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Can information from marine protected areas be used to inform control-rule-based management of small-scale, data-poor stocks?

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Many small-scale, nearshore fisheries lack the historical catch and survey information needed for conventional stock-assessment-based management. The potential use of the ratio of the density of fish outside a marine protected area to that inside it each year (the density ratio, DR) in a control rule is evaluated to determine the direction and magnitude of change in fishing effort in the next year. Management strategy evaluation was used to evaluate the performance of this DR control rule (DRCR) for a range of movement rates of larvae and adults and other biological scenarios, and the parameters of the control rule that maximized cumulative catch (over 95 years) for each scenario were found. The cumulative catch under the optimal DRCR was 90% of the cumulative catch from an optimal constant effort rule (CER). A small range of parameter values for the DRCR produced 75% or more of the cumulative catch produced from optimal CERs for a variety of assumptions about biology and initial stock status. The optimal DRCR was most sensitive to the movement patterns of larvae and adults and survey variability.

Keywords: control rules, data-poor, fisheries management, management-procedure approach, management strategy evaluation, marine protected areas, nearshore, spatial management.

Introduction

Many nearshore fisheries are difficult to manage because the data required for conventional stock-assessment-based management, such as historical catch, and catch and discard rates, are missing or uncertain (Lleonart and Maynou, 2003; Key *et al.*, 2008). Further difficulties associated with managing nearshore fisheries include problems tracking catches accurately, lack of species-specific catch data, infrequent (or non-existent) fishery-independent survey data, and an uncertain relationship between fishery catch per unit effort (cpue) and abundance owing to changes in the geographic location of fishing effort, the implementation of marine protected areas (MPAs), and other management regulations. Information for these fisheries is commonly aggregated over large spatial areas, although the spatial scale of both biological and physical dynamics and the corresponding spatial extent of fish stocks and fisheries are often much smaller (Gunderson *et al.*, 2006, 2008). For example, the stock assessment for blue rockfish (*Sebastes mystinus*) aggregates data over most of the coastline of California (Key *et al.*, 2008), whereas Jorgensen *et al.* (2006) found that these fish have a home range of 100 m or less. Managing resources at a spatial scale larger than that of the system dynamics means that the fishing intensity on individual populations may be too high in some locations, e.g. locations close to a fishing port, and too low in others, leading to the possibility of localized depletions or forgone catch (Walters and Martell, 2004). The absence of informed management at an appropriate spatial scale leaves nearshore fisheries vulnerable to rapid overexploitation owing to the classic boom-and-bust cycle that follows when a new market is found for a resource (Berkes *et al.*, 2006).

In recent years, MPAs have been implemented in many marine ecosystems to conserve biodiversity, highly vulnerable species, and habitats (Anon., 1979, 1992; HDAR, 1992; GBRMPA, 2004; MLPA, 2008). Modelling studies show that MPAs can be expected to increase biomass in the absence of other fishing regulations for populations that are overexploited and in decline (Holland and Brazee, 1996; Lauck *et al.*, 1998; Gerber *et al.*, 2003). Nevertheless, they do not protect areas that are open to fishing from overexploitation (Horwood *et al.*, 1998; Hilborn *et al.*, 2006; McGilliard and Hilborn, 2008). Moreover, MPAs are often located based on objectives other than those of conventional fisheries management, such as increasing biodiversity, protecting bottom habitat, or achieving socio-economic or political goals, so may not protect a significant proportion of the range of a target species (Ward *et al.*, 1999; Sala *et al.*, 2002; Sorensen and Thomsen, 2009; Semmens *et al.*, 2010). Therefore, management measures are needed in addition to MPAs to minimize the risk of severely overexploiting nearshore resources.

In addition to MPAs, the nearshore rockfish fishery along the California coast, the Australian North West Slope and Western Deepwater Trawl Fisheries, and the Australian Coral Sea Fishery set catch limits as a fraction of the highest historical catch (NFMP, 2002; Dowling *et al.*, 2008). However, the highest historical catch contains little information about the dynamics of a stock.

Moreover, historical catch is often based on and applied to a multispecies assemblage, rather than to a single species. New methods are needed to manage nearshore fish stocks that will ensure sustainable fishing practices at appropriate spatial scales, do not require reliable historical information on the fisheries or unbiased information on current catch, and will provide quality data to inform and improve future management.

We evaluate here a survey-based control rule (the density ratio control rule, DRCR). The density ratio (DR) is the ratio of the fish density outside an MPA to that inside it, based on stratified random sampling. The DR is used as an indicator of stock status where the density inside an MPA is the best available representation of unfished conditions. Unlike a point estimate of unfished biomass from a typical stock assessment, the density inside an MPA is subject to the same fluctuations in environmental conditions as the fished portion of the stock. The DRCR that we evaluate uses the DR in the current year to determine a multiplier specifying a direction and magnitude of change in allowable effort that is implemented in the subsequent year. Fishing effort rather than catch is assumed to be controlled. Effort controls are more common than catch controls in many recreational and artisanal nearshore fisheries. For example, several fisheries are managed by various combinations of gear restrictions, time-area closures, and limited entry rules, including Argentinian scallops, Chilean loco and sea urchin, nearshore Mediterranean fisheries, and assemblages of coral reef species worldwide (McClanahan and Mangi, 2001; Hilborn *et al.*, 2005; Orensanz *et al.*, 2005; Campbell *et al.*, 2008; Little *et al.*, 2009; Morales-Nin *et al.*, 2010).

In this paper, the performance of the DRCR as a management strategy is assessed for small-scale nearshore fisheries, and the DRCR is compared with a constant-effort rule (CER), determining the parameters of the DRCR that maximize cumulative catch over 20, 30, 60, and 95 years. Optimal values for the parameters of the control rule are always unknown because of irreducible uncertainties. Therefore, we evaluate the long-term effects of using a range of non-optimal DRCRs relative to those of non-optimal constant effort strategies. In data-poor situations, successful management strategies need to be robust to uncertainties about biology and stock status, so it is necessary to evaluate the sensitivity of outcomes to a range of assumptions on life history, movement patterns of larvae and adults, initial stock status, spatial heterogeneity in abundance, and alternative sample sizes.

Methods

We used management strategy evaluation (MSE; Punt, 2006) to evaluate the performance of the DRCR. MSE tests strategies for assessing and managing fisheries by (i) simulating the true biology of the natural system (referred to as the operating model, OM), (ii) sampling from the true population, (iii) calculating the measures of stock status (assessment), (iv) calculating recommended fishing restrictions using control rules, and (v) applying updated restrictions to the fishery, which allows the dynamics of the true population to be updated. Although MSE can allow for implementation error, i.e. where the restrictions on the fishery differ from those inferred from the control rule, the analyses conducted here ignore this source of uncertainty, for simplicity.

The DRCR tested is a control rule in which a change in effort for the subsequent year is linearly related to the value of the DR in the current year (Figure 1). The DRCR has two parameters: the *x*-intercept and the slope. The *x*-intercept is the DR at which no change in effort is recommended by the control rule (0.4 in Figure 1), and the slope of the control rule controls the magnitude of the change in allowable effort as a function of the DR.

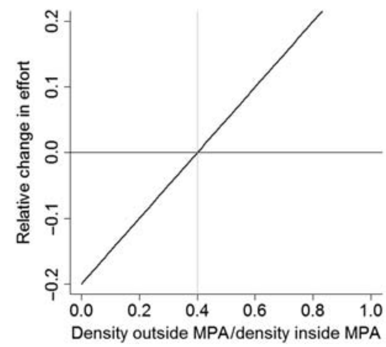

Figure 1. Example of a DRCR. The ratio of the density outside to inside the MPA (*x*-axis) determines the direction and relative amount of recommended change in fishing effort in the following year (*y*-axis). The vertical grey line shows the *x*-intercept of the DRCR.

The OM was age- and space-structured with a fishery taking place in the middle of the year (see the Supplementary material for more detail). It consisted of 30 cells alongshore and 5 cells representing the inshore–offshore direction (Figure 2). Two spatial dimensions are modelled to capture the effects of sampling a population that is distributed heterogeneously over its geographic range. Larvae and adults move during each time-step.

Management strategy

Simulations were initiated with fishing effort at the constant level that brought the fishery from unfished to a specified initial depletion level over 60 years. A single no-take MPA was implemented in the middle six cells in the alongshore direction and spanned the inshore–offshore cells (Figure 2). Management strategies (a DRCR or a CER) were implemented 5 years after the MPA came into effect.

Sampling

Sampling followed a stratified random design with three strata: (i) cells open to fishing and within a distance of three cells from the MPA; (ii) cells open to fishing and farther than three cells from the MPA; and (iii) cells within the MPA (Figure 2). Fished areas closest to the MPA are expected to have higher densities of fish and therefore will be subject to more fishing effort. Dividing the fished area into two strata is expected to lower survey variance

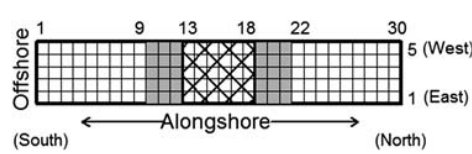

Figure 2. The spatial configuration of the OM. A no-take MPA is implemented in year 1 in cells 13–18 alongshore and 1–5 in the offshore direction (black hatched). Stratified random sampling occurs in three strata: the MPA (black hatched), the open areas near the MPA (grey), and the remaining open areas (white). The labels east, west, north, and south denote the orientation of directional larval movement.

and ensures that both strata are sampled each time there is a survey. Within strata, cells were selected for sampling randomly without replacement each year. In the base-case scenario, one-eighth of the cells of each stratum were sampled. Samples within each surveyed cell were represented by the true abundance before removals in the middle of the year, modified by survey selectivity, which was assumed to be the same as fishery selectivity, and subject to the lognormal observation error with a standard error of the log σ_{survey}.

Assessment

Stock status was determined as the ratio of the sampled density in fished areas, $\tilde{D}_{\text{open}_t}$, to that in the MPA, $\tilde{D}_{\text{closed}_t}$, where the simulated sample densities were

$$\tilde{D}_{\text{open}_t} = \left(\frac{\tilde{N}_{\text{near}_t} + \tilde{N}_{\text{far}_t}}{n_{\text{near}_t} + n_{\text{far}_t}}\right); \quad \tilde{D}_{\text{closed}_t} = \frac{\tilde{N}_{\text{closed}_t}}{n_{\text{closed}_t}}, \qquad (1)$$

where n_{near_t}, n_{far_t}, and n_{closed_t} were the number of cells open to fishing (near and far from the MPA) and in the MPA, respectively; and $\tilde{N}_{\text{near}_t}$, $\tilde{N}_{\text{far}_t}$, and $\tilde{N}_{\text{closed}_t}$ were the sampled number of fish in each stratum. The DR, $\tilde{\rho}_t$, was $\tilde{D}_{\text{open}_t}/\tilde{D}_{\text{closed}_t}$.

Control rule

The control rule was a linear function where the change in effort from year t to year $t+1$ (ΔE_{t+1}) was a function of the DR ($\tilde{\rho}_t$), an x-intercept, and a slope (Figure 1).

$$\Delta E_{t+1} = \text{slope}(\tilde{\rho}_t - x\,\text{intercept}); \quad \text{i.e. } E_{t+1} = E_t + \Delta E_{t+1}. \qquad (2)$$

A grid of control-rule parameters consisting of 20 x-intercepts over the range [0,1] and 20 slopes over the range [0,4] was evaluated to identify the "optimal" set of parameters, i.e. those that produced the maximum cumulative catch summed over 95 years of simulation and 50 sets of random deviates. In addition, control-rule parameters were found that maximized cumulative catch summed over 20, 30, and 60 years, and 50 sets of random deviates.

CERs as an alternative to the control rule

CERs were evaluated as a reference for comparison with DRCRs. Simulations were conducted to determine the cumulative catch when a single effort level was applied for the entire duration of a simulation. Simulations were conducted for each of 200 effort levels (E where $E_t = E$ for all t), for the same 50 sets of random deviates used for testing the DRCR. The optimal constant effort levels that produced the maximum cumulative catch over 20, 30, 60, or 95 years were identified.

Base-case OM

The optimal values for the control-rule parameters were found for a base-case OM, which had an initial depletion (spawning-stock biomass, SSB, divided by unfished equilibrium SSB) equal to 25% of that at maximum sustainable yield (MSY). The initial effort level ($E_{t=0}$) was the constant effort level required to achieve a depletion level of 25% of that at MSY over 60 years, starting at MSY conditions. Larvae and adults were sedentary, and larvae experienced density-dependent mortality locally in each cell. The base-case OM accounted for process and observation uncertainty by including local and global recruitment variation with a total recruitment variance of 0.72, and $\sigma_{\text{survey}} = 0.2$. Additional base-case OM conditions are listed in Supplementary Table S1.

Sensitivity analyses

Sensitivity analyses explored the extent to which initial stock size, biological parameters, and sampling error influencing which DRCRs and CERs were optimal, and the potential consequences of using suboptimal control rules. Sensitivity analyses examined initial depletion levels of 10, 50, 100, and 200% of the depletion corresponding to MSY, alternative steepness (h; Francis, 1992) levels of 0.3, 0.5, and 0.9, natural mortality (M) values of 0.05, 0.2, and 0.4 year^{-1}, survey *CV*s of 0.1, 1, and 1.5, and spatial *CV*s of catchability of 0.2 and 0.42. Sensitivity analyses were selected to represent a range of uncertainties common to nearshore fisheries where few data exist and few analyses have been performed to determine initial stock status and biological parameters. In addition, survey *CV*s may vary as a consequence of the patchy distribution of fish, the small scale of nearshore fish populations, and the small sample sizes caused by funding constraints.

Several scenarios were conducted to examine the impact of assumptions on movement of larvae and adults (Tables 1 and 2; the phrases in these tables are used as abbreviations for the scenarios). Table 1 describes scenarios in which larvae experienced density-dependent mortality after larval diffusion and Table 2 those in which density-dependent mortality applied to the entire larval pool, the survivors of which then recruited to a diffuse patch of spatial cells. The scenarios in Table 2 captured the hypothesis that density-dependent mortality is caused by global environmental conditions such as food availability during a planktonic stage (Hjort, 1914; Cushing, 1990), rather than habitat availability for juveniles at the time of settlement, as for the scenarios in Table 1 (Myers and Cadigan, 1993a, b). The scenario "recruitment to inside the MPA and adult diffusion" (Table 2) was similar to the scenario "recruitment to outside the MPA and adult diffusion", except that larvae recruited to a patch of cells inside the MPA.

Performance measures

The main performance measure was the cumulative catch (over 95 years and 50 sets of random deviates) relative to that under the optimal CER. Other performance measures were the cumulative catch over 20, 30, and 60 years, the probability of falling below

Table 1. Descriptions and parameter values for sensitivity tests of larval and adult movement for scenarios with local (within-cell), post-dispersal density-dependent mortality; (σ_{Lalong}, σ_{Loff}) represents the extent of larval diffusion in the north–south and east–west directions, respectively, and (σ_{Aalong}, σ_{Aoff}) represents the same for adult diffusion.

Description	Larval diffusion (σ_{Lalong}, σ_{Loff})	Adult diffusion (σ_{Aalong}, σ_{Aoff})
Short-distance larval diffusion	(1,0.5)	(0,0)
Medium-distance larval diffusion	(5,0.5)	(0,0)
Long-distance larval diffusion	(10,0.5)	(0,0)
Short-distance larval and adult diffusion	(1,0.5)	(1,0.5)
Long-distance larval and short-distance adult diffusion	(10,0.5)	(1,0.5)
Medium-distance larval and adult diffusion	(5,0.5)	(5,0.5)

Table 2. As for Table 1, but for scenarios with global (pooled) density-dependent mortality; σ_{Lnorth}, σ_{Lsouth}, σ_{Least}, and σ_{Lwest} are the standard deviations of larval movement in each cardinal direction, and σ_{Aalong} and σ_{Aoff} represent the extent of adult diffusion in the north–south and east–west directions, respectively.

Description	Central cell of larval settlement	Alongshore spatial spread of larvae around central cell (σ_{Lnorth}, σ_{Lsouth})	Offshore spatial spread of larvae around central cell (σ_{Least}, σ_{Lwest})	Adult diffusion (σ_{Aalong}, σ_{Aoff})
Recruitment to outside the MPA and adult diffusion	(6,2.5)	(7,7)	(7,7)	(3,0.5)
Recruitment to inside the MPA and adult diffusion	(15.5,2.5)	(7,7)	(7,7)	(3,0.5)

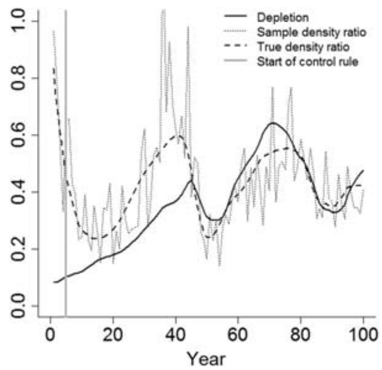

Figure 3. An example realization of the DRCR for the base-case scenario. An MPA is implemented in year 0, and the DRCR with (x-intercept, slope) = (0.42, 1.05) in year 5 (vertical grey line). Depletion (solid black line), the true DR with no observation error (heavy dashed line), and the sample DR (light dotted line) are shown.

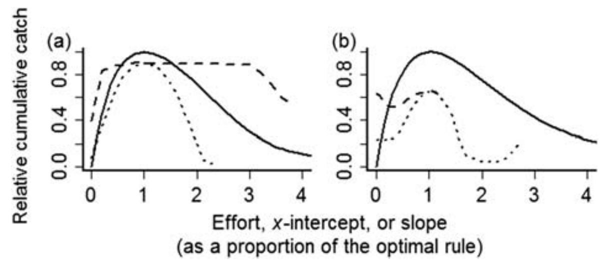

Figure 4. Profiles of cumulative catch calculated over (a) 95 years and (b) 30 years (scaled to the cumulative catch of the optimal CER) over CERs (solid lines), and x-intercepts (dotted lines) and slopes (dashed lines) under DRCRs for the base-case scenario. For the profile over x-intercepts, the slope is fixed at its value for the optimal DRCR [optimal slope was (a) 1.05 and (b) 3.37)], for the profile over slopes, the x-intercept is fixed at its value for the optimal DRCR [optimal x-intercept was (a) 0.42 and (b) 0.37)].

25% of B_{MSY}, average and 5 and 95% quantiles of the *CV* of catch and cpue interannually and among simulations in year 100 (termed "intersimulation *CV*"), average depletion in year 100, and the average catch and cpue in year 100 relative to that at MSY. For each scenario, the optimal values of x-intercept and slope were found and used to assess whether the optimal set of values defining the control rule was similar across biological and sampling scenarios, and hence the potential for the same set of values to be used for more than one scenario.

In addition, the parameter values for a max–min DRCR (defined below) and CER were identified based on all scenarios (the base-case OM and all the sensitivity analyses). The max–min rule was the DRCR (x-intercept and slope) or CER (effort) that maximized the lowest cumulative catch across all scenarios. More specifically, the max–min rule was found by (i) identifying, for each parameter value (for CERs) or each combination of parameter values (for DRCRs), the scenario that led to the lowest cumulative catch, and then (ii) finding the parameter value or combination of parameter values for which the lowest cumulative catch was greatest. The max–min rule minimized potential losses assuming that nothing was known about the biology, survey *CV*, or initial status of the stock being managed.

Results

Base-case scenario

The base-case scenario resulted in an optimal DRCR in which (x-intercept, slope) = (0.42, 1.05). An example realization of the base-case scenario using the optimal DRCR (Figure 3) shows that initially there was little difference between densities inside and outside the MPA because of the similar fishing history before MPA implementation. The true and sampled DR began to follow the true trend in depletion after 15–20 years with no fishing in the MPA. The sampled DR was sometimes very different from the true depletion level, but major shifts in true depletion levels were captured in sample DRs (Figure 3).

Profiles of cumulative catch calculated over 30 years for a range of parameter values are influenced by transient dynamics such as stochasticity and oscillations in dynamics after the control rule is implemented (Figure 4b). After 95 years, the length of the catch series is sufficient to identify the parameter values that are optimal without the substantial influence of transient dynamics (Figure 4a). The parameter space producing high cumulative catches changes depending on the period over which the control rule is evaluated, but converges after ~60 years (Figure 5). Results for shorter periods (i.e. 5–10 years) are not shown in Figure 5 because the optimal DRCR caused the population to collapse quickly. Over 30 years, the optimal DRCR produced ~60% of the cumulative catch produced by the optimal CER (Figure 4b). In contrast, over 95 years, the optimal DRCR produced 90% of the cumulative catches produced by the optimal CER (Figure 4a). We focus on the cumulative catch over 95 years for the balance of this paper to avoid confounding optimality with the effects of transient dynamics.

The cumulative catch over 95 years was insensitive to the value of the DRCR slope (Figures 4a and 5d). With the DRCR x-intercept fixed at its optimal value, the cumulative catch was maintained near the maximum for slopes ranging from 20 to

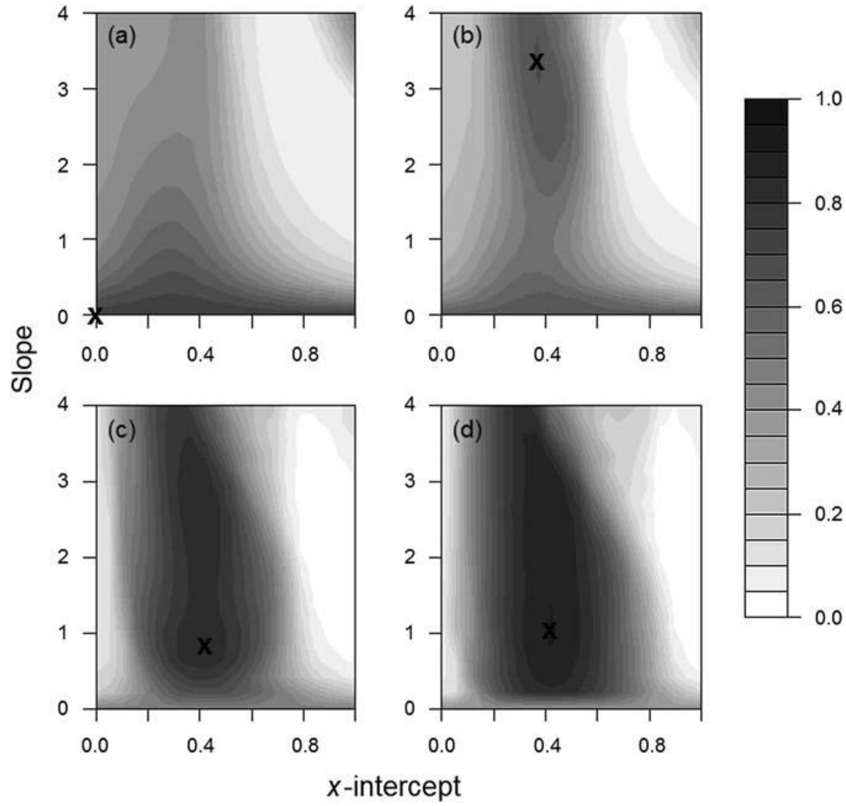

Figure 5. Profiles of cumulative catch calculated over (a) 20, (b) 30, (c) 60, and (d) 95 years over a range of *x*-intercepts and slopes relative to the cumulative catch for the optimal CER. The maximum cumulative catch is marked with a black cross.

300% of the optimal slope (Figure 4a). In contrast, the cumulative catch was highly sensitive to the *x*-intercept (Figure 4a). Cumulative catches under constant-effort policies were maintained at high values for a fairly wide range of effort levels (Figure 4a).

Interannual variability in catch under DRCRs was smallest when a small slope was used and for *x*-intercepts close to the optimal *x*-intercept (Figure 6a). The probability of falling below 25% of B_{MSY} was greatest when the slope was large and the *x*-intercept was small relative to the optimal *x*-intercept (Figure 6b).

Sensitivity analyses

Fish movement

The optimal DRCR *x*-intercept and slope were most sensitive to movement patterns of larvae and adults (Table 3, Figure 7a). The scenarios with short-, medium-, and long-distance larval diffusion only had a small effect on the optimal DRCR (Table 3, Figure 7a), but the optimal *x*-intercept increased as diffusion rates of adults increased (Table 3, Figure 7a). The *x*-intercept was larger for higher diffusion rates of adults because adult diffusion dampened the magnitude of difference between fished areas and MPAs. In addition, catches in year 100 were greater for higher diffusion rates of adults (Table 3).

The optimal *x*-intercept was very large (0.79) for the scenario with recruitment to outside the MPA and adult diffusion. More fish (especially young fish that were newly available to the survey catch) were concentrated in fished areas for this scenario, causing DRs to be higher. Hence, a larger *x*-intercept was required to limit fishing effort (Table 3, Figure 7a).

Observation error

The optimal *x*-intercept increased as survey *CV* increased, forcing decreases in effort over a wider range of DRs (Table 3, Figure 7b). Optimal slope decreased as survey *CV* increased, indicating that the largest cumulative catches were produced when changes in effort were small as the quality of data declined (Table 3).

Initial stock status

The initial size of the stock relative to B_{MSY} before the implementation of the DRCR did not affect the optimal *x*-intercept (Supplementary Table S2). Optimal slopes were largest when the stock was initially overfished or very lightly fished (150–200% of depletion at MSY; Supplementary Table S2). The optimal slope was 0 when the population was initially at MSY (Supplementary Table S2; the long-term effort level was already optimal, so changes in effort only lowered long-term cumulative catch). When initial stock size was small, a greater decrease in effort caused the population to recover to biomass levels and productivity close to that at MSY more rapidly. Likewise, when stocks were initially lightly fished, a control rule that allowed large increases in effort behaved like a constant escapement policy, reducing the population size quickly to MSY conditions, and maximizing productivity by reducing the effects of density-dependent mortality.

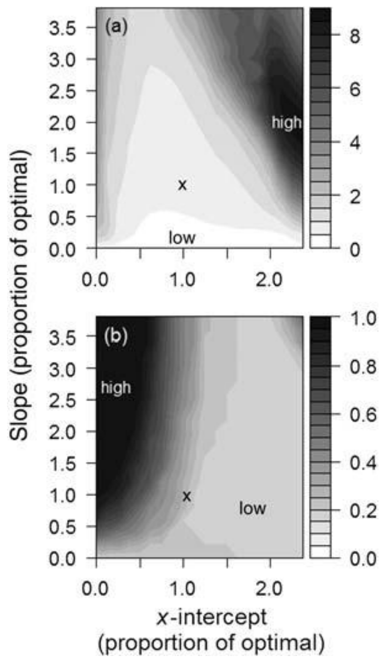

Figure 6. Two-dimensional profile of (a) the interannual variation (*CV*) of the catch (averaged over simulations), and (b) the probability of falling below 25% of B_{MSY} during any time-step after year 5 (the time of control-rule implementation) over values for *x*-intercepts (as a proportion of the optimal *x*-intercept) and slopes (as a proportion of the optimal slope) of DRCRs for the base-case scenario. The optimal DRCR is marked with a black cross.

Steepness and natural mortality

The optimal slope was 0 at very low steepness ($h = 0.3$; Supplementary Table S2) because the stock was very sensitive to changes in effort and even a small increase in effort resulted in long recovery times; it was better to use an effort level that did not result in population collapse over 60 years of fishing. The optimal slope increased with steepness and the optimal *x*-intercept decreased, so it was optimal to make large changes in effort and to allow increases in effort at lower DRs for a stock with a steepness of $h = 0.9$ (Figure 7c, Supplementary Table S2). Such a stock is very productive, so higher fishing levels can be sustained.

The optimal *x*-intercept increased with increasing values for natural mortality (Figure 7d, Supplementary Table S2). The population was spread over a wide range of age classes when natural mortality was low (e.g. $M = 0.05$ year^{-1}), so the proportion of mature fish for a particular population level was larger than for a population with high rates of natural mortality (e.g. $M = 0.3$ or 0.4 year^{-1}). Therefore, a population with low natural mortality can be fished relatively harder and produces more eggs than if it were fished at the same level with higher natural mortality. Optimal slope was larger for populations with low rates of natural mortality than for populations with high rates; making large reductions in effort on fish populations with higher natural mortality results in lost catches that cannot be taken later, because fish that are not caught are more likely to succumb to natural mortality. Typically, age-at-maturity is higher for fish with low rates of natural mortality; changing the age-at-maturity concurrently with changes in natural mortality rates would likely dampen the effects of changing the rates of natural mortality.

Max–min rules

The max–min DRCR had a larger *x*-intercept and a smaller slope than the optimal DRCR for the base-case scenario (Table 4). The max–min DRCR lowered the probability of falling below 25% of B_{MSY} for most scenarios, except three scenarios ($h = 0.3$, medium-distance larval and adult diffusion, and recruitment to outside the MPA and adult diffusion) where the optimal *x*-intercepts were larger than the *x*-intercept for the max–min DRCR (Table 4, Supplementary Table S3). In most scenarios, catches were smaller and cpue higher for the max–min DRCRs than for the optimal DRCRs (Table 4, Supplementary Table S3). The max–min effort rule was based on a slightly lower level of effort than the optimal CER for the base-case scenario and led to lower probabilities of falling below 25% of B_{MSY} and larger catches than the DRCR, except where h was 0.3.

Discussion

A DRCR may be a viable strategy for managing fish stocks for which the data needed to use conventional stock assessment tools are unavailable, and when catches are uncertain. A DRCR with an *x*-intercept of 0.4–0.5 and many values for the slope produced a form of so-called pretty good yield (PGY; Hilborn, 2010), defined here as 75% or more of the cumulative catch over 95 years produced by the optimal CER for a variety of assumptions about fish biology and initial stock status. Therefore, a DRCR has the potential to produce high cumulative catches (PGY) when biological information on fish stocks is uncertain, except for the most extreme life histories. Setting the *x*-intercept just above its optimal value reduced the probability of falling below 25% of B_{MSY} while maintaining high cumulative catches. The slope of the DRCR did not have a great effect on cumulative catch, but smaller slopes minimized the probability of falling below 25% of B_{MSY}, lowered interannual variation in catch and cpue, and widened the range of *x*-intercepts over which the cumulative catch remained high. CERs may outperform DRCRs, assuming that the correct level of effort can be identified.

The MPA and reference points

The population both inside and outside the reference MPA may be affected in the same way, causing no change in the DR during a period of high natural mortality or low productivity as a result of environmental conditions. In this way, the DRCR is subject to an additional risk because fishing effort will not be lowered immediately in response to declines in productivity. However, a year of successful recruitment throughout the range of a stock would also not change the DR and fishing effort would not be increased in response to the increased recruitment (although catch would increase because of greater abundance). Steepness and natural mortality did not change the parameter values of the optimal DRCR, nor those producing large cumulative catches (Figure 7c and d); this means that the same parameter values should be appropriate if there were changes in productivity or natural mortality. However, future studies should explore the implications of using the DRCR in the presence of shifts in productivity and natural mortality.

The DRCR will not be effective when an MPA is new because the DR does not mimic the true depletion level then. For example, a DRCR that is implemented before enough build-up of biomass within the MPA will result in a management recommendation that effort be increased for fisheries that are severely

Table 3. Performance measures for scenarios using optimal DRCRs and CERs.

Description	Optimal *x*-intercept	Optimal slope	Optimal effort relative to optimal effort for the base case	Probability of $B < 0.25\ B_{MSY}$ after time-step 5	Average depletion in year 100	Average cpue in year 100 relative to cpue at MSY	Average catch in year 100 relative to MSY	Average interannual *CV* of catch (5%, 95% quantiles)
Base case								
CER	–	–	1.00	0.00	0.45	0.90	0.76	0.22 (0.15, 0.29)
DRCR	0.42	1.05	–	0.28	0.44	0.86	0.72	0.62 (0.45, 0.82)
Short-distance larval diffusion only								
CER	–	–	1.09	0.02	0.43	0.84	0.78	0.22 (0.15, 0.29)
DRCR	0.42	0.84	–	0.26	0.43	0.84	0.78	0.54 (0.40, 0.70)
Medium-distance larval diffusion only								
CER	–	–	1.18	0.02	0.41	0.82	0.83	0.22 (0.15, 0.29)
DRCR	0.42	0.84	–	0.30	0.39	0.76	0.80	0.42 (0.32, 0.55)
Long-distance larval diffusion only								
CER	–	–	1.27	0.02	0.39	0.79	0.85	0.22 (0.16, 0.29)
DRCR	0.47	0.84	–	0.26	0.40	0.82	0.82	0.41 (0.31, 0.53)
Short-distance larval and adult diffusion								
CER	–	–	1.27	0.02	0.35	0.84	0.90	0.22 (0.15, 0.29)
DRCR	0.53	1.26	–	0.52	0.31	0.74	0.89	0.53 (0.39, 0.67)
Long-distance larval and short-distance adult diffusion								
CER	–	–	1.36	0.06	0.33	0.81	0.92	0.22 (0.15, 0.29)
DRCR	0.58	2.11	–	0.54	0.31	0.77	0.96	0.76 (0.56, 0.98)
Medium-distance larval and adult diffusion								
CER	–	–	1.27	0.06	0.31	0.88	0.95	0.22 (0.14, 0.29)
DRCR	0.79	0.21	–	0.80	0.29	0.82	0.88	0.25 (0.15, 0.39)
Recruitment to outside the MPA and adult diffusion								
CER	–	–	1.00	0.02	0.32	0.89	0.93	0.22 (0.14, 0.29)
DRCR	0.95	1.05	–	0.74	0.34	0.94	0.96	0.57 (0.45, 0.71)
Recruitment to inside the MPA and adult diffusion								
CER	–	–	1.36	0.06	0.30	0.81	0.97	0.22 (0.14, 0.29)
DRCR	0.47	3.16	–	0.68	0.26	0.70	0.99	0.82 (0.63, 1.04)
Survey *CV* = 0.1								
CER	–	–	1.00	0.00	0.45	0.90	0.76	0.22 (0.15, 0.29)
DRCR	0.42	1.89	–	0.26	0.46	0.91	0.71	0.63 (0.48, 0.85)
Survey *CV* = 1								
CER	–	–	1.00	0.00	0.45	0.90	0.76	0.22 (0.15, 0.29)
DRCR	0.79	0.21	–	0.28	0.43	0.82	0.64	0.94 (0.43, 1.97)
Survey *CV* = 1.5								
CER	–	–	1.00	0.00	0.45	0.90	0.76	0.22 (0.15, 0.29)
DRCR	0.89	0.42	–	0.62	0.40	0.70	0.29	2.80 (1.38, 5.14)

overexploited. Wilson *et al.* (2010) suggest a way to adjust for this when using information from an MPA as a reference point; further simulation testing would be necessary to determine whether this correction factor is appropriate in the absence of knowledge regarding initial depletion. Nevertheless, established MPAs exist worldwide and they could, in principle, be used for management with DRCRs (HDAR, 1992; McClanahan and Mangi, 2001; GBRMPA, 2004).

Multispecies perspective

Most nearshore fisheries target an assemblage of species, so a need exists for multispecies management tools. This study considered single-species scenarios, but offers insights about the performance of the DRCR in a multispecies context. A multispecies DRCR could calculate the DR for each species and use an arithmetic or geometric mean of the ratio of the DR to the *x*-intercept for each species as input to the control rule, rather than the DR itself. This would allow the use of different *x*-intercepts for different species. DRCRs could be combined with additional regulations for species that do not follow the general trend in DR across species, because abundance does not increase notably for all species within an MPA (Bohnsack *et al.*, 2003). In some instances, effort could be lowered for species with the lowest DR:*x*-intercept ratios through non-retention of those species on particular days, seasons, or in particular fishing areas.

Implementing a DRCR

The DRCR is effort-based; effort-based management can be used when catches are difficult to track. However, increases in the efficiency of fishing gear and new technology cause increases in effort for which there is no accounting under a CER. A control rule that adjusts allowable effort based on continually updated information on stock status will diminish the effects of effort creep by continually lowering effort, unless the DR is high enough. A DRCR could be created to manage catch rather than effort, even when the magnitude of unreported catch is uncertain, particularly where legal constraints require defined catch limits (MSFCMA, 2007). Further analysis would be needed to assess the performance of a catch-based DRCR.

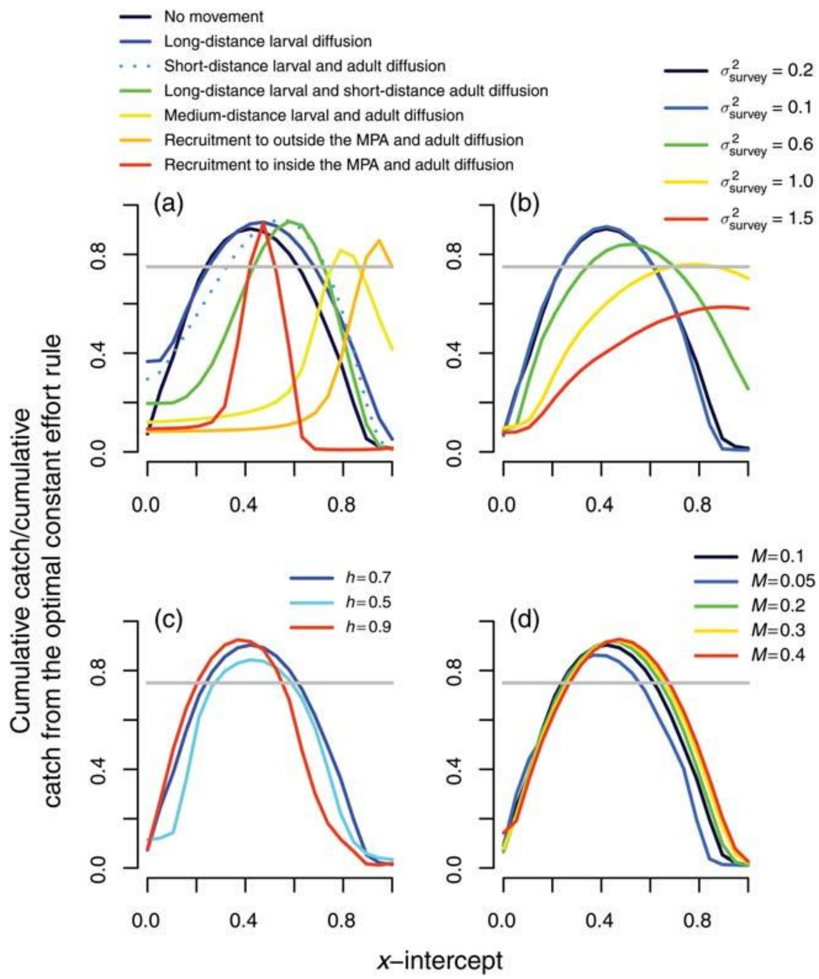

Figure 7. Cumulative catch (scaled to the cumulative catch of the scenario-specific optimal CER) for values of the *x*-intercept of the DRCR for (a) movement patterns of larvae and adults, (b) survey CVs (σ^2_{survey}), (c) steepness (*h*), with $h = 0.3$ omitted because the optimal slope was 0, so all *x*-intercepts produced the same cumulative catch, and (d) natural mortality (*M*). For each line (scenario), the slope of the DRCR was fixed at the value for the scenario-specific optimal DRCR. The horizontal grey line shows 75% of the cumulative catch under the optimal CER for each scenario.

Another challenge for fisheries management is that many nearshore fish stocks with low rates of movement and local, port-based fisheries are currently managed at a coast-wide spatial scale; localized depletion and forgone catch are risks when management is at a larger spatial scale than that of the fish stocks and fisheries. The DRCR is a potential tool for managing fish stocks at the spatial scale at which the population and fishery dynamics occur. The DRCR can be used without a high level of quantitative expertise, so is a candidate for use at a local level as part of community-based monitoring and fishery management.

Finally, implementation of a DRCR would benefit from some knowledge of larval dispersal and adult movement to determine an appropriate *x*-intercept. Exploration of movement patterns of larvae and adults is an active area of research, and much information exists for many nearshore species (e.g. Jorgensen *et al.*, 2006; Hyde and Vetter, 2009). Changing the rate of diffusion is expected to have similar effects on optimal parametrization of the DRCR as changing the size of the MPA or allowing fishing mortality within the MPA. Therefore, successful implementation of the DRCR would benefit from careful consideration of the size of the MPA relative to movement rates and the rate of fishing mortality expected within the MPA.

Potential for future work

Several changes could be made to the DRCR.

(i) First, many large and sudden changes in effort regulations could be destabilizing for fishing communities, so future studies should find ways to minimize the variance in sample DRs, which would in turn lower variability in effort recommendations generated by the control rule (Figure 3). For example, time-averaged DRs, e.g. averaging the DR over the previous 3 years, could be used as an input to the DRCR. Further, the use of a more sophisticated survey design could lower the variance among years in sample DRs. Some management systems do not allow conventional surveys within MPAs, and future studies should explore the implications of basing the DR on non-extractive surveys.

Table 4. Performance measures for scenarios using the max–min CER and DRCR (relative to those using the optimal rules for all performance measures except "probability of $B < 0.25B_{MSY}$ after time-step 5").

Description	Max–min *x*-intercept relative to optimal *x*-intercept	Max–min slope relative to optimal slope	Max–min effort relative to optimal effort	Probability of $B < 0.25\,B_{MSY}$ after time-step 5	Average depletion in year 100 relative to that for the optimal rule	Average cpue in year 100 relative to cpue for the optimal rule	Average catch in year 100 relative to that for the optimal rule	Average interannual *CV* of catch relative to that for the optimal rule
Base case								
CER	–	–	0.91	0.00	1.06	1.10	1.00	1.02
DRCR	1.75	0.20	–	0.16	1.70	2.27	0.78	0.59
Short-distance larval diffusion only								
CER	–	–	0.83	0.00	1.11	1.20	1.00	1.04
DRCR	1.75	0.25	–	0.16	1.71	2.27	0.73	0.63
Medium-distance larval diffusion only								
CER	–	–	0.77	0.00	1.16	1.28	0.98	1.04
DRCR	1.75	0.25	–	0.16	1.76	2.34	0.76	0.64
Long-distance larval diffusion only								
CER	–	–	0.71	0.00	1.22	1.36	0.97	1.04
DRCR	1.56	0.25	–	0.16	1.67	2.08	0.77	0.62
Short-distance larval and adult diffusion								
CER	–	–	0.71	0.00	1.30	1.36	0.97	1.07
DRCR	1.40	0.17	–	0.36	1.78	1.96	0.91	0.56
Long-distance larval and short-distance adult diffusion								
CER	–	–	0.67	0.00	1.38	1.44	0.96	1.07
DRCR	1.27	0.10	–	0.38	1.54	1.63	0.90	0.35
Medium-distance larval and adult diffusion								
CER	–	–	0.71	0.00	1.36	1.35	0.97	1.07
DRCR	0.93	1.00	–	0.90	0.46	0.47	0.81	0.95
Recruitment to outside the MPA and adult diffusion								
CER	–	–	0.91	0.02	1.10	1.10	1.00	1.02
DRCR	0.78	0.20	–	1.00	0.01	0.01	0.04	1.59
Recruitment to inside the MPA and adult diffusion								
CER	–	–	0.67	0.00	1.45	1.45	0.96	1.08
DRCR	1.56	0.07	–	0.50	3.18	3.19	0.32	0.28
Survey *CV* = 0.1								
CER	–	–	0.91	0.00	1.06	1.10	1.00	1.02
DRCR	1.75	0.11	–	0.16	1.69	2.20	0.74	0.54
Survey *CV* = 1								
CER	–	–	0.91	0.00	1.06	1.10	1.00	1.02
DRCR	0.93	1.00	–	0.34	0.95	0.90	0.99	0.98
Survey *CV* = 1.5								
CER	–	–	0.91	0.00	1.06	1.10	1.00	1.02
DRCR	0.82	0.50	–	0.60	0.76	0.50	1.53	0.70

The max-min DRCR was (x-intercept, slope) = (0.74, 0.21); the max-min CER was 0.91 of the optimal constant effort for the base-case scenario.

(ii) This study used cumulative catch to evaluate the performance of control rules. Control rules could be analysed using other performance measures, depending on fishery objectives (Quinn and Deriso, 1999; Deroba and Bence, 2008), such as those related to fluctuations in catch, cumulative net profits (Clark, 1973), or differences between current depletion and a target depletion level (Deroba and Bence, 2008).

(iii) As historical data accumulate, it may be more effective to carry out conventional quantitative stock assessments than to continue to use the DRCR. Future work should explore whether this is true, and if so, find the average number of years required to produce a stock assessment containing better quality information than the DR.

(iv) Other ratios than that based on density could be used, for example, to capture more information on age or length structure, fecundity, or maturity. For instance, a ratio could be formulated based on a rough estimate of spawning biomass-per-recruit or lifetime egg production (O'Farrell and Botsford, 2005).

(v) The control rule could be implemented based on fishery cpue data, rather than survey data. Cpue data are often biased and are not linearly related to density, but often more cpue data exist than could be gathered using a small survey. Future analyses that consider the use of cpue data could examine whether or not the use of biased cpue data is better than using an unbiased, but smaller, survey.

(vi) Only linear control rules were considered in this study; non-linear control rules may have desirable properties. For instance, a control rule that has constraints on the maximum and minimum extent of change in effort in a given year may produce more stable catches and could lessen the probability of falling below a biomass threshold by not allowing very large increases in effort.

(vii) Other data-poor methods of providing fisheries management advice exist and are in various stages of development (O'Farrell and Botsford, 2005; Kai and Shirakihara, 2008; Cope and Punt, 2009; MacCall, 2009; Wilson *et al.*, 2010). A comparative analysis of data-poor methods, including the approach developed here, would illustrate which methods are likely to perform best given the shortcomings of data and the dynamics of fish species and fisheries in particular data-poor situations.

Supplementary material

Details of the operating model, Supplementary Table S1 (parameter values for the base-case scenario), and Supplementary Tables S2 and S3 (extensions of Tables 3 and 4 containing the results of additional sensitivity analyses) are available at the *ICESJMS* online version of this paper.

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