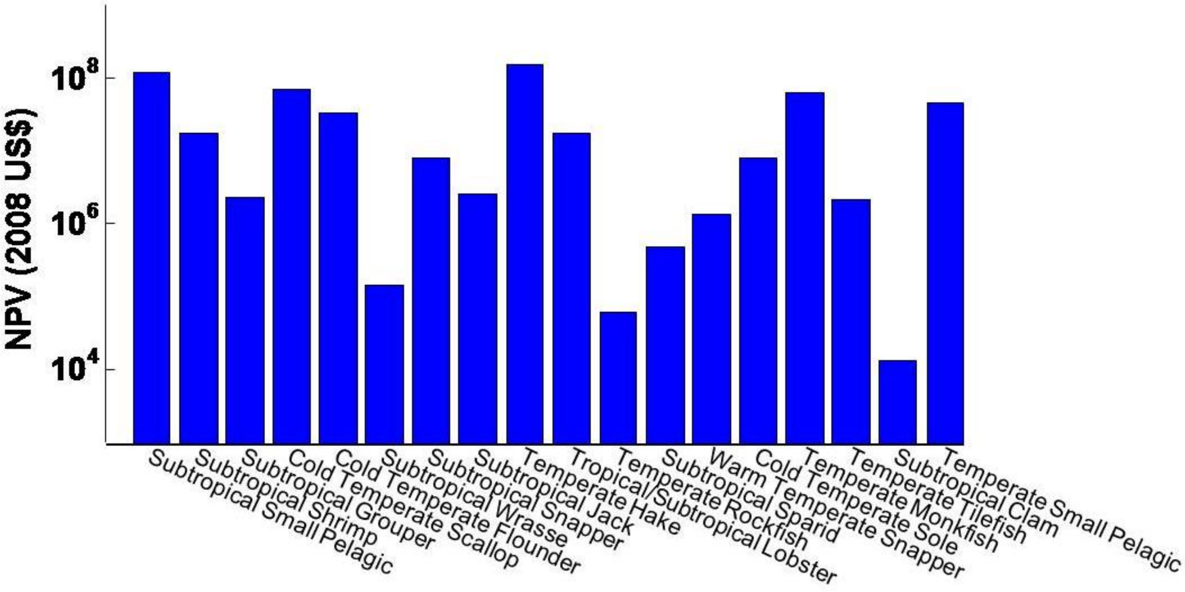


Note: Dynamics of (a) stock biomass, (b) catch, and (c) net profit are illustrated for two scenarios, V2 and V3. In scenario V2 (solid magenta lines), the fishery is collapsed and then rebuilt according to the optimum economic policy. In scenario V3 (dashed green lines), the fishery is harvested such that it remains in a constant rebuilt state over the same time frame. The value of avoiding collapse is calculated as the net present value (NPV) of profit generated under scenario V3 minus that generated under scenario V2. NPV is evaluated at time 0 and considers all profit generated over the illustrated time period.

Table 8. Values of avoiding collapse with times to collapse from the optimal economic equilibrium state.

ID	V2 (USD)	V3-V2 (USD)	Avoid collapse response ratio	Log (Avoid collapse response ratio)	Time to collapse (years)
1	111 204 895	118 052 410	1.06	0.06	31
2	11 891 306	17 242 254	1.45	0.37	7
3	2 719 693	2 285 361	0.84	-0.17	8
4	93 783 307	66 276 687	0.71	-0.35	11
5	35 742 783	31 779 272	0.89	-0.12	5
6	150 595	142 567	0.95	-0.05	16
7	4 401 181	7 728 011	1.76	0.56	39
8	2 722 456	2 413 161	0.89	-0.12	12
9	208 504 699	150 299 026	0.72	-0.33	5
10	26 177 518	16 601 824	0.63	-0.46	4
11	57 375	59 386	1.04	0.03	178
12	546 838	465 264	0.85	-0.16	41
13	1 577 276	1 317 286	0.84	-0.18	19
14	10 788 992	7 899 962	0.73	-0.31	11
15	120 219 934	62 328 510	0.52	-0.66	11
16	3 091 538	2 039 942	0.66	-0.42	17
17	48 421	12 929	0.27	-1.32	15
18	25 623 085	43 798 206	1.71	0.54	146

Figure 7. Value of avoiding collapse (V3-V2) for 18 hypothetical fisheries

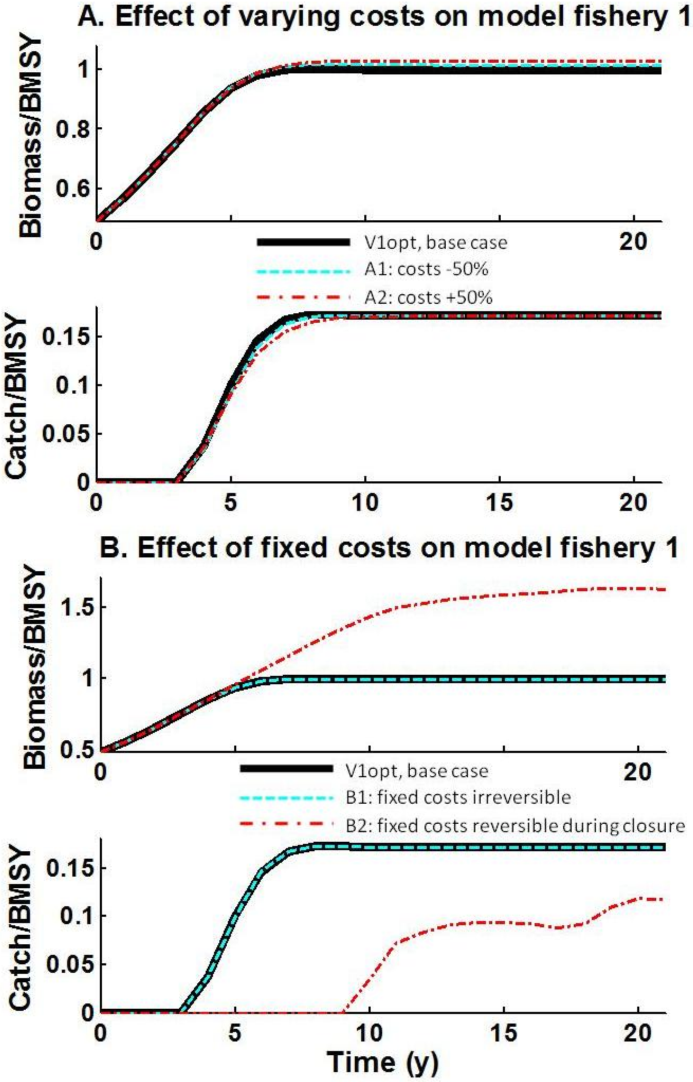


Note: Annualised net present value, 2008 US dollars.

Sensitivity to economic assumptions

In order to standardise the parameterisation of costs for our hypothetical species, we have assumed that costs equal revenues (zero net profit) at a specified point on the yield curve for unselective effort, defined such that zero net profit occurs at an equilibrium stock biomass of 40% Bmsy, and considered all costs to be linearly related to effort without fixed costs. However, real fisheries will have higher and lower costs per unit effort, and some portion of costs may be fixed (independent of effort). Furthermore, depending on how flexible and easily reversible capital and labor costs are in a given context, fixed costs may or may not exist when the fishery is completely closed. We use hypothetical fishery N°1, a temperate small pelagic, to illustrate the effects of different cost structures on the value and dynamics of the economically optimal rebuilding policy (Figure 8). We examined four different cost scenarios, explained in Table 9. We examine two scenarios with no fixed costs but higher or lower per unit effort costs (scenarios A1 and A2). We then examine two scenarios with lower per unit effort costs but annual fixed costs such that the total cost at the zero net profit equilibrium is the same as the base case (scenarios B1 and B2); for one case fixed costs are subtracted from the annual net profit equation in every year and in the other, fixed costs are only applied when the fishery is open.

Figure 8. Effects of changing cost structure on economic optimum rebuilding trajectory, illustrated for model fishery 1



Note: Cost structure scenarios and relative economic values are described in Table 9.

A) Effect of increasing or decreasing costs, when no fixed costs exist.

B) Effect of irreversible fixed costs and fixed costs reversible during closure years.

Table 9. Effects of varying cost structure for hypothetical species 1, a temperate small pelagic.

Cost scenario	Scenario Description
Base case	cost per unit effort = $48,596; no fixed costs
A1	cost per unit effort 50% lower than base case; no fixed costs
A2	cost per unit effort 50% higher than base case; no fixed costs
B1	cost per unit effort 10% lower than base case; annual fixed costs USD 17,694,342; fixed costs imposed annually regardless of closure[1]
B2	cost per unit effort 10% lower than base case; annual fixed costs USD 17,694,342; fixed costs imposed only in years that fishing occurs[1]

	Values			Rebuilding times (yrs)	Economically optimal closure (yrs)
	V0	**V1opt**	**V1opt-V0**		
Base case	USD 38,704,717	USD 102,941,134	USD 64,236,417	8	3
		Change from base case			
A1	218%	21%	-97%	no change	no change
A2	-218%	-8%	119%	+1	no change
B1	-3%	-28%	-44%	no change	no change
B2	-3%	-46%	-72%	+10	+6

1. Fixed costs are chosen such that they represent 10% of total costs at the zero net profit equilibrium, and cost per unit effort is reduced accordingly so that the total cost is the same as the base case.

Our examination of different cost scenarios for one hypothetical fishery provides some insight into how much our overall result would change under different assumptions about the magnitude and nature of costs. Decreasing per unit effort costs by 50% results in a large increase in value of the fishery at collapse (V0) and a smaller increase in the value of the rebuilding trajectory (V1); thus, the net result is a large decrease (97%) in the value of rebuilding (V1-V0) (Table 9). For per unit effort costs that are 50% higher, these effects reverse in direction; there is a substantial increase (119%) in the value of rebuilding. In these two scenarios, the total rebuilding time and the economically optimal closure period (3 years) remain unchanged or virtually unchanged (Table 9) and there are no major differences in the trajectory of rebuilding (Figure 8a). The addition of fixed costs, despite total costs at the zero net profit equilibrium staying constant, does decrease the value of rebuilding by 44% to 72% for this fishery, resulting from a slight decrease in value at the collapsed state combined with a larger decrease in value of the rebuilding trajectory (Table 9). If fixed costs are only realised in years that the fishery is open, it is optimal to close the fishery for longer, until the stock biomass recovers to a point where fishing revenues can outweigh fishing costs (Figure 8a). Because of discounting, this longer closure period makes the value of rebuilding substantially lower than if fixed costs were imposed every year or if there were no fixed costs at all (Table 9). In these later two cases, it is economically optimal to fish at low levels while the stock recovers.

Our fixed cost results indicate that we may be over-estimating the value of rebuilding for the fisheries in our analysis given that fixed costs are likely to exist for many fisheries. They also highlight the importance of a careful consideration of fixed costs when applying this analysis to real fisheries, and in particular, of determining whether fixed costs are maintained or eliminated during periods of fishery closure. On the other hand, it could also be argued that, fixed costs aside, we are underestimating the value of rebuilding for two reasons. First, we have ignored price premiums often realised on larger size fish whose relative share in the catch generally increases as the stock biomass increases under rebuilding (Table 10). Second, we have only accounted for direct value benefits to commercial fisheries. A full economic valuation would also look at indirect use values, option value, bequest value, and existence value and would also consider direct value benefits to processors and retailers, recreational fisheries, tourism, and other affected sectors.

In addition to examining how cost assumptions influence our results, we also examined the sensitivity of our rebuilding results to our discount rate assumption. 1% is a very low discount rate that is consistent with a social desire to rebuild collapsed fisheries, but this assumption could make rebuilding appear more attractive compared to other investments, detracting from more productive social investments and programs. It is also unlikely to reflect discount rates that influence decisions of individual fishermen or private investors in fishing interests, which are likely to be different under different economic incentive structures. For example, Akroyd *et al.* (1999) found that discount rates declined toward the market interest rate in New Zealand fisheries following implementation of an ITQ system. For our purposes, the choice of discount rate can affect three important characteristics of the rebuilding policy: the relative value of rebuilding compared to collapse, the length of time to rebuild, and the biomass threshold for economically optimal rebuilding. We explored sensitivity of all of these features of the economic optimum policy to the use of higher discount rates (r=3% and r=7%; Table 10). First, we find that at higher discount rates, some fisheries are no longer economically optimal to rebuild (marked with "." in Table 10). In fact, it is already well known that in these types of multi-cohort fishery bioeconomic models, the economically optimal policy switches from a sustainable to an unsustainable policy at sufficiently high discount rates, and this switch point will vary from species to species, as is the case for our 18 fisheries. Previous work suggest that the switch point will generally occur at lower discount rates for species with a shallower stock recruitment function and for species that are relatively slow growing (low biomass growth rate and/or high natural mortality) (Horwood and Whittle 1986).

For species for which the economically optimal policy continues to allow for rebuilding at these higher discount rates, the details of the effects of discount rate on rebuilding times, rebuilding biomass levels, and rebuilding values are complex and variable across fisheries, and thus are not amenable to simple characterisation (Table 10). This is unsurprising, as the analytic intractability of optimal controls in even relatively simple fisheries bioeconomic models is a classic problem in mathematical biology (Horwood and Whittle 1986). However, overall the discount rate sensitivity analysis suggests that for many species, a higher discount rate does not have a dramatic effect on our results (Table 10). Increasing the discount rate from 1% to 3% results in, on average, a 3% decrease in the relative value of rebuilding (range: 18% increase to 28% decrease) across the 18 fisheries. Increasing the discount rate from 1% to 7% leads to a slightly higher average decrease in relative rebuilding value of 17% (range: 34% increase to 80% decrease). Excluding the outlier of shrimp, our analysis using a discount rate of 1%

Table 10. Effects of discount rate (r) on the relative value, time, and biomass threshold of optimal economic rebuilding (V1opt).[1]

ID	(V1opt-V0)/V0			Change in (V1opt-V0)/V0		Rebuilding time			Rebuilding biomass (B/Bmsy)		
	r=1%	**r=3%**	**r=7%**	**∆(1 to 3%)**	**∆(1 to 7%)**	**r=1%**	**r=3%**	**r=7%**	**r=1%**	**r=3%**	**r=7%**
1	1.66	.	.	.	.	8	.	.	1.00	<1	<1
2	61.19	60.79	60.01	-1%	-2%	4	4	4	1.02	1.02	1.02
3	1.78	1.78	.	0%	.	5	7	.	1.20	1.23	<1
4	4.03	3.88	3.39	-4%	-16%	15	15	11	1.26	1.26	1.05
5	3.90	4.60	3.57	18%	-9%	6	11	7	1.13	1.19	1.11
6	2.23	1.76	.	-21%	.	10	6	.	1.20	1.07	<1
7	1.59	.	.	.	.	8	.	.	1.15	<1	<1
8	3.88	4.22	1.78	9%	-54%	8	10	4	1.92	1.95	1.67
9	4.01	3.95	3.92	-1%	-2%	7	7	7	1.01	1.01	1.00
10	2.73	2.69	2.53	-2%	-7%	6	6	6	1.29	1.29	1.31
11	0.75	.	.	.	.	26	.	.	1.00	<1	<1
12	2.89	.	.	.	.	22	.	.	1.08	<1	<1
13	3.52	3.31	.	-6%	.	17	17	.	1.46	1.46	<1
14	1.18	1.24	1.59	5%	34%	5	9	12	1.13	1.85	1.80
15	4.47	4.39	.	-2%	.	19	20	.	1.00	1.00	<1
16	2.27	1.98	.	-13%	.	12	11	.	1.00	1.05	<1
17	0.08	0.06	0.02	-28%	-80%	4	4	4	1.33	1.33	1.33
18	2.31	2.28	.	-1%	.	24	135	.	1.06	1.08	<1
Mean	**2.55[2]**	**2.78[2]**	**2.40[2]**	**-3%**	**-17%**						
Stdev	**1.28[2]**	**1.40[2]**	**1.38[2]**	**12%**	**37%**						

1. Missing values (.) denote cases in which the economic optimum policy did not result in rebuilding of vulnerable stock biomass to at least the Bmsy threshold, as indicated in the last three columns of the table.

2. Species 2 was excluded from these calculations to limit effect of this species' extremely high rebuilding value.

suggested that the average relative rebuilding value would be a 255% increase (relative to collapse). The predicted average relative rebuilding value changes only slightly for a 3% or 7% discount rate, to 278% or 240%, respectively. Thus, with higher discount rates, fewer fisheries will be economically optimal to rebuild, but those that still are good candidates for rebuilding efforts will have similarly high predicted proportional value increases as was expected under a lower discount rate.

Conclusion

What is the value of recovering a fishery from a collapsed state? Our results suggest that the value can be quite large (perhaps 2-5 fold increases or greater in the value of a collapsed fishery), but that the value may be strongly dependent on ecological, economic, and regulatory characteristics of the fishery. A similar degree of heterogeneity exists in the economically optimal rebuilding time for a fishery (typically 8-20 years), which, according to our results, typically involves small (though non-zero) increasing harvest during the rebuilding trajectory. Furthermore, while a biologically based rebuilding policy that does not consider economics will always be the fastest way to achieve rebuilding goals, our results show that the economically optimum policy often takes a little longer but leads to substantially higher value. Pushing beyond that (fishing harder than the optimum) delays recovery further and also compromises value. The results of this report, and in particular the modeling and analysis framework employed here, may help inform targeted recovery efforts when priorities must be set across a suite of fisheries.

While our results provide a strong economic argument for either avoiding collapse or recovering fisheries that have collapsed, we have not addressed the critical question of how to implement an optimal recovery plan given the complex political and socioeconomic landscape that exists for most fisheries. There are numerous management tools available for reducing exploitation rates, including gear restrictions, closed areas, capacity reduction, reductions in total allowable catch, limited access, and dedicated access/catch shares. Which tools will work best is likely to be context dependent (Worm *et al.* 2009). However, we would argue that appropriately designed rights based systems (catch shares) can better align individual fisher incentives to promote sustainable fisheries. In fact, a recent review highlights how catch shares can decrease fishery collapse, enhance sustainability, and increase the dynamic stability of several ecological and economic performance measures (Costello *et al.* 2010). For the case of collapsed fisheries, if it is economically optimal to rebuild that fishery, rebuilding is expected to arise endogenously in a dedicated access fishery. Of course, for high discount rates, it will not always be optimal to rebuild. However, there there is some evidence from the literature that discount rates drop under dedicated access management (Akroyd *et al.* 1999). Furthermore, our results indicate that the long-run, low-discount rate economically optimal rebuilding threshold would meet most other biological criteria of rebuilding, suggesting that rebuilding efforts driven by incentives will often meet the same goals as those imposed by regulations.

Finally, the results of this study rely on assumptions about the magnitude and structure of costs and the discount rate. For some species, this may be unimportant, and for others it may play a critical role in whether rebuilding efforts are economically optimal, particularly slow growing species and those slow to rebuild a spawning stock. Moreover, because our relatively small set of hypothetical species will seldom apply perfectly in any particular real-world fishery, a detailed bioeconomic analysis using parameters specific to that fishery will always be advisable if important decisions are to be made based on estimates of the economically optimal rebuilding policy. Getting better estimates of cost parameters and discount rates will be particularly useful given that these parameters are often poorly known and yet affect results as shown here. For a detailed bioeconomic model of a particular fishery, one would probably want to use a smaller time step and better size resolution (or age structured model if appropriate data available to parameterise), and solve for economic optimum on a finer grid. This would require a significant computational investment, but might be justified given the values. Of course, one might also wish to include stochasticity, environmental variability, and spatial structure in such a detailed case study, and the addition of these factors might make other methods of searching for optimal or near-optimal policies more attractive.

Simple mathematical formulas for optimal controls are notoriously intractable even for relatively simple fisheries bioeconomic models (Horwood and Whittle? 1986). Here, we combine numerical solutions to the optimal control problem with parameterisations based on a variety of real-world fisheries, and find that patterns emerge with respect to various characteristics of the fisheries. This suggests a novel and useful approach to develop heuristics that can guide predictions of fishery rebuilding value from basic biological, economic, and regulatory features. These heuristics may aid in identifying fisheries particularly likely to exhibit high rebuilding value (or high value of avoiding collapse) as candidates for detailed dynamic bioeconomic studies in support of fishery management decisions.

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Annex

Parameterisation of fishery species, harvest conditions, and economic characteristics

For each hypothetical fishery, we obtained estimates for the following biological traits based on literature values for reference species (Table A1):

- The annual *natural mortality rate*, *M*, of fish biomass.
- The *Von Bertalanffy Growth Function* (VBGF) describing individual growth in length in terms of individual growth-in-length rate, k, asymptotic maximum length, Linf, and either an initial age or an initial length (t0 or L0). For discussion of alternative formulations of the VBGF for t0 or L0, see Hilborn and Walters (1992).
- The allometric scaling relationship between fish length L and fish weight W, defined by a power law regression of the form L=a*W^b. Parameters a and b are estimated by regression on fish length and weight measurements, and widely reported in the literature.
- Length at reproductive maturity (Lrepro) was estimated as the median or typical length at which sexual maturity was reached (for the slower-maturing of the two sexes, if applicable).
- Length at legal size or first vulnerability to capture (Llegal) defined the length above which fish become vulnerable to capture in the model. When possible, we estimated the median or typical legal size from real fisheries representative of the hypothetical species. Otherwise, where no legal minimum sizes were defined, we examined selectivity curves from real-world fisheries to determine a representative length at which substantial capture began.

Estimates of catchability were obtained from the literature (Table A2) and subsequently refined as part of the model fitting procedure. We obtained approximate prices (per kg) for US species by searching the US National Marine Fisheries Service annual landings database (www.st.nmfs.noaa.gov/st1/commercial/landings/annual-landings.html) and other sources. Costs per unit effort are seldom reported and can vary widely even within a fishery at different times and places. Since our aim was to produce a general analysis of the potential value of rebuilding for a range of hypothetical species, rather than bioeconomic case studies of particular fisheries, we defined costs using a standardised, model-based algorithm. We believe this is a good alternative to basing costs on limited and idiosyncratic available data, but the realism of cost assumptions needs to be considered in applying our results to any particular fishery.

Table A1. Real fisheries used as guidelines for parameterizing hypothetical species

ID	Hypothetical fishery	Example fishery	References			
			Von Bertalanffy Growth Fxn	**Length-Weight Relationship**	**Size at vulnerabilty**	**Size at reproductive maturity**
1	Subtropical Small Pelagic	Pacific sardine, Sea of Cortez, Mexico	INAPESCA 2003	INAPESCA 2003	INAPESCA 2003	INAPESCA 2003
2	Subtropical Shrimp	Brown shrimp, Sea of Cortez, Mexico	INAPESCA 2005	INAPESCA 2005	INAPESCA 2005	INAPESCA 2005
3	Subtropical Grouper	Gag grouper, United States	SEDAR-10 2006	Bohnsack and Harper 1988	SEDAR-012 2009	SEDAR-10 2006
4	Cold Temperate Scallop	Sea scallop, Gulf of Maine and George's Bank, United States	NEFSC 2007	NEFSC 2007	NEFSC 2007	NEFSC 2007
5	Cold Temperate Flounder	Summer flounder, United States	NEFSC 2008	NEFSC 2008	NEFSC 2008	Kerns 2006
6	Subtropical Wrasse	Hogfish, United States	SEDAR-6 2004	SEDAR-6 2004	SEDAR-6 2004	SEDAR-6 2004
7	Subtropical Snapper	Yellowtail snapper, United States	SEDAR-3 2003	SEDAR-3 2003	SEDAR-3 2003	SEDAR-3 2003
8	Subtropical Jack	Greater amberjack, United States	SEDAR-9 2006	SEDAR-9 2006	NMFS 2009	Thompson *et al.* 1999
9	Temperate Hake	Pacific hake/whiting, United States	Stauffer 1985	Stauffer 1985	Helser *et al.* 2006	Helser *et al.* 2006
10	Tropical/Subtropical Lobster	Caribbean spiny lobster, United States	SEDAR-8 2005	SEDAR-8 2005	SEDAR-8 2005	SEDAR-8 2005
11	Temperate Rockfish	Blackgill rockfish, United States	Helser 2005	Love *et al.* 1990	Helser 2005	Helser 2005
12	Subtropical Sparid	Red porgy, United States	SEDAR-01 2006	SEDAR-01 2006	SEDAR-1 2002	SEDAR-01 2006
13	Warm Temperate Snapper	Red snapper (SE Atl.) , United States	SEDAR-15 2008	SEDAR-15 2008	SEDAR-15 2008	SEDAR-15 2008
14	Cold Temperate Sole	Yellowfin sole, United States	Wilderbuer *et al.* 2008	Wilderbuer *et al.* 2008	Wilderbuer *et al.* 2008	Wilderbuer *et al.* 2008
15	Temperate Monkfish	Monkfish, United States	NFSC 2007	NFSC 2007	NFSC 2007	NFSC 2007
16	Temperate Tilefish	Golden tilefish (Mid Atl.), United States	Vidal 2009	Turner *et al.* 1983	SDWG 2005	SDWG 2005
17	Subtropical Clam	Chocolate clam, Bahia La Paz, Mexico	Rodriguez 2008	Rodriguez 2008	Rodriguez 2008	Rodriguez 2008
18	Temperate Small Pelagic	Atlantic mackerel, United States	Pauly 1978	NEFSC 2005	NEFSC 2005	O'Brien 1993

Note: With references for the biological parameter values reported in Table 1

Table A2. Initial values and sources of natural mortality (*M*) and catchability (*q*) parameters

ID	M	M reference	q	q reference
1	0.853	INAPESCA 2003	1.000E-04	INAPESCA 2003
2	2.32	INAPESCA 2005	7.500E-04	INAPESCA 2005
3	0.15	SEDAR-10 2006	1.000E-04	n/a[1]
4	0.1	NEFSC 2007	3.200E-05	NEFSC 2007
5	0.225	NEFSC 2007	4.800E-06	NEFSC 2007
6	0.13025	SEDAR-6 2004	7.000E-06	SEDAR-6 2004
7	0.2	SEDAR-3 2003	6.000E-06	SEDAR-3 2003
8	0.25	SEDAR-15 2008	1.000E-05	n/a[1]
9	0.23	Stauffer 1985	1.000E-05	n/a[1]
10	0.34	SEDAR-8 2005	2.000E-05	SEDAR-8 2005
11	0.04	Helser 2005	5.000E-02	n/a[1]
12	0.225	SEDAR-9 2006	1.000E-04	n/a[1]
13	0.1	SEDAR-15 2008	1.600E-06	SEDAR-15 2008
14	0.12	Wilderbuer *et al.* 2008	1.000E-04	n/a[1]
15	0.25	NFSC 2007	1.000E-04	n/a[1]
16	0.1	NEFSC 2005	2.320E-04	NEFSC 2005
17	1.61	Rodriguez 2008	7.000E-03	Rodriguez 2008
18	0.2	NEFSC 2005	1.000E-04	n/a[1]

Note: used as first guesses in the constrained non-linear least-squares model fitting procedure.

1. Value of *q* chosen in conjunction with definition of nominal effort units to agree with estimated fishing mortality. See fishing mortality estimates in reference for catch-effort time series (Table A3).

Table A3. Sources and characteristics of catch-effort time series to which the final biological-harvest model was fit[1]

ID	Year Start	Year End	Source
1	1969	1990	INAPESCA 2003
2	1988	1999	INAPESCA 2005
3	1962	2004	SEDAR-10 2006
4	1999	2007	Clifford 2008
5	1995	2003	NEFSC 2007
6	1992	2001	SEDAR-6 2004
7	1985	2001	SEDAR-3 2003
8	1981	2006	SEDAR-15 2008
9	1997	2007	Stauffer 1985
10	1985	2002	SEDAR-8 2005
11	1995	2004	Helser 2005
12	1972	2004	SEDAR-9 2006
13	1993	2006	SEDAR-15 2008
14	1964	2008	Wilderbuer *et al.* 2008
15	1997	2006	NFSC 2007
16	1979	2004	NEFSC 2005
17	1992	2004	Rodriguez 2008
18	1985	2005	NEFSC 2005

1. The model fitting procedure was used to "tune" the M and q parameters of the biological model, and to fit alpha and beta parameters of the Beverton-Holt stock-recruitment function, as described in Methods.

Stock-harvest model

To represent the biological dynamics of the hypothetical fish stocks, we defined a size structured state-space model. The general features and assumptions of this modeling framework are discussed in Hilborn and Walters (1992). Briefly, we use discrete time, non-spatial equations to describe changes in the size-structured stock biomass **X** over time as a function of size-structured biomass growth matrix **Ψ**, natural mortality **M**, harvest **H**, and recruitment **R**. Here, bold face type is used to indicate vector or matrix variables. In this model, the growth matrix is derived from the individual growth-in-length relationship, a Von Bertalanffy Growth Function:

$$\overline{L}_i = L_\infty - (L_\infty - L_0)\ \exp(-k\overline{a_i})$$

combined with the length-weight relationship:

$$\overline{\mathrm{W}}_{\mathrm{i}} = \alpha(\overline{\mathrm{L}}_{\mathrm{i}})^{\beta}$$

For the purposes of the model described here, all lengths are expressed in centimeters (with the length measurement dependent on the type of species, as shown in Table 1) and all individual weights in grams whole-body wet weight. (Note, however, that prices are defined per kg of fish weight, and stock biomasses and catches are given in metric Tonnes). Deriving **Ψ** from the parameters in Table 1 involves several steps. First, the Von Bertalanffy Growth Function (VBGF) is solved for N_{age} discrete monthly age classes from 0 to the maximum age and midpoint lengths of each age class are converted to individual biomass using the length-weight relationship. An N_{age} x N_{age} matrix is then constructed with superdiagonal elements equal to the ratio of each age class' midpoint biomass divided by the previous age class' midpoint biomass, element ($N_{age,}$ N_{age}) equal to 1, and zero elements elsewhere. This is the age-structured biomass growth matrix and represents the contribution of biomass in age class *i* at time t to biomass in age class *j* at time t+1:

$$\Psi_{i,j}^{age}$$

Assuming that all individuals recruit at the same time and grow at the same rate, this is also the gross biomass growth matrix for the population (neglecting population growth due to recruitment, which will be represented separately). This matrix is re-binned by integrating under the stable age distribution to produce a 3x3 size-structured matrix at a yearly time interval. The result is a biomass growth matrix that enters the final model (for a detailed discussion of transition between age-structured and size-structured growth matrices, see Hilborn and Walters 1992):

$$\Psi_{i,j}^{size}$$

This size-structured matrix represents the contribution of biomass in size class *i* in year t to biomass in size class *j* in year t+1. We use three size classes, corresponding to small, medium, and large individuals (size classes 1, 2, and 3). Size classes were defined in a standardised way relative to the individual growth-in-weight curve that arises from the combination of VBGF and length-weight relationships. Two lengths defined the transition points from small to medium and medium to large size classes, respectively:

Lmedium = length at [(age at which 95% of asymptotic maximum weight achieved)/3.5]

Llarge = length at [(age at which 95% of asymptotic maximum weight achieved)/2]

The equation of state for stock biomass is given by:

$$X_{j,t+1} = \sum_i [X_{i,t} - H_{i,t}]\,\Psi_{i,j} \mathrm{e}^{-M_i} + R_j\left(\sum_i F_i [X_{i,t} - H_{i,t}] e^{-M_i/2}\right) e^{-M_1/2}$$

where $H(i,t)$ is harvest, $R_j(S)$ denotes the Beverton-Holt recruitment function for size class j:

$$R_1(S) = \frac{AS}{B+S} \qquad R_2,\ R_3 = 0$$

S is the spawning stock biomass, and **F** is a vector of weights representing the relative contributions of each size class to the spawning stock. Note that recruitment is not explicitly represented in the growth transition matrix, but rather by the separate vector function $R_j(S)$. The value of X at time t is the vector of biomass in each size class before harvest (i.e. at the start of the fishing season). Reproduction and recruitment are assumed to occur on average in the middle of the year (though the model is robust to this assumption because *alpha* and *beta* of the recruitment function are free parameters fit to catch-effort data, as described below). Note that density dependence is assumed to enter only through the stock-recruitment relationship. Growth and mortality are not density dependent. The output of the stock model is *X*(i,t), the pre-harvest stock biomass in each of the three size classes at time t.

$H_{i,t}$ is the biomass of size class *i* harvested in year *t*, given by the harvest model equation:

$$H_{i,t} = q_i E_{i,t} X_{i,t} V_i$$

subject to the constraint

$$(X_{i,t} - H_{i,t}) > 0$$

The harvest policy is determined by the effort *E*(i,t) exerted on size class *i* at time *t*. *E*(i,t) is treated as an exogenous "control" or input variable, defined by the effort policy (equivalently called the harvest policy or fishing policy). The parameter q relates E(i,t) to the amount of biomass removed (harvested). The harvest equation assumes that catch biomass is linearly related to vulnerable stock biomass for each size class, by a constant catchability parameter *q*. The units of q are biomass caught per unit vulnerable stock biomass per unit effort. The output *H(i,t)* is the biomass of fish caught in size class *i* at time *t*. Harvest, *H(i,t)*, serves as a link from the stock-harvest model, described above, to the economic model, described in the *Methods* section of this report. The size at first vulnerability, Llegal, determines the vector of fractions of each size class that are vulnerable to harvest, **Vuln**. Similarly, the length at reproductive maturity, Lrepro, is used to calculate the first element of the reproductive fraction vector **F**.

Initial guesses of natural mortality and catchability were taken from the literature, and subsequently "tuned" within a narrow range by nonlinear least-squares constrained minimisation fit to catch-effort time series data (using the *fmincon* routine in the Matlab v7.1 R2007b Optimisation Toolbox). Stock-recruitment parameters *alpha* and *beta* and the unknown initial biomass vector **Xinit** were also fit to the time series as free parameters, constrained only to be positive. Input catch-effort time series were taken from real-world fisheries representative of our hypothetical species (Table A3). The unit of catch was metric Tonnes (= 1000kg). The objective function for the minimisation algorithm was the negative root-mean-squared error of the actual versus predicted total catch in the time series. Parameters fit by nonlinear constrained minimisation were q, M, A, B, and Xinit (all other parameters were fixed). Initial guesses for q and M are given in Table A2. The fitting procedure was allowed to explore values from 25% to 400% of the initial guess. *Alpha*, *beta*, and **Xinit** were positive free parameters. **Xinit** is the vector of initial biomasses in each size class for the start of the time series period, and is not subsequently used in our analysis. Model fit statistics are given in Table A4 and final parameters from the model fitting procedure are given in Table A5.

Table A4. Statistics of biological-harvest model fit to catch-effort time series

ID	Catch Prediction RMSE (mt)	Relative error (RMSE/mean)
1	8384	8.4%
2	696	17.9%
3	45	9.1%
4	282	4.0%
5	1185	11.5%
6	17	11.4%
7	106	8.6%
8	153	8.1%
9	66395	23.2%
10	394	14.1%
11	1	0.4%
12	38	9.2%
13	10	15.6%
14	39694	36.8%
15	1103	4.2%
16	433	26.5%
17	68	26.9%
18	17353	31.5%

For example fisheries (see Table A3).

Table A5. Final values of all parameters of fitted biological-harvest model for the 18 hypothetical species

ID	q	M1	M2	M3	Psi (1,1)	Psi (1,2)	Psi (1,3)	Psi (2,2)	Psi (2,3)	Psi (3,3)	alpha	beta	Vuln (1)	Vuln (2)	Vuln (3)	F(1)	F(2)	F(3)
1	7.383E-05	0.705	0.853	0.853	0.692	2.176	0.0	0.556	0.682	1.0	2762785	2142910	0.128	1.0	1.0	0.095	1.0	1.0
2	2.138E-03	2.320	2.109	2.109	0.011	0.957	5.106	0.0	1.524	1.0	38501	58.2	0.631	1.0	1.0	0.077	1.0	1.0
3	1.311E-04	0.182	0.182	0.182	0.833	0.782	0.0	0.778	0.371	1.0	441	2	0.325	1.0	1.0	0.272	1.0	1.0
4	1.799E-05	0.121	0.100	0.100	0.692	1.309	0.0	0.556	0.780	1.0	3588	526	0.437	1.0	1.0	0.730	1.0	1.0
5	2.183E-06	0.186	0.225	0.272	0.750	0.858	0.0	0.667	0.507	1.0	9950	2	0.514	1.0	1.0	0.776	1.0	1.0
6	4.336E-06	0.130	0.123	0.108	0.80	0.639	0.0	0.667	0.511	1.0	45.3	31.3	0.160	1.0	1.0	0.751	1.0	1.0
7	3.399E-06	0.199	0.165	0.165	0.50	1.388	0.0	0.333	1.037	1.0	3014	8857	0.190	1.0	1.0	0.672	1.0	1.0
8	6.564E-06	0.207	0.207	0.207	0.692	1.206	0.0	0.60	0.638	1.0	1471	241	0.0	0.732	1.0	0.065	1.0	1.0
9	1.150E-05	0.278	0.278	0.278	0.750	1.221	0.0	0.636	0.598	1.0	1900849	77.5	0.317	1.0	1.0	0.142	1.0	1.0
10	1.506E-05	0.411	0.411	0.340	0.714	1.531	0.0	0.60	0.653	1.0	2650	3.09	0.425	1.0	1.0	0.451	1.0	1.0
11	2.059E-02	0.048	0.040	0.040	0.952	0.191	0.0	0.935	0.104	1.0	84.9	1013	0.327	1.0	1.0	0.094	1.0	1.0
12	1.154E-04	0.272	0.186	0.186	0.818	0.627	0.0	0.714	0.420	1.0	1030	1606	0.118	1.0	1.0	0.397	1.0	1.0
13	2.206E-06	0.100	0.121	0.121	0.80	0.964	0.0	0.714	0.50	1.0	191	196	0.275	1.0	1.0	0.617	1.0	1.0
14	2.582E-04	0.122	0.120	0.145	0.882	0.728	0.0	0.833	0.289	1.0	69993	28.8	0.566	1.0	1.0	0.0	0.481	1.0
15	1.073E-04	0.250	0.207	0.250	0.900	1.258	0.0	0.857	0.250	1.0	48983	6712	0.250	1.0	1.0	0.278	1.0	1.0
16	8.501E-05	0.083	0.100	0.121	0.895	0.668	0.0	0.846	0.274	1.0	698	202	0.466	1.0	1.0	0.368	1.0	1.0
17	7.776E-03	1.610	1.610	1.331	0.240	9.807	0.0	0.500	1.322	1.0	2889	357	0.0	0.193	1.0	0.0	0.594	1.0
18	1.788E-04	0.171	0.200	0.242	0.778	0.704	0.0	0.667	0.494	1.0	1492403	7534012	0.572	1.0	1.0	0.384	1.0	1.0

Table A6. Biological features (and units) screened for effects on rebuilding time and rebuilding value

Biological feature	Units
Time to reach 95% of maximum length	Years
Time to reach 99% of maximum length	Years
Time to reach reproductive maturity	Years
Time to reach legal or vulnerable length	Years
Asymptotic maximum weight	Kg
Time to reach 95% of maximum weight	Years
Time to reach 99% of maximum weight	Years
Weight at reproductive maturity	kg
Weight at legal/vulnerable length	kg
Weight at transition from "small" to "medium" size class	kg
Weight at transition from "medium" to "large" size class	kg
Maximum age inferred from natural mortality rate	Years
Maximum age inferred from Von Bertalanffy growth rate	Years
Intrinsic growth rate of biomass at transition from small to medium size	Years^{-1}, exponential biomass growth rate
Intrinsic growth rate of biomass at transition from medium to large size	Years^{-1}, exponential biomass growth rate
Intrinsic growth rate of biomass, averaged over size classes	Years^{-1}, exponential biomass growth rate
Natural annual biomass survivorship fraction, small size class	Fraction (annual, biomass)
Natural annual biomass survivorship fraction, medium size class	Fraction (annual, biomass)
Natural annual biomass survivorship fraction, large size class	Fraction (annual, biomass)
Natural annual biomass mortality rate, averaged over size classes	Years^{-1}, exponential biomass loss rate
Steepness (*h*) of the Beverton-Holt stock-recruitment function	Unitless
Von Bertalanffy growth rate	Years^{-1}, exponential growth rate in length
Exponent of allometric length-weight relationship	Unitless
Von Bertalanffy asymptotic maximum length	Cm
Median or typical length at reproductive maturity	Cm
Median or typical length at legal size or first vulnerability	Cm
Length at reproductive maturity as a fraction of asymptotic maximum length	Unitless
Length at vulnerability to capture as a fraction of asymptotic maximum length	Unitless
Length at reproductive maturity as a fraction of length at vulnerability to capture	Unitless

Table A7. Values of biological features of fishery not reported elsewhere (i.e. derived quantities calculated from basic parameters).

ID	Time to reach 95% of max. length	Time to reach 99% of max. length	Time to reach repro. maturity	Time to reach legal or vulnerable length	Asymptotic maximum weight	Time to reach 95% of maximum weight	Time to reach 99% of maximu m weight	Weight at repro. maturity	Weight at legal or vulnerable length	Weight at transition from small to medium size class	Weight at transition from medium to large size class
1	7.587	11.822	2.213	1.995	213.926	10.518	14.789	47.335	39.822	3.005	5.259
2	1.389	2.178	0.497	0.138	158.661	1.910	2.705	53.637	7.744	0.546	0.955
3	15.343	23.814	3.768	3.409	18944.966	20.941	29.483	3242.417	2714.122	5.983	10.471
4	7.918	12.209	0.778	1.697	60.341	10.748	15.075	1.328	7.174	3.071	5.374
5	9.321	14.943	0.632	1.511	3501.685	13.342	19.015	180.900	459.581	3.812	6.671
6	11.428	18.103	0.973	3.948	1659.575	15.783	22.518	98.045	579.439	4.509	7.892
7	5.057	8.110	0.568	1.552	768.852	6.938	10.016	96.145	270.715	1.982	3.469
8	8.285	12.976	2.935	3.781	22479.486	11.248	15.979	7826.727	10614.960	3.214	5.624
9	10.017	15.382	3.156	2.296	1367.817	13.317	18.723	351.112	197.917	3.805	6.658
10	8.813	13.546	1.416	1.504	4859.554	11.795	16.569	326.336	372.446	3.370	5.897
11	52.057	81.866	18.095	12.041	1750.556	72.337	102.411	558.394	309.339	20.668	36.169
12	12.946	20.611	2.485	4.371	1849.994	17.931	25.660	327.855	651.399	5.123	8.966
13	11.974	18.411	1.630	3.354	10140.274	16.436	22.932	343.817	1754.280	4.696	8.218
14	20.858	31.806	11.085	2.700	545.796	28.340	39.384	265.344	11.070	8.097	14.170
15	24.993	37.604	4.358	4.660	33066.610	33.335	46.054	988.030	1268.908	9.524	16.668
16	22.893	34.922	4.902	3.964	20789.593	31.588	43.727	1482.663	818.298	9.025	15.794
17	4.970	7.638	1.562	2.183	113.785	6.851	9.543	23.781	42.344	1.958	3.426
18	10.097	16.240	2.272	1.465	1130.743	14.416	20.613	239.853	143.834	4.119	7.208

For units of each quantity, see Table A6.

Table A7, continued

ID	Max. age inferred from natural mortal. rate	Max. age inferred from VBGF	Intrinsic growth rate of biomass at transition from small to medium size	Intrinsic growth rate of biomass at transition from medium to large size	Intrinsic growth rate of biomass, averaged over size classes	Natural annual biomass survivor-ship fraction, small size class	Natural annual biomass survivor-ship fraction, medium size class	Natural annual biomass survivor-ship fraction, large size class	Steepness (*h*) of the Beverton-Holt stock-recruit function	Length at repro. maturity as fraction of asymptotic max. length	Length at vulner. to capture as fraction of asymptotic maximum length	Length at repro. maturity as a fraction of length at vulner. to capture
1	12.343	13.158	0.368	0.588	0.478	0.494	0.426	0.426	0.387	0.615	0.581	1.057
2	3.228	2.451	4.511	2.040	3.275	0.098	0.121	0.121	0.975	0.692	0.358	1.930
3	38.609	26.316	0.182	0.251	0.217	0.834	0.834	0.834	0.999	0.549	0.517	1.062
4	47.443	13.333	0.368	0.588	0.478	0.886	0.905	0.905	0.988	0.273	0.485	0.563
5	34.435	17.464	0.288	0.405	0.347	0.830	0.799	0.762	1.000	0.398	0.532	0.749
6	47.526	20.738	0.223	0.405	0.314	0.878	0.884	0.898	0.903	0.378	0.696	0.543
7	28.966	9.488	0.693	1.099	0.896	0.819	0.848	0.848	0.594	0.468	0.683	0.685
8	30.231	14.577	0.368	0.511	0.439	0.813	0.813	0.813	0.958	0.687	0.765	0.897
9	25.147	16.667	0.288	0.452	0.370	0.757	0.757	0.757	1.000	0.608	0.493	1.233
10	18.393	14.706	0.336	0.511	0.424	0.663	0.663	0.712	0.999	0.382	0.400	0.954
11	140.809	92.593	0.049	0.067	0.058	0.953	0.961	0.961	0.587	0.687	0.566	1.214
12	28.094	23.810	0.201	0.336	0.269	0.762	0.830	0.830	0.592	0.550	0.697	0.789
13	59.445	20.000	0.223	0.336	0.280	0.905	0.886	0.886	0.911	0.336	0.568	0.592
14	56.143	34.014	0.125	0.182	0.154	0.885	0.887	0.865	1.000	0.790	0.279	2.828
15	36.144	39.185	0.105	0.154	0.130	0.779	0.813	0.779	0.972	0.304	0.331	0.919
16	77.414	37.369	0.111	0.167	0.139	0.921	0.905	0.886	0.976	0.445	0.371	1.200
17	5.588	8.292	1.427	0.693	1.060	0.200	0.200	0.264	0.628	0.610	0.732	0.833
18	37.471	19.084	0.251	0.405	0.328	0.843	0.819	0.785	0.341	0.612	0.520	1.176

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The Economic Value of Rebuilding Fisheries:

Addendum

The following addendum includes additional analysis and discussion regarding some key issues that have arisen from in the report, *The economic value of rebuilding fisheries* (hereafter referred to as "OECD report"). The key goals and findings of our follow-up research include:

- Representativeness of modeled species: The OECD report presented a bioeconomic model to determine the economic value of rebuilding collapsed fisheries and of avoiding collapse, examining 18 model fisheries with diverse life history, management, and economic characteristics. While our intent was for these fisheries to be broadly representative of global fisheries, the example real-world fisheries and data sources that guided development of these fisheries were based on a geographically restricted selection from the United States and Mexico. In order to evaluate whether our results can be generalised to fisheries around the world, we examined how the hypothetical fisheries compare to a dramatically larger subset of global commercial fisheries. Using a global database of four key life history parameters for commercial marine species, we find that the modeled species in the OECD report are largely representative of commercial fishery species worldwide.

- Slow rebuilding strategy: In this addendum, we investigate a more realistic slow rebuilding strategy to better understand the economic and biological consequences of rebuilding a stock at a slower rate than the economically optimal policy. In the original report, we modeled the "slow" rebuilding strategy by multiplying the optimal size-targeted effort by a factor of 1.2 (i.e. a 20% increase in effort over the optimal policy). Modeling the effort increase as a multiplicative function of the optimum policy carries the assumptions that optimal size-targeting is perfectly achieved and adopted under the slow rebuilding strategy, and that periods identified as closures by the optimum policy remain as closures under the slow policy. Given these generous assumptions, the cost of choosing a slow rebuilding policy over an optimal policy is relatively small (1.5% of the optimum value, on average; Figure 3b). In this addendum, we explore an alternative, and more realistic slow rebuilding policy ("new slow"), where a fraction (20%) of the collapse level of fishing effort continues unabated (regardless of closures occurring for the optimal policy) during the rebuilding period, without size targeting. This is modeled by adding 20% of the collapse effort to the optimal policy function for each size class. For this new slow policy, the cost over the optimal policy is much higher (average of 11.6% of the optimum value; Figure 3c), and is similar to the cost of the fast policy used in the OECD report (average 12.6%; Figure 3a). Similar to our previous conclusions, we find that rebuilding strategies that are slower than the economically optimal strategy, because they allow a higher level of fishing effort to continue during the rebuilding period, result in a decrease in rebuilding value and an increase in rebuilding time. Comparing two alternative slow strategies highlights how the magnitude of these changes depends on assumptions about the level and size-targeting of fishing effort relative to the optimum.

- Discount rate: We evaluate in more detail the sensitivity of our modeled results to the choice of a very low discount rate of 1% and identify how high the discount rate would need to be for each fishery in order to change the economically optimal policy to commercial collapse of the stock. Specifically, we conducted additional discount rate sensitivity analyses, confirming our previous finding that fisheries tend to show similar relative rebuilding values at increasing discount rates, until a “switch point” discount rate is reached, above which it is no longer economically optimal to rebuild. We calculated the switch point, which occurs on average at a discount rate of 6% ± 2.6% (mean ± 1SD) for our modeled fisheries. Based on the subset of species in our analysis and our new analysis of the discount rate switch point, between 44% and 72% of fisheries would be economically optimal to rebuild given a discount rate between 5% and 7%, and between 78% and 100% of fisheries would be economically optimal to rebuild with discount rates from 2% to 3%. This new analysis makes clear that economic policies, financing mechanisms, and/or regulatory changes that reduce effective discount rates in fisheries will be a critical part of any broad effort to realise economic benefits from rebuilding.

Representativeness of modelled species

In our original report to the OECD, we developed a bioeconomic model to determine the economic value of rebuilding collapsed fisheries and of avoiding collapse, examining 18 model fisheries. Our intent was to select a group of fisheries for which data were readily available and which also span a spectrum of different life history, management, and economic characteristics. If the hypothetical fisheries are representative of a large class of global fisheries, our general results identifying the economic benefits of rebuilding collapsed fisheries can be extrapolated, with an appropriate degree of caution, to any geography or even to global fisheries management policy. However, the example real-world fisheries and data sources that guided development of each hypothetical fishery are based on a geographically restricted selection from the United States and Mexico, shedding doubt on whether our results can be generalised beyond US and Mexican fisheries.

In order to evaluate whether our modeled species are broadly representative of fisheries worldwide, we examined how the hypothetical fisheries compare to a dramatically larger subset of global commercial fisheries for important life history characteristics. We queried the Microsoft Access version of the Fishbase database (www.fishbase.org) for all marine species (i.e. all species that spend at least one stage of development in saltwater) of commercial importance (i.e. capture fisheries categorised in Fishbase as “highly commercial,” “commercial,” and “minor commercial”). For each of these species, we then queried Fishbase for the following life history parameters: Von Bertalanffy asymptotic maximum length (Linf, in cm), Von Bertalanffy growth rate (k, in 1/y), natural mortality rate (M, in 1/y), and median or typical length at reproductive maturity (Lrepro, in cm). When multiple estimates of a parameter were available for a single species, we calculated the average across all estimates. The resulting global dataset, which includes any species for which estimates of at least one of the four life history parameters were available in Fishbase, includes 1024 species; not all parameters were available for each species in the dataset (Table 1). When comparing the parameter values for our modeled species to the range of values observed in the global dataset, we find that our modeled species mostly fall between the 5th and 95th percentiles for each parameter, and never exceed or fall short of the maximum and minimum values, respectively, observed for global fisheries (Table 1).[1]

For the fisheries in the global commercial dataset for which all four parameter values were available (n=202 species), we conducted a Principal Components Analysis (PCA) to examine the representativeness of our modeled species within the context of all four key life history parameters. Lrepro was first converted to a relative value by dividing by Linf (i.e. % of maximum length at which the species typically reaches reproductive maturity). Data were transformed for normality prior to PCA as follows: Log10(Linf), Log10(k), Log10(M), and Log10(1+[Lrepro /Linf]). Transformed variables passed Shapiro-Wilk tests for normality. PCA on centered and standardised variables identified two leading principal components that explained 85% of the total variance. Principal component 1 (PC1) was most strongly associated with variation in Linf (positive correlation) and k and M (negative correlation). Loadings on PC1 were [Linf, k, M, Lrepro/Linf] = [0.5590, -0.6031, -0.5598, -0.1019], and 59.3% of the variance was associated with this component. Principal component 2 (PC2) was most strongly associated with variation in Lrepro/Linf (positive correlation) and to a lesser extent M (negative correlation). Loadings on PC2 were [Linf, k, M, Lrepro/Linf] = [-0.0357, 0.0016, -0.2151, 0.9759], and 25.3% of the variance was associated with this component. The PCA and all related statistical analyses were conducted using Matlab R2007b with Statistics Toolbox (The Mathworks, Inc.; Natick, MA).

We plotted PC2 versus PC1 for global marine fisheries, and then transformed and plotted our fisheries in this same principal component space (Figure 1a). Qualitatively, the subset of fisheries chosen for our modeling analysis appears representative of global marine fisheries, without major biases or omissions. We also compared the empirical cumulative distribution function (CDF) of our species' PC1 and PC2 values to the CDF of the larger dataset (Figure 1b,c), and used two-tailed Kolmogorov-Smirnov (K-S) tests to evaluate evidence that the distributions of our chosen species differed from the global dataset. High p-values obtained in a K-S test ($p>0.05$) indicate that the sample distribution is likely to come from the same distribution as the larger dataset. K-S tests of PC1 ($p=0.18$) and PC2 ($p=0.61$) distributions provided no evidence to reject the null hypothesis that our sample subset was drawn from the global distribution. We conclude that, qualitatively and quantitatively, the subset of species that we modeled is broadly representative of global marine commercial fisheries in terms of basic life history parameters (Linf, k, M, and Lrepro).

If only two of the four life history parameters are examined, we can evaluate the representativeness of our modeled fisheries in the context of an even larger set of global commercial fisheries. Figure 2 shows a plot of growth rate (k) versus asymptotic maximum length (Linf) for the 927 species in the global dataset described above for which both of these parameters were available. Again, it is qualitatively apparent that our modeled species are representative of growth model parameters observed in commercial fisheries from around the world (Figure 2). It is important to note, however, that the global dataset does have some much larger, slower growing species (bottom right of the figure) that are not represented by any of our modeled species. Our modeled fisheries do

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[1] All tables and figures are in the Annex.

not include any sharks or larger predatory fishes, and therefore our results should only be extrapolated to these types of fisheries with caution. Slow growing species are not only more vulnerable to overfishing, but are also less likely to be economically optimal to rebuild at higher discount rates.

Beyond the life history traits examined in detail here, we argue that our modeled species are broadly representative of global commercial fisheries along other biological dimensions and economic and institutional dimensions, with some caveats discussed in our original report and elaborated upon below. Based on price data and unfished biomass, our modeled fisheries spanned a range of potential economic values (price per pound ranging from USD 0.13 to USD 15.36, in 2008 US dollars; unfished biomass ranging from ~1,200 to 20 000 000 metric tonnes; see OECD report). Our modeled fisheries use a range of different gear types and fleet types and include invertebrates, finfish, pelagic and reef-associated species. Targeted species reside in a range of different environments, although we do not have any exclusively tropical or polar species. In summary, our modeled fisheries are diverse enough to represent many of the common commercially fisheries found worldwide and therefore we believe that generalities arising from our results can be applied to fisheries in OECD countries.

A notable ecological caveat is that certain species groups (e.g. sharks) and geographic areas (e.g. southern hemisphere) are not well-represented in the subset of species we selected, and may deserve further study if those groups/areas are of particular interest. However, the results of the analysis would not be fundamentally different since the parameter ranges spanned by our modeled fisheries includes the ranges for those groups/areas. We also note that we have not considered ecosystem impacts and feedbacks or interactions among multiple species or multiple ocean uses in our model of rebuilding—such topics will be important for future research.

Regarding economic characteristics, the modeled fisheries are oriented toward North American industrialised nations and therefore under-represent some sets of economic conditions, such as those prevalent in small-scale artisanal and subsistence fisheries in Africa and Micronesia. We also do not attempt to distinguish between fisheries management structures (e.g. open access, limited access, catch shares). Further study would be necessary to examine systematic differences in results under different economic systems. As a first step toward understanding the effects of variation in economic structure on our results, see the “changes in cost structure” analysis presented in our original report (Figure 8 and Table 9 in the OECD Report).

Finally, detailed determination of the optimal policy for any particular species will always depend on species-specific parameterisation, and models that capture more detail of a particular fishery’s ecological and economic characteristics are always desirable as part of developing a rational rebuilding plan. The results presented in the OECD report are intended to provide general guidance when fishery-specific bioeconomic optimisation models are not available.

Slow rebuilding strategy

In the original OECD report, we examine three strategies for reaching a “rebuilt” state from a “collapsed” state: “optimal” (V1opt), “fast” (V1fast) and “slow” (V1slow). In all three, the fishery begins in the collapsed state. In the optimal scenario, the fishery is rebuilt by fishing according to the economic optimum policy until the stock biomass exceeds 99% of the rebuilt state. In the fast scenario, the fishery is closed (i.e. effort=0) until the stock biomass exceeds 99% of the rebuilt state, and thereafter fishing proceeds according to the optimal policy. In the slow scenario, fishing effort is 20% higher than effort for the optimal policy (Slow Effort = 1.2 * Optimal Effort, for each size class) for the time period it would have taken to rebuild, preventing the rebuilding threshold from being reached in the same time as it would have been under V1opt; the policy then reverts to the optimum until the biomass is within 99% of the rebuilt threshold. Values of all scenarios in this scenario set (V0, V1opt, V1fast, and V1slow) are calculated over the time horizon defined by the slowest rebuilding time (V1slow). When comparing the results for the slow and optimal rebuilding strategies, there was only a 1.5% reduction in value for the slow relative to the optimal strategy, suggesting that suboptimal management strategies that take longer to rebuild may result in little economic loss, while possibly having large social benefits. In this addendum, we have examined this slow strategy in more detail, and have compared it to an alternative slow strategy ("new slow"), to determine if there truly is significant flexibility in management.

Upon closer examination, we found that the slow strategy used in the OECD report is much closer to the optimal strategy in terms of fishing effort exerted than one might expect by the description "20% higher effort than optimal". This occurs because the increase in effort over the optimal policy is multiplicative, and the optimal effort is often zero or near zero at the beginning of the rebuilding period. Thus, fishing effort early in the rebuilding period differed very little (or not at all, in the case of zero effort for the optimal) between the original slow and optimal policies. Therefore, it is not surprising that the economic value would be so similar, and we suggest caution before interpreting our results to indicate extremely flexible management options. The slow strategy we used was not really analogous to the fast strategy in terms of how much the strategy diverged from the optimal based on fishing effort.

A slow strategy that is more comparable to the divergence captured by the fast strategy would instead apply an amount of fishing effort equal to 20% of the fishing effort in the collapsed state, *added to* the effort that would be exerted under the optimal policy, continuing for the time period it would have taken for the fishery to rebuild optimally (Slow Effort = [0.2 * Collapsed Effort] + Optimal Effort). The policy then reverts to the optimum to allow rebuilding to complete (i.e. until the biomass is within 99% of the rebuilt threshold). We will refer to this strategy as the "new slow" strategy. This could also be referred to as a slower strategy, as rebuilding tends to take longer than it did for the original slow strategy used in the OECD report (Table 2).

Table 2 and Figures 3 and 4 present the results of analyses using the new definition of the slow strategy, and show how they compare to the fast strategy and to the slow strategy used in the OECD report. The new slow strategy results in an average of 11.6% ± 3.2% (mean ± 1SD) reduction in value, expressed as a percentage of the optimal trajectory (Figure 3c). This is substantially greater than the reduction in value seen with the original slow rebuilding strategy (Figure 3b; mean ± 1SD = 1.5% ± 1.5%), and similar in magnitude to the reduction in value seen under the "fast" rebuilding strategy (Figure 3a; mean ± 1SD = 12.3% ± 8.9%). Similar results are seen with respect to rebuilding times. The new slow strategy results in an average 5.6 ± 4.3 year (mean ± 1SD) increase in rebuilding time (Table 2). This is substantially greater than the increase in rebuilding time seen with the original slow rebuilding strategy (Table 2; mean ± 1SD = 1.8 ± 2.7 years), and similar in magnitude to the decrease in rebuilding time seen under the "fast" rebuilding strategy (Table 2; mean ± 1SD = 5.4 ± 4.3 years).

These results highlight the fact that the magnitude of the reduction in value (and increase in rebuilding time) that occurs from employing a slower rebuilding strategy depends on how and by how much fishing effort exceeds the economically optimal effort level. An infinite number of slow rebuilding strategies are possible, and many will still result in economic benefits over maintaining the collapsed state. All such strategies will incur an economic loss relative to an optimal strategy, however, and will tend to increase rebuilding time. The new slow strategy suggests that these impacts can be substantial. However, in evaluating the degree of management flexibility in a real-world fishery, one would want to carefully consider how the details of the fast and slow alternatives presented here relate to real alternative management scenarios. The likely magnitude of the impacts of slower rebuilding would then have to be weighed against the other social goals that decision-makers are attempting to balance with economic benefits.

Given that the new slow strategy better represents a comparable departure from the optimal strategy relative to the fast strategy, we provide a new examination of the trajectories of recovery to update the analysis provided in the OECD report (Figure 4). When we compare temporal patterns in stock biomass, catch and profits and for the optimal, new slow and fast rebuilding strategies, we find a broad range of dynamics across species and these three rebuilding tactics. Under the optimal policy, catch is initially reduced to an average of about 20% of its collapse level, but increases as stock biomass is rebuilt and typically surpasses collapsed-state catches within the first 30-40% of the rebuilding period. Net profits exceed that of the collapsed fishery sooner than catch because of rationalisation of fishing effort; on average this occurs before 10% of the rebuilding period has elapsed. In a small number of the fisheries (subtropical and temperate small pelagics, subtropical and warm temperate snappers, temperate rockfish), the optimum rebuilding policy requires a complete fishery closure. Under the slow rebuilding policy, catch is only reduced initially to an average of about 40% of its collapse level, stock biomass rebuilds more slowly, and as a consequence long-term catches are rapidly outpaced by the optimal policy. Average net profits are lower than those for the optimal policy at all times. For the fast strategy, in which the fishery is closed until it recovers, stock biomass rebuilds more quickly but catches and profits take longer to surpass collapse levels. The overall differences among optimal, fast, and slow strategies can be seen in the comparison of the ensemble mean trajectories (average across all fisheries) plotted in panels (d), (h), and (l) of Figure 4.

Discount rate

In the OECD report, we used a very low discount rate of 1% for the main analyses in the paper and also included a discount rate sensitivity analysis, examining how the results changed at somewhat higher discount rates (3% and 7%). We chose to center our results and discussion using the low discount rate for two main reasons. The first reason was philosophical, the second practical:

- Rebuilding is usually conceived of as a long-term (inter-generational) social goal, for which public investment rates are most relevant. Moreover, there is considerable uncertainty regarding broader impacts of rebuilding; if not achieved, a variety of negative economic impacts not directly accounted for by the analysis may accrue.

- At higher discount rates it is not economically optimal to rebuild some species; at 1%, all species were optimal to rebuild and we could therefore compare all species in the subset simultaneously at the same discount rate.

The first argument has been extensively debated in the literature, and a wide range of opinions exists on the "proper" social discount rate (for recent reviews see Newell and Pizer [2004], Zhuang et al. [2007], and Azar [2009]), particularly for environmental problems for which the full ramifications of failure cannot be readily incorporated into a tractable economic analysis (Newall and Pizer 2004). We will not reiterate this debate here, but recognize that different discount rates may be appropriate dependent on subjective values, priorities, and economic characteristics of the user. For this reason, it is important to consider the implications of the particular choice of a discount rate on our results. We believe the second argument is an excellent reason for keeping the core discussion of our results centered on a 1% discount rate. However, this argument also underscores the need to identify at what discount rate a given type of fishery will switch from one for which it is optimal to rebuild to one for which rebuilding would occur at an economic loss relative to a policy that does not rebuild. Our original discount rate sensitivity analysis found that the character of the optimal policy function did not change substantially with increasing discount rate until reaching an abrupt switch point. At the switch point, the policy changes to one in which either the stock is rebuilt, but to a biomass lower than Bmsy, or alternatively one in which "mining" of the remaining resources at collapse ensues (further collapsing and effectively eliminating the resource). The stability of the relative shape of the policy function up to the switch point provides support for focusing discussion on the 1% results, because the results are qualitatively similar for any discount rate greater than 1% but lower than the switch point. Our analysis of 3% and 7% discount rates approximately identified the switch point for some species, indicated by dots "." in some cells of Table 10 in the original report.

In order to determine a more precise discount rate switch point for each of the modeled species, we have expanded our sensitivity analysis to look at a full spectrum of discount rates to determine for each fishery the discount rate above which it is no longer economically optimal to rebuild to at least the Bmsy threshold. This is a computationally challenging problem, requiring us to devise an efficient heuristic search algorithm that successfully identified discount rate switch points to within 1% for each fishery in our analysis. To find the discount rate switch point, we first examined our initial sensitivity analysis results (with discount rates of r=1%, 3%, and 7%). The annual net profit time series generated by the r=1% optimal policy was discounted at rates ranging from 0.01% to 99.99% and the annualised NPV (time horizon=100y) was plotted versus discount rate. Examining this curve, we observed that the switch point for every fishery lay in or near an interval defined by the minimum of the first derivative and the maximum of the second derivative of the NPV versus discount rate curve. We used this observation to define a range of integer percent discount rates over which to search for the switch point, which spanned 1% to either side of the range defined by nearest-integer-rounded minima/maxima of the appropriate derivative of the NPV vs. rate curve. For example, if [min(d*NPV*/d*r*), max(d2*NPV*/d*r*2)] = [4.1%, 6.8%], which rounds to [4%, 7%], then we searched the discount rate set {3%, 4%, 5%, 6%, 7%, 8%}. We eliminated some values from the search sets based on results of the preliminary sensitivity analysis with rates of 3% and 7%. We then stepped through the search sets running the value function iteration (VFI) numerical optimisation algorithm (described in the original report) for each fishery/discount rate combination, starting from the midpoint rate for each fishery's search set and updating the search set based on the results. Continuing the example given above, if rebuilding was optimal at 5%, then we know it is also optimal at 4% and do not need to run the optimisation for that value. Similarly, if rebuilding is not optimal at 7%, it will not be optimal at 8% either. In this example, knowing the results at 5% and 7% would allow us to limit our subsequent search set to the single rate {6%}. The estimated switch point represents the highest discount rate at which rebuilding to at least the Bmsy threshold was optimal, at the 1% resolution of our numerical analysis. Rather than run the VFI algorithm for all 100 years required to calculate the final policy and value functions, we stopped the VFI if the policy function was observed to have diverged from the optimal policy (indicating that the rate being tested was above the switch point).

In Table 3, we report the discount switch point and the changes in relative rebuilding values as one moves from the r=1% case to the r=3% or r=7% case. We provide two alternative calculations of the mean and standard deviation of percent changes in relative rebuilding value resulting from changes in discount rate. First, we present the original calculation which excluded fisheries for which rebuilding was not economically optimal at a given discount rate (Table 3; note that a minor error in the original calculation of the standard deviation has been corrected). Second, we present the calculation including all species, assigning a percent change in value of -100% to any fishery that was not optimal to rebuild at a given discount rate. Obviously, the latter calculations lead to larger average percent reductions in value. These reductions in value reflect the joint effects of two phenomena that occur with increasing discount rate: fisheries become non-optimal to rebuild, and subtler changes occur in the character of the relative rebuilding policy (which, on their own, tend to reduce but can in some cases increase relative rebuilding value).

On average, the discount rate at which the optimal policy switches from rebuilding to non-rebuilding is 6% ± 2.6% (range: 2% to 10%) (Table 3). For relatively simple resource production functions, it can be established analytically that it may be optimal to harvest a fishery to the brink of biological extinction and put the money in the bank when the economic discount rate (δ) is greater than the intrinsic growth rate of fish biomass (r) (Clark 1973). Under the same simple model, the economically optimal strategy will *always* be to drive the fish stock to extinction when $\delta > 2r$ (Clark 1973). Such a criterion has not been analytically established for the more realistic size-structured model that we have employed here, but it is reasonable to expect a relationship between the growth rate of vulnerable stock biomass near collapse and the discount rate switch point. We would expect the $\delta > 2r$ criterion to be an upper bound on the discount rate switch point, since we have defined the switch point to occur when it is no longer economically optimal to rebuild to Bmsy, whereas Clark's (1973) results were formulated in terms of biological extinction. Nonetheless, the $\delta > 2r$ criterion derived from Clark's model provides a useful benchmark to place our results in the context of more general theory.

We therefore examine the empirical relationship between the discount rate switch point estimated from our numerical analysis, and the intrinsic rate of biomass increase for our modeled stocks at collapse (Figure 5). Because our model incorporates a size-dependent vulnerability and deals with commercial collapse (rather than extinction) as an alternative outcome to rebuilding, we estimate the intrinsic growth rate of vulnerable biomass as the natural log of the finite rate of increase of the vulnerable biomass growth curve as biomass approaches the collapse level. This provides an analogous calculation for comparison with Clark's (1973) simpler model. Empirically, we find a positive log-linear relationship between intrinsic growth rate and the discount rate switch point (dashed black line in Figure 5, R2=64%, p<0.001). As expected from theory, the discount rate switch point always occurs at discount rate $\delta \leq 2r$, and the discount rate switch point increases with increasing growth rate. The empirical results suggest that rebuilding is unlikely to be optimal at any reasonable discount rate for species with extremely low growth rates (r<0.03, i.e. a doubling time of >23 years). However, rebuilding is found to be optimal for all species with growth rates>0.3 (doubling times <2.3 years) for discount rates ≤ 6%.

A recent empirical analysis suggests that given reasonable assumptions about prices, costs and discount rates, it may be economically optimal to harvest fish sustainably, even for low growth rates and high discount rates (Grafton *et al*. 2007). Our results indicate that rebuilding can also be economically optimal for reasonable discount rates (2-10%) for all but the slowest-growing species (r ≥ about 0.03). We note that Grafton *et al.* (2007) model included an increasing cost of fishing at low density and a fixed component of fishing costs, both of which could raise the discount rate switch point in our analyses; therefore, our switch point analysis could be considered conservative. This could be explored by expanded numerical studies that included a range of fixed costs and functional forms of the cost curve. The distribution of discount rate switch points among fisheries has important implications for rebuilding policy. First, we note that the average discount rate switch point of 6%, and the switch points for more than three-quarters of the analyzed fisheries (14 out of 18), are greater than typical discount rates used in public policy analysis (for reference, the average US OMB nominal discount rate over the period from 1990-2010 was approximately 5.6%), indicating that it will often be economically optimal to rebuild a single fishery even when other oft-cited benefits of rebuilding are not considered (for example, non-consumptive uses and improved ecosystem function). However, our results also underscore the critical role that economic structure and policy can play in determining whether a fishery will, in a purely economic sense, be optimal to rebuild. Implied discount rates in modern fisheries, estimated by a variety of indirect methods range from 1% to >50% (Asche, 1999; Akroyd *et al*., 1999; Asche, 2001). Thus, economic policies and financing mechanisms that promote the application of lower discount rates should be a critical part of any effort to promote fishery rebuilding from the standpoint of single-stock economic benefits. Without sufficiently low discount rates, no economically optimal policy that also meets rebuilding goals will exist. There is evidence that secure property rights systems, such as ITQs, can reduce discount rates to within the range of the discount rate switch points found here (Akroyd *et al*., 1999; Asche, 2001).

The broader issue of choosing appropriate discount rates for long-term social cost-benefit analysis is complex and actively debated in the literature (see, for example, recent reviews by Newell and Pizer, 2004; Zhuang, 2007; and Azar, 2009), and a detailed discussion is beyond the scope of this document. Generally speaking, rates of of 3-10% are widely used for intra-generational (shorter term) projects, with typical values between 5-7%. Rates in the range of 0.5% to 4% are often proposed for longer-term, inter-generational projects, with typical values between 2-3% (Newell and Pizer 2004, Zhuang *et al*. 2007, Azar 2009). For example, the US Environmental Protection Agency analytical guidance for cost-benefit analysis suggests intra-generational projects use rates in the 2–3% range subject to sensitivity analysis in the range of 2–3% and at 7%, and that inter-generational projects use rates in the range 0.5–3% with sensitivity analyses in that range and at 7% (summarised in Zhuang *et al*., 2007). Based on the subset of species in our analysis and our new analysis of the discount rate switch point, between 44% and 72% of fisheries would be economically optimal to rebuild given a 5-7% range of discount rates, and between 78% and 100% of fisheries would be economically optimal to rebuild with discount rates in the 2-3% range.

As a final point, we note that one can always find an economically optimal way to rebuild a fish stock if the rebuilding threshold is incorporated directly into the analysis as a constraint (in economic terms, the requirement that the fishery be rebuilt at least to a specified threshold is made endogenous). However, when rebuilding is “forced” by incorporating an endogenous constraint, the resulting policy is only optimal subject to this constraint – and therefore is not appropriate for broader policy debates considering whether or not to rebuild. Our approach differs in that it examines the global optimum economic policy (over a specified long planning horizon) and applies it to determine the trajectory of the fishery from collapse—which allows for the possibility that rebuilding may not in fact be the economically optimal outcome. The discount rate switch point analysis presented here is a useful supplement to our original results in that it identifies the range of discount rates under which rebuilding would be the economically optimal decision. In cases where a decision has already been made to rebuild, then it may be desirable to calculate economically optimal policies with an endogenous rebuilding constraint. For example, Larkin (2000) outlined an approach that incorporates such a constraint and calculated net present values of rebuilding trajectories under different time frames for a hypothetical fishery with discount rates ranging from 0% to 24%.

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Addendum Annex

Tables and figures

Table 1. Selected life history characteristics for the modeled species from the OECD report and a survey of global commercial fishery species.

Modelled Species				
Fishery ID	**Linf**	**K**	**M**	**Lrepro**
1	25.8	0.38	0.8	15.9
2	15.9	2.04	2.18	11
3	118.1	0.19	0.18	64.8
4	14.7	0.38	0.11	4
5	66.9	0.29	0.23	26.7
6	43.8	0.24	0.12	16.6
7	44.7	0.53	0.18	20.9
8	119.4	0.34	0.21	82
9	60.9	0.3	0.28	37
10	19	0.34	0.39	7.25
11	49.5	0.05	0.04	34
12	51	0.21	0.21	28.1
13	89.4	0.25	0.11	30.1
14	35.8	0.15	0.13	28.3
15	130	0.13	0.24	39.5
16	107.9	0.13	0.1	48
17	8.2	0.6	1.52	5
18	47.3	0.26	0.2	28.9
Global Commercial Fisheries				
Statistics	**Linf**	**K**	**M**	**Lrepro**
Sample size	927	972	362	438
Maximum	1683	12	11.3	961
95th percentile	201	1.4	2.22	109.6
5th percentile	17.1	0.09	0.11	10.1
Minimum	2	0.03	0.03	2.1

Table 2. Length of rebuilding period (in years) for optimal, fast and two alternative slow rebuilding scenarios

Fishery ID	**Optimal**	**Fast**		**Slow, original report**		**Slow, new**	
		Rebuilding time	**Δ from optimal**	**Rebuilding time**	**Δ from original**	**Rebuilding time**	**Δ from original**
1	8	7	*-1*	9	1	10	*+2*
2	4	2	*-2*	4	0	5	*+1*
3	5	3	*-2*	5	0	7	*+2*
4	15	5	*-10*	16	1	26	*+11*
5	6	3	*-3*	7	1	9	*+3*
6	10	4	*-6*	10	0	13	*+3*
7	8	7	*-1*	8	0	11	*+3*
8	8	4	*-4*	8	0	11	*+3*
9	7	2	*-5*	7	0	12	*+5*
10	6	2	*-4*	6	0	10	*+4*
11	26	19	*-7*	29	3	34	*+8*
12	22	6	*-6*	29	7	36	*+14*
13	17	6	*-11*	18	1	26	*+9*
14	5	4	*-1*	6	1	7	*+2*
15	19	3	*-16*	28	9	33	*+14*
16	12	4	*-8*	18	6	18	*+6*
17	4	4	*0*	5	1	5	*+1*
18	24	14	*-10*	25	1	33	*+9*
Mean	**11.4**	**5.5**	***-5.4***	**13.2**	**1.8**	**17**	***+5.6***
Stdev	**7.2**	**4.4**	***4.3***	**9.1**	**2.7**	**11.1**	***+4.3***

Changes from the optimal rebuilding time are shown in italics alongside the three alternative policies

Table 3. Effects of three discount rates (r=1%, 3%, 7%)1 on the relative value µ of optimal economic rebuilding (V1opt) and these estimated discount rate switch point

ID	**(V1opt-V0)/V0 [1]**			**Discount rate switch point [2]**	**Change in (V1opt-V0)/V0**		**Change in (V1opt-V0)/V0**	
	r=1%	**r=3%**	**r=7%**		**Δ(1 to 3%)**	**Δ(1 to 7%)**	**Δ(1 to 3%)**	**Δ(1 to 3%)**
1	1.66	.	.	2%	.	.	-100%	-100%
2	61.19	60.79	60.01	10%	-1%	-2%	-1%	-2%
3	1.78	1.78	.	6%	0%	.	0%	-100%
4	4.03	3.88	3.39	8%	-4%	-16%	-4%	-16%
5	3.9	4.6	3.57	8%	18%	-9%	18%	-9%
6	2.23	1.76	.	6%	-21%	.	-21%	-100%
7	1.59	.	.	2%	.	.	-100%	-100%
8	3.88	4.22	1.78	8%	9%	-54%	9%	-54%
9	4.01	3.95	3.92	8%	-1%	-2%	-1%	-2%
10	2.73	2.69	2.53	8%	-2%	-7%	-2%	-7%
11	0.75	.	.	2%	.	.	-100%	-100%
12	2.89	.	.	2%	.	.	-100%	-100%
13	3.52	3.31	.	6%	-6%	.	-6%	-100%
14	1.18	1.24	1.59	8%	5%	34%	5%	34%
15	4.47	4.39	.	6%	-2%	.	-2%	-100%
16	2.27	1.98	.	6%	-13%	.	-13%	-100%
17	0.08	0.06	0.02	8%	-28%	-80%	-28%	-80%
18	2.31	2.28	.	4%	-1%	.	-1%	-100%
Mean	**2.55 [3]**	**2.78 [3]**	**2.40 [3]**	**6%**	**-3%**	**-17%**	**-25%**	**-63%**
Stdev	**1.28 [3]**	**1.40 [3]**	**1.38 [3]**	**2.60%**	**12%**	**35%**	**43%**	**48%**

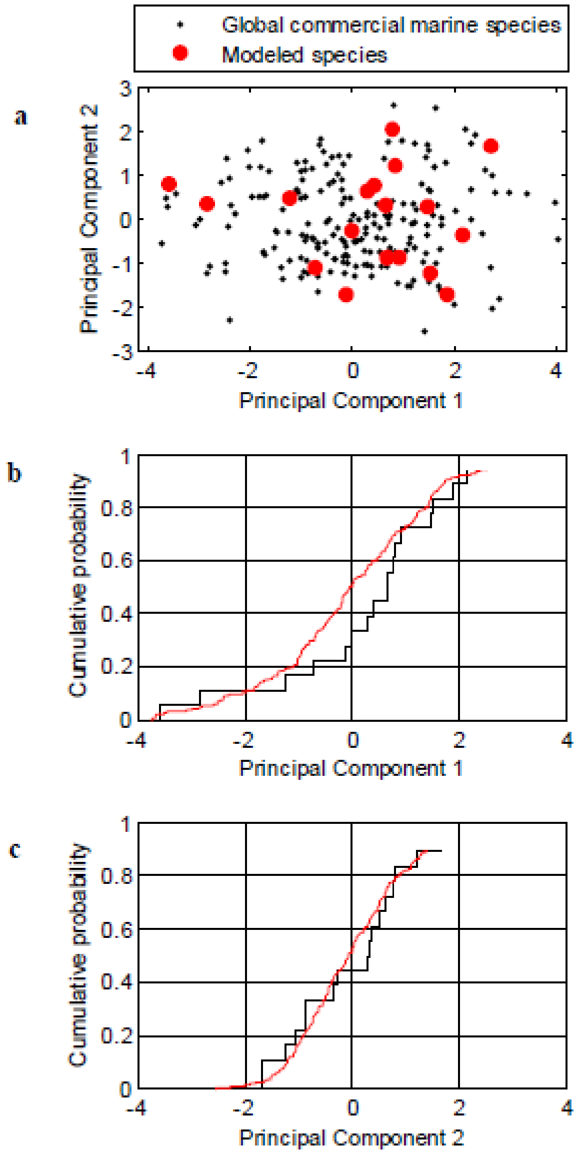
The Change in relative value is calculated excluding fisheries for which it is no longer economically optimal to rebuild (as in OECD report) and including these fisheries (final two columns; fisheries for which it is no longer optimal to rebuild have a 100% decrease in relative rebuilding value)

1. Missing values (.) denote cases in which the economic optimum policy did not result in rebuilding vulnerable stock biomass to at least the Bmsy threshold.

2. Estimated within 1% using the algorithm described in this Addendum. Highest discount rate at which rebuilding to ≥ Bmsy th.reshold is the economically optimal policy.

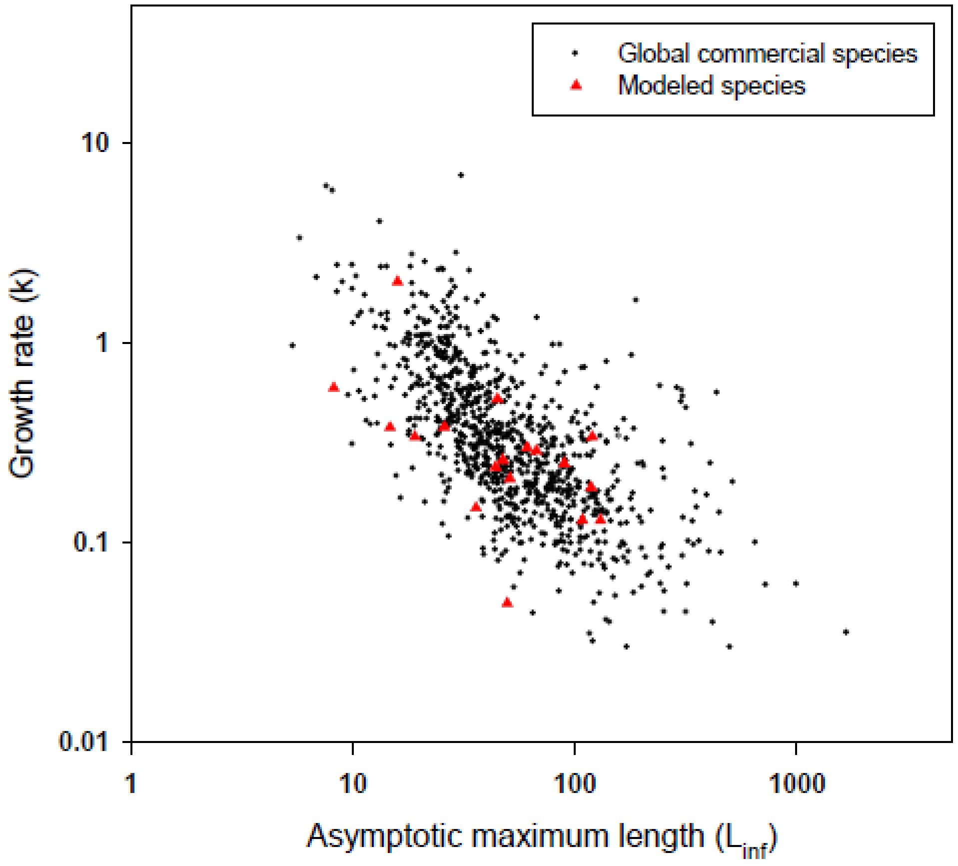
3 Species 2 was excluded from these calculations to limit effect of this species' extremely high relative rebuilding value.

Figure 1. **Principal components analysis (PCA) of representativeness of our modeled species (red)**



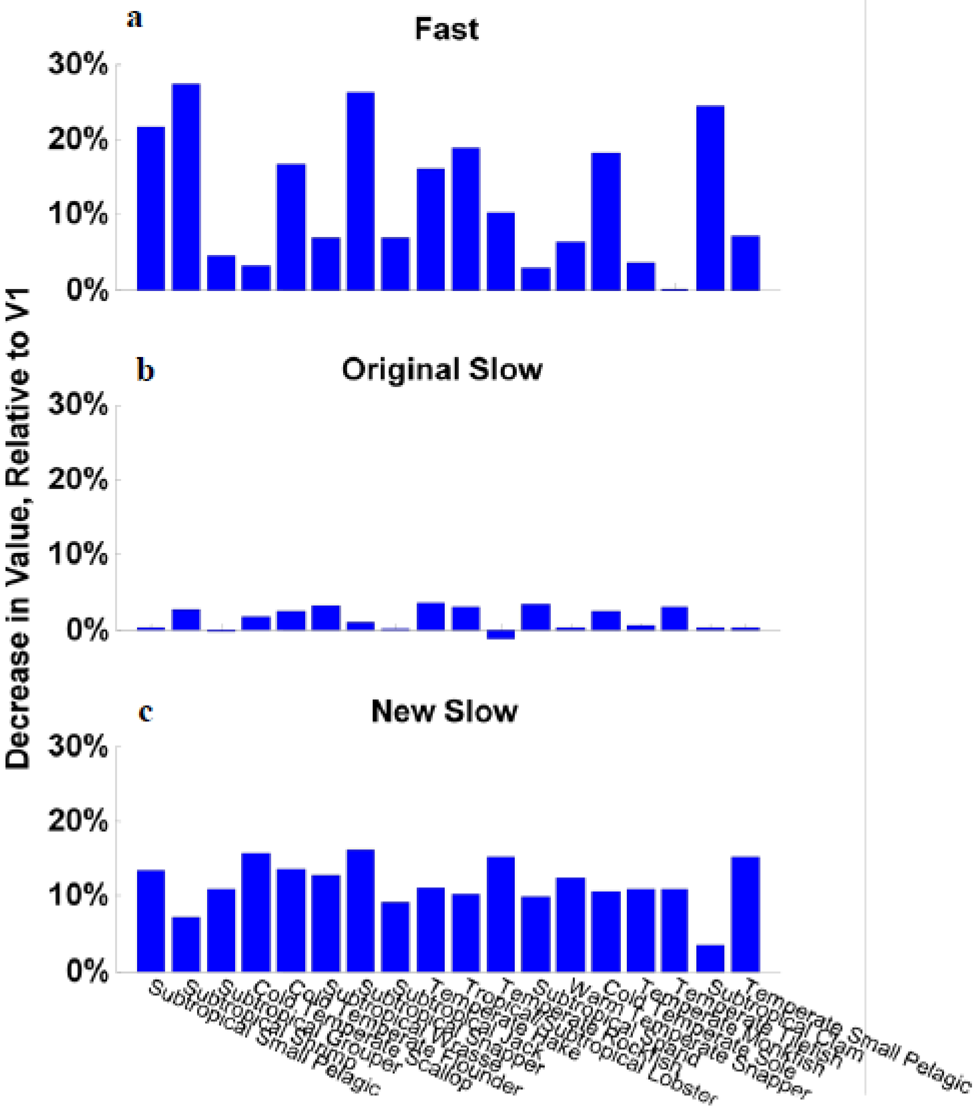
Note: in comparison to the distribution for global commercial marine fishery species worldwide for which four basic life history parameters were available (Linf, k, M, and Lrepro; n=202 species; black). (a) Biplot of leading principal components, which together explained 85% of the variance in these four life history parameters. (b) and (c) Cumulative distribution functions for PC1 and PC2 of subset (red) and global marine fishery species (black). Statistical tests on cumulative distribution functions indicated that subset distributions did not significantly differ from the global distribution (see text for details).

Figure 2. Von Bertalanffy growth rate versus asymptotic maximum (k, Linf)



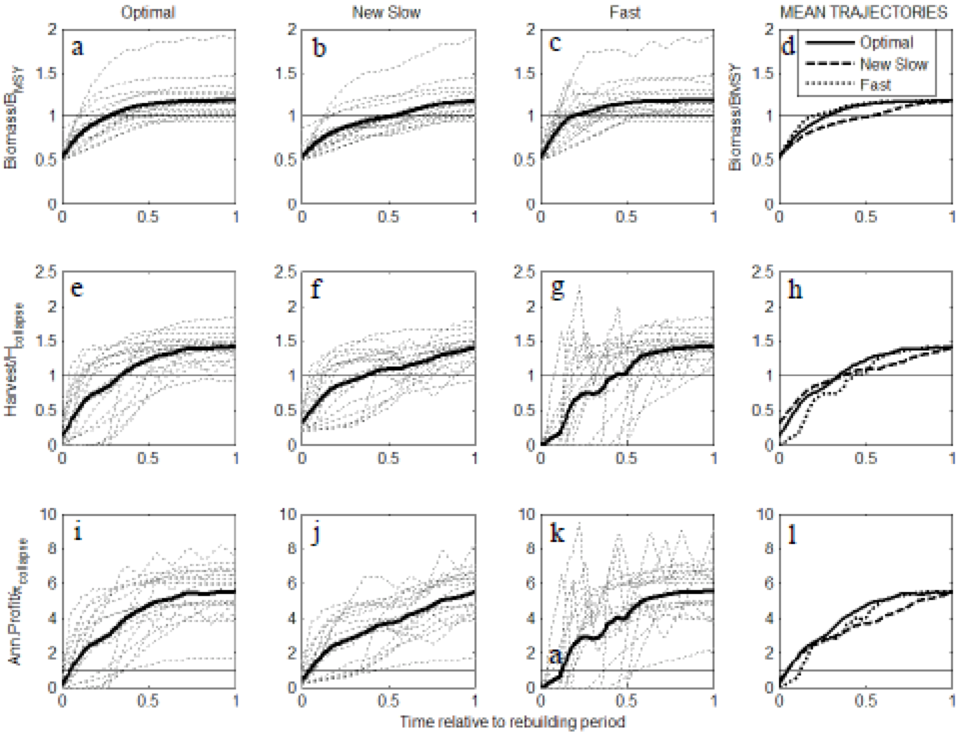
Note: For a broad survey of all commercial marine fishery species worldwide for which k and Linf estimates were available (n=927 species), shown by the black dots, and for the modeled species from the OECD report, shown by red triangles

Figure 3. Reduction in rebuilding value under alternative scenarios (fast, old slow, new slow), as a percentage of the optimal value of rebuilding



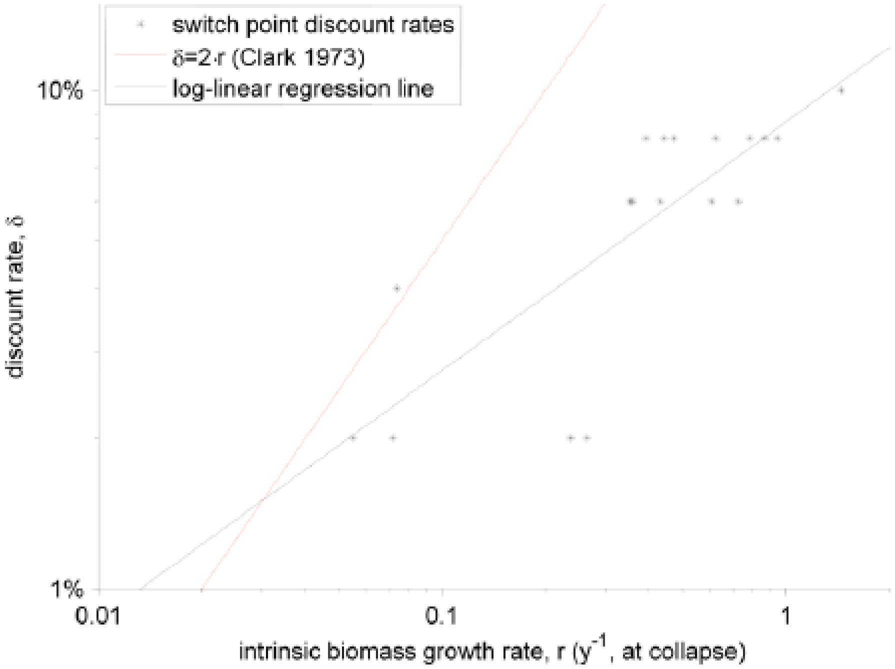
Note: (a) Fast rebuilding policy (100%*[V1opt-V1fast]/V1opt). (b) Original slow rebuilding policy (100%*[V1opt-V1slow]/V1opt). (c) New slow rebuilding policy (100%*[V1opt-V1slow, new]/V1opt). Note that the calculation of these percentages differs somewhat from the calculations in Figure 3 of the original report, to more clearly show how the reductions relate to the optimal value of the fishery.

Figure 4. Dynamics of rebuilding under economically optimal fishing policy (V1opt, 1st column) and alternative policies (fast, new slow)



Note: Dashed grey lines show trajectories from collapsed to rebuilt states for each hypothetical fishery, and solid black line shows the ensemble mean. Time is scaled to the length of the rebuilding period, so that values on the horizontal axis represent the fraction of the rebuilding period that has elapsed. (a,b,c) Stock biomass (Bt) as a fraction of biomass at MSY (BMSY). (d) Ensemble mean biomass trajectories for optimal, fast, and new slow policies compared. (e,f,g) Catch (Ht) relative to harvest in the collapsed state (Hcollapse). (h) Ensemble mean harvest trajectories for optimal, fast, and new slow policies compared. (i,j,k) Annual net profit (πt) relative to the collapsed state (πcollapse). (l) Ensemble mean annual net profit trajectories for optimal, fast, and new slow policies compared. Species 2 is excluded from profit panels because of its extremely high value. Horizontal reference lines are plotted at 1.0.

Figure 5. Switch point discount rates (δ) plotted versus growth rates of vulnerable stock biomass at collapse ®



Note: Intrinsic growth rates were calculated as the natural logarithm of the finite rate of increase of the vulnerable biomass growth curve evaluated at the Bcollapse reference point. Red solid line indicates Clark's (1973) criterion: for the simpler model of fish biomass dynamics he analyzed, $\delta > 2r$ was sufficient to imply that it would be economically optimal to drive the resource to extinction and invest the money, rather than continue to fish. Black dashed line shows the least squares regression fit to log-transformed variables ($\ln(\delta) = 0.50 \ln(r) - 2.45$, $R2=0.64$, $p<0.001$). Note Log10 scale on both axes.