Evaluating sustainability of fisheries bycatch mortality for marine megafauna: reference points and risk assessment for data-poor populations

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Abstract

Keywords

Fisheries bycatch threatens populations of many marine vertebrate taxa worldwide, including marine megafauna such as reptiles, birds, mammals and numerous elasmobranchs (Lewison et al. 2004, Barker and Schluessel 2005, Read 2008, Wallace et al. 2010, Anderson et al. 2011). Mitigation efforts have successfully reduced some threats (e.g., Werner et al. 2006, Cox et al. 2007, Moore et al. 2009, Gilman 2011, Dunn et al. 2011). But mitigation can be expensive (Bisack and Sutinen 2006, Huang and Leung 2007, Gallaway et al. 2008) and difficult to implement or enforce (Rodríguez-Quiroz et al. 2010, Gilman 2011), hindering broader use of best practices for minimizing the collateral damage of fishing on marine ecosystems.

Another important barrier to more widespread practice of bycatch mitigation is the scarcity of information on population-level effects of bycatch within and across fisheries. Lack of knowledge regarding the degree to which a fishery might be reducing a population's abundance, growth rate, or persistence viability – and in turn, altering its ecological role – leads to little incentive or guidance for strategically addressing bycatch. Despite the precautionary management paradigm agreed to in the FAO Code of Conduct for Responsible Fisheries (FAO

1995, 1996), fishing industry and decision-makers may be reluctant to mandate costly mitigation measures without evidence that fisheries bycatch is preventing population recovery, contributing to population declines, or substantially altering marine ecosystems (Gilman 2011). Demonstrating the impacts of fisheries on non-target species is thus an important component of building the necessary public and industry support and political will for changing fishing practices or modifying fishing effort.

Once bycatch is identified as a management priority that must be addressed, fisheries managers need guidance as to the level of bycatch reduction required to bring impacts within acceptable levels as defined by management objectives. Simply reducing incidental mortality to apparently low levels may not be sufficient, as in the case of rare and highly vulnerable species or when reduced bycatch is confounded with declines in population abundance (e.g,.Jaramillo-Legoreto et al. 2007, Tuck 2011). Given the limited amount of information typically available for non-target populations, determining the level at which bycatch will not have a negative effect on a non-target population is challenging. However, several analytical approaches developed for data-poor populations can address this issue by estimating *reference points* that can be used to assess the sustainability of fisheries bycatch. Reference points, based on measurable population indicators or performance measures (e.g., mortality, catch rates, etc), provide a means of evaluating whether management actions initiated by industry or agencies are sufficient to satisfy management goals. Their use is fundamental component of sustainable fisheries management (Quinn and Deriso 1999, Garcia et al. 2000, Hall and Mainprize 2004, Garcia and Cochrane 2005, Rueter et al. 2010), and the FAO (2010) has called on States and international fishery management organizations to establish mortality limits for bycatch species using a precautionary approach.

Methods to estimate reference points for data-poor non-target populations have been developed separately for different taxonomic groups, but their application to management of bycatch of marine megafauna is uncommon relative to the global number of managed fisheries and the diversity of affected populations. To facilitate wider use of these tools and maximize their effectiveness in fisheries management, a synthesis of available methods is needed to guide their application. We review a complementary set of approaches that can be used to estimate sustainable impact levels for a diverse group of taxa. Our review describes several reference point estimators, including their data requirements, assumptions, advantages and disadvantages, and applicability to different taxa. We discuss the critical role simulation-based Management Strategy Evaluation (MSE; Smith 1994) plays to assess the effectiveness of reference point estimators and optimize their performance given the ubiquity of limited data and high uncertainty, and we highlight the value of applying these methods within broader risk assessment frameworks to facilitate assessment of large numbers of species. Finally, we discuss challenges associated with using reference points as tools to guide bycatch management and offer some suggestions to address these.

Reference Point Estimators as Management Models

The majority of non-target populations affected by fisheries can be characterized as having insufficient demographic data to guide management via fully analytical stock assessment or population impact assessment models. In data-poor situations, management can still be guided by quantitative *management models*. Taylor et al. (2000) defined management models as tools to make decisions that result in management objectives being met. Thus they have a fundamentally different goal than do more complex ecological models, which aim to increase our understanding of system dynamics or accurately forecast system change. Management models may not predict system dynamics because they necessarily oversimplify ecosystem processes, but they use existing data to guide management to achieve desired outcomes.

Numerous reviews have described the importance of reference point management models for managing single or multiple target or non-target species and ecosystems (e.g., Taylor et al. 2000, Hall and Mainprize 2004, Jennings 2005, Punt 2006, Rueter et al. 2010). These have highlighted several key properties of useful management models: they are transparent and accessible to stakeholders who do not have quantitative scientific expertise; they estimate reference points that are linked explicitly to management objectives; their use as part of a performance-tested management strategy or procedure ensures a high probability of meeting objectives in spite of uncertainty (e.g., due to model uncertainty, bias and sampling error in parameter estimates, environmental stochasticity); and they use existing or available data for the specific study system. We discuss three general types of management models developed to estimate reference points for non-target populations of different taxonomic groups acrossa range of levels of data availability (Table 1).

Potential Biological Removal and PBR-like models

The Potential Biological Removal (PBR) model was developed to assess marine mammal stocks under the U.S. Marine Mammal Protection Act (MMPA) (Barlow et al. 1995, Wade 1998). PBR estimates the maximum human-caused mortality (in terms of total individuals) that could occur while still allowing management goals for that population to be met, accounting for uncertainty in population dynamics. Specifically, PBR identifies a mortality limit that ensures that stocks are likely to return to or remain at or above their “maximum net productivity level”

(MNPL), a term analogous to the maximum sustainable yield level used in fisheries stock assessments. It is calculated using a simple algebraic equation: PBR = $0.5R_{max}N_{min}F_R$, where $0.5R_{max}$ is one-half of the maximum growth rate of the population expected at low density, N_{min} is a minimum estimate of total population size (20^{th} percentile from a lognormally distributed population estimate), and F_R is a "recovery factor" that can vary between 0.1 and 1.0 depending on stock-specific management considerations.

The PBR has proven to be an effective management model (Wade 1998, Taylor et al. 2000). PBR provides a clear and definable reference point against which to evaluate a measurable indicator (total bycatch mortality) estimated from observer programs. It can also be modified to reflect management goals other than MNPL. The model has minimum data requirements, needing only an estimate of population abundance. The other parameter that must be 'estimated' is R_{max}, but as this is typically not measurable, PBR models often utilize default values (Wade 1998). Finally, PBR is robust to uncertainty in a flexible way; poor data on population abundance results in a lower PBR because of wider confidence intervals for N_{min}, while F_R takes on more conservative values for stocks with more precarious conservation status.

PBR's simplicity and minimal data requirements have resulted in its application to a broad range of data-deficient taxa and systems apart from the U.S. MMPA context. For example, it has been applied to numerous marine mammal populations outside the U.S. (e.g., Maunder et al. 2000, Marsh et al. 2004, Underwood et al. 2008), as well as to estimate sustainable limits for hunting wild meat (Milner-Gulland and Akçakaya 2001, Parry et al. 2009) and to evaluate mortality of migratory birds, including bycatch of seabirds (Niel and Lebreton 2005, Dillingham and Fletcher 2008, 2011, Runge et al. 2009).

The PBR approach has been vetted and validated using simulations to test its performance under realistic levels of data imprecision, biases, and assumption violations for marine mammals in the U.S. (Wade 1998, Taylor et al. 2000). However, application of PBR to other taxonomic groups can require modifications of the methodology and simulation-based testing using plausible ranges and types of error appropriate to the systems in question. PBR parameters might not be measurable for all types of species, and PBR assumes the affected population exhibits non-structured logistic growth and incurs non-structured anthropogenic mortality (e.g., mortality rates assumed to be constant with age). Dillingham and Fletcher (2011) modified PBR for seabirds to use adult survival, age at first breeding, and number of breeding pairs as inputs parameters, since these are generally better known than total abundance or R_{max}. Curtis and Moore (submitted) developed a generalization of PBR for sea turtles termed MGD (Maximum Growth Decrement), , which incorporates a matrix model of population dynamics to deal with strongly age-structured demographic and bycatch rates for sea turtles. The MGD model uses reproductive value rather than individuals as the unit of population currency.

Sustainability Assessment for Fishing Effects (SAFE)

Another reference point management model is the SAFE model, initially developed to rapidly assess hundreds of fish bycatch species in the Australian Northern Prawn Trawl fishery (Zhou and Griffiths 2008; Zhou et al. 2009). Whereas the indicator used for PBR-like models is total mortality (e.g., number of individuals), the SAFE management model focuses on fishing mortality *rate* as its indicator (Zhou and Griffiths 2008, Zhou et al. 2009, 2011). The SAFE model derives the fishing mortality rate through estimation of spatial overlap between species distribution and fishing effort, adjusted by gear-specific formulations of catchability and postcapture mortality. Reference points proposed to date with SAFE (Zhou et al. 2011) are taken from the traditional fisheries management paradigm, for example: F_{MSY} (maximum sustainable fishing mortality rate, corresponds to a population maintained at its maximum sustainable yield biomass, B_{MSY}); F_{limit} (fishing mortality rate corresponding to a populations limit biomass, B_{lim}, which is assumed to equal $0.5*B_{msy}$); and F_{crash} (minimum unsustainable fishing mortality rate that will theoretically lead to eventual population extinction). These are derived from life-history parameters – namely maximum productivity (r) or natural mortality rates (M) – which have been published for many fish species or are assumed to be estimable from other life history characteristics such as length and growth parameters, max body length, temperature, longevity, and age at maturity (e.g., Pauly 1980, Hoenig 1983, Punt et al. 2005). For example, based on the simplest of six methods used by Zhou et al. (2011), $F_{MSY} = 0.5r$, $F_{lim} = 0.75r$, and $F_{crash} = r$.

The key input data to SAFE include fishery effort distribution and species distribution, which are likely available in many fisheries. Species spatial distribution may be obtained from surveys, fishery records, or observer programs. When none of these data types are available, suitable habitat range may be used as a surrogate. Fishing mortality estimates are most accurate if relative density of populations in fished and unfished areas can be estimated (e.g., based on surveys or catch-per-unit- effort data, Zhou and Griffiths 2007). Catchability is calculated from estimates of encounterability and selectivity and may be derived based on field study (Zhou et al. 2009), literature review, meta-analysis of biological characteristics (Zhou et al. 2007), or by simply assuming unity (Pope et al. 2000).

Since its original development, SAFE has been improved and extended for application to major Australian fisheries using various gears, including fish trawl, seine, gillnet, and longline (Zhou et al. 2007, 2009, 2011). Currently SAFE is most suitable for fish species (teleosts and chondrichthyes) as the linkage between traditional fishery reference points and life-history traits are well studied for fish (Quinn and Deriso, 1999), and for non-aggregating species, given the method of using species distributions to estimate fisheries mortality rates (Zhou and Griffiths 2008). It has also been applied to sea snakes (Zhou et al. 2012). Theoretically, PBR and SAFE are related – for example, PBR divided by N_{min} is analogous to the F_{MSY} used by SAFE – but the two methods require different inputs and trade-offs in assumptions. SAFE as a management model is being incorporated into Ecological Risk Management by the Australia Fishery Management Authority (http://www.afma.gov.au/managing-our-fisheries/environment-and-sustainability/ecological-risk-management/).

As with PBR, advantages of SAFE include its low-cost and minimal data requirements, transparency, and flexibility that allows alternative methods for estimating fishing impact depending on available data. Uncertainty can be quantified in both indicator and reference points. However, the estimated fishing mortality as calculated under SAFE is prone to particularly high uncertainty when data are limited and it is not yet clear how robust the method is to this and its simple assumptions.

Depletion-Corrected Average Catch and Depletion-based Stock Reduction Analysis

For populations for which only a time series of catch (or bycatch) is available, the Depletion-Corrected Average Catch (DCAC) model (MacCall 2009, based on work by Restrepo et al. 1998) may be used to assess the sustainability of that catch. DCAC calculates a reference point called sustainable yield: $Y_{sust} = \Sigma C / (n + W/Y_{pot})$, where C is the sum of removal (abundance or biomass) since the beginning of exploitation (or least over a relatively long time span), n is the number of years over which that removal occurred, W is an unsustainable ‘windfall’ catch representing a one-time reduction in population biomass or abundance to a sustainable yield level, and Y_{pot} is the potential annual yield. The windfall ratio (W/Y_{pot}) is the magnitude of windfall removal relative to a single year of sustainable removal (e.g., removing 100 units is like removing 10 units for 10 years) and is approximated using other parameters since W and Y_{pot} are not usually known. Importantly, Y_{sust} is an estimate of a sustainable yield that would prevent decline from current abundance levels. It is not an estimate of the Maximum Sustainable Yield (MSY).

The minimal data requirement for DCAC is an estimate of total catch from the population over a specified time period. Additional parameters have to be specified, including the proportional reduction of the population from virgin biomass or carrying capacity ($\Delta = B_t/K$), the fraction of unexploited biomass at which MSY occurs (i.e., B_{MSY}/K), a size-independent natural mortality rate (M), and an estimate of the ratio (c) of fishing mortality (F) to M that will maintain MSY (i.e., $F_{MSY} = cM$); this ratio is commonly between 0.6-1.0 (MacCall 2009). Together, these parameters enable estimation of the windfall ratio; they are not likely to be known for data-poor stocks but, as for PBR and SAFE models, can be assigned default or precautionary values or be estimated based on meta-analysis or expert judgment. DCAC is best suited for bycatch species whose catch has been monitored for an extended period of time, and is only appropriate for species with low mortality rates (≤ 0.20 yr^{-1}), which are usually slow growing and late maturing. The model has been used by the Pacific Fishery Management Council (PFMC) in the U.S. to set overfishing limits for six species of rockfish (Sebastes spp.) and the tope shark Galeorhinus galeus (Berkson et al. 2011). Developed for target fish stocks, the method could potentially be applied to non-target populations of elasmobranchs for which bycatch data have been collected.

A recent extension of the DCAC framework is the Depletion-Based Stock Reduction Analysis (DB-SRA) (Dick & MacCall 2011). Compared to DCAC, it requires specification of an extra parameter – age at maturity – and a simple population growth function. This extra information enables estimation of various fishery reference points such as MSY and unfished biomass K. DB-SRA been used by the PFMC for 42 groundfish stocks (Berkson et al. 2011). Like all reference point estimators summarized here, these models are attractive because of their simplicity, minimal data requirements, and clear relationship to defined management objectives. Moreover, uncertainty is handled explicitly in DCAC and DB-SRA by inputting distributions rather than point estimates for the reference point parameters to obtain probability distributions for the estimates using Monte Carlo methods. Simulation-based evaluation of DCAC and DB-SRA by Wetzel and Punt (2011) suggest the methods may be fairly robust to bias in several input parameters. However, the methods are also sensitive to the specified B_t/K, and their basic data requirement (accurate catch history) may be difficult to satisfy for most data-poor non-target species.

The need for assessing management model performance: Management Strategy Evaluation

Simple reference point estimators may serve as a good starting point for guiding management of data-poor populations because of their simplicity and minimal data requirements. However, the reference points calculated from these methods rely heavily on assumptions and educated guesses for key parameters (e.g., based on meta-analyses or expert judgment) as substitutes for empirical information about the study system. Before they can be considered to be good management models, they must be evaluated through simulation analyses to assess their ability to cope with realistic levels of uncertainty and be useful for achieving management aims in spite of it (Butterworth and Punt 1999, Punt 2006). Management Strategy Evaluation is a formal mechanism for conducting this assessment.

A Management Strategy Evaluation (MSE; Smith 1994) involves simulating a whole managed system – involving feedbacks between "true" population dynamics, data collection with sampling and model error, and indicator-based management decisions – to evaluate the "performance" of a management model in terms of its ability to achieve desired outcomes (see reviews by Sainsbury et al. 2000, Punt et al. 2001, Punt 2006, de Oliveira et al. 2009, Bunnefeld et al. 2011; see Rochet and Rice 2009 for a critique of the approach). The MSE concept in fisheries originated in several forms; for example, it is often referred to as management procedure evaluationwith a substantial history of development by the International Whaling Commission (IWC) to manage commercial whaling (Cooke 1999, Butterworth and Punt 1999, Butterworth et al. 2010, Cooke et al. 2012). As relates to non-target or data-poor populations, MSE or MSE-like approaches have been used to develop PBR for marine mammals (Wade 1998), MGD for marine turtles (Curtis and Moore, submitted), and to assess performance of DCAC and DB-SRA tools for data-poor fish stocks (Wetzel and Punt 2011). The SAFE tool has not been evaluated using MSE, nor have applications of PBR to non-marine mammals. In general, the application of MSE to reference point models for bycatch management in specific fisheries is not common, and more work is needed to evaluate context-specific performance of different reference point estimators for their suitability as effective management models.

MSE is a general framework of analysis, rather than a specific model; most MSEs are composed of multiple interacting models that can even include economic and social processes (Dichmont et al. 2008). MSEs can act as a form of sensitivity analysis, revealing how data quality, assessment model structure, reference points, and the management process itself affect the *performance* of a given management tool. Importantly, MSE can identify the conditions under which a management model is likely to fail, e.g., lead to severe over-or under-estimation of stock status or allowable take.

A typical MSE for bycatch management based on the use of reference points would have three primary components (Figure 1). (1) An **operational model** serves as an underlying “true” population model, with biological characteristics drawn from distributions obtained from the literature. This model contains the biological information and population processes that are thought to affect the population of interest. Uncertainty in vital rates and density-dependent responses can be included in the operational model, as can environmental stochasticity or cycles. (2) An **estimation or observation model** simulates the process by which data are collected and population status is evaluated by managers given sampling error in the data, model assumptions, and uncertainties about the operational model. This key objective is generating realistic levels of error to enable determination of the level of precaution needed to set removal rules that use available information, and to learn the potential value of improved monitoring (Rosenberg and Restrepo 1994). (3) A **management or removal model** describes the management procedure, i.e., by which data (simulated in the observation model) are used by managers to estimate reference points (if they are dynamic) and indicators, and that rules are implemented in response to a change in indicator status (such as harvest control rules) to limit the level of population removal each year. Outcomes of the management procedure feed back into the next time step of the operational model. Through many stochastic iterations of the MSE simulation, one can estimate the probability that pre-defined management objectives are achieved given the specified uncertainties.

The propensity of a management model to be overly conservative can be tested as well as its likelihood of failing to adequately protect a population or determine its status (Snover and Heppell 2008, McElhany 2010). Ideally, MSEs should be used to compare different management models and their trade-offs in risk probabilities for competing management goals (Sainsbury et al. 2000). For example, Maunder et al. (2000) used a Bayesian approach to assess the trade-off between risk of failing to adequately protect Hookers' sea lion and loss of squid catch associated with implementing a PBR-like management model in New Zealand. The extent to which trade-offs are considered in an MSE depend in part on the legal context of management; for example, the MSE conducted by Wade (1998) only considered risk of failing to achieve conservation objectives because of statutory mandates of the U.S. MMPA.

Value of reference point models to multi-species risk assessments

While many of the examples cited above focused on a single species of taxa, reference point models can also be used to developed multi-species assessment of sustainable take. Using management models based on simple reference point estimators can facilitate and increase the value of multi-species risk assessments by permitting quantitative assessment of larger numbers of populations than would be possible if relying on more data-hungry ecological models. Taylor et al. (2000) clearly documented the value of the PBR management model as a way of satisfying statutory mandates to assess status for large numbers of U.S. marine mammal stocks for which more data-hungry assessment methods were impossible. We highlight a couple more recent examples, one using the SAFE tool applied in Australia within a hierarchical risk assessment framework called ERAEF (Ecological Risk Assessment for the Effects of Fishing), and the other using PBR for seabirds to inform fishery policy in New Zealand.

ERAEF is a precautionary triage framework founded on principles of ecosystem-based fisheries management (EBFM) (Hobday et al. 2007, 2011, Smith et al. 2007a). Its hierarchical structure is designed to efficiently deal with assessing impacts for large numbers of species under data-poor circumstances. Designed initially in the context of the Australian Fisheries Management Authority (AFMA), the framework has been adopted by the Marine Stewardship Council (MSC) as part of their fisheries certification process (MSC 2009) and has been used in other international contexts, such for assessing threats to seabirds by longline fisheries in the Atlantic (Tuck et al. 2011). Briefly, the framework consists of four levels of analysis ranging from mostly qualitative (Scoping and Level 1), where management objectives are defined and potential impacts described and scored using expert judgment, to quantitative (Levels 2 and 3). Level 2 is a productivity-susceptibility analysis (PSA), used to rank relative risk for individual species evaluated within a fishery (e.g., Stobutzki et al. 2001, Cortes et al. 2009, Arrizabalaga et al. 2011, Tuck et al. 2011). Species ranked as high risk in Level 2 are assessed at Level 3, which consists of a fully quantitative assessment to provide a more absolute measure of risk. Using the SAFE tool as a Level 3 analysis method, Zhou et al. (2011) evaluated hundreds of fish species and identified numerous non-target chondrichthyan species as being killed at unsustainable levels in several Australian fisheries.

New Zealand's draft seabird policy is based on a two-level risk assessment framework, the second of which is quantitative and uses PBR (the same equation as applied to marine mammals) to identify populations with high probability of risk to unsustainable mortality ((New Zealand Ministry of Fisheries 2011, Richard et al. 2011). Recent application of this framework permitted dozens of bird species to be ranked by their relative risk, and several of the most at-risk species were identified as having a high probability of being impacted by unsustainable mortality levels. In both of the above examples, most species evaluated could not have been assessed but for the use of simple reference point models.

Although the above examples highlight the utility of data-poor reference points in the context of conducting multi-species assessments, only the example for marine mammals described by Taylor et al. (2000) used a management model that had undergone performance testing via MSE. SAFE and applications of PBR to seabirds have yet to undergo the simulation-based MSE performance testing necessary to ensure their effectiveness as precautionary management tools. Still, by comparing status (indicator vs. reference point) for many species within a multi-species assessment framework, the use of reference points at least allows for efficient ranking of relative risk and helps focus research and mitigation priorities. Risk ranking is improved if reference points and/or indicators can be described by posterior probability distributions (rather than point estimates), thus permitting probabilistic statements about the likelihood of an indictor exceeding a reference point (e.g., Richard et al. 2011, Moore submitted).

Challenges and suggestions to facilitate use of reference points for risk assessment

Implementation of reference points to inform and guide bycatch (and catch, in the case of elasmobranchs) management of marine vertebrate megafauna remains rare. Even fewer are applications of reference point models that have been performance-tested by MSE. Several common technical, operational, and political challenges continue to hamper the widespread application of these tools, but there is precedent for addressing many of these.

Technical challenges

From a technical standpoint, the challenge of meeting even minimal data requirements, is the primary hurdle to developing and calculating robust reference points and measurable indicators for a given population. Several reference point estimators we reviewed rely at least to some extent on default life history parameter values and model assumptions that can be equivocal, because basic population metrics such as static abundance estimates or catch series data might not exist. Insufficient information on data-poor species can be compensated for in part through meta-analyses of existing demographic information for related species (e.g., Pauly 1980, Hoenig 1983, Frisk et al. 2001, Niel and Lebreton 2005, Le Quesne & Jennings 2012). Even with these strategies, , the absence of basic data needs may limit many species assessments to a more qualitative level of a multi-tiered assessment framework (e.g., Richard et al. 2011).

Population information, such as species-specific data for bycatch mortality or mortality rates,are also often lacking. At-sea observer programs sample only a fraction of existing fisheries and, where available, often have insufficient coverage to estimate bycatch especially for infrequently encountered species (Lewison et al in review). Moreover, estimates based on observer or self-reported data may be biased by numerous factors, such as changes in fishing practices on observed trips, hiding or underreporting of discards, unobservable bycatch (e.g., animals lost before gear retrieval), and unknown post-release mortality rates (e.g., Gales et al 1998, Haigh et al. 2002, Kelleher 2005, Brothers et al. 2010). Some of these issues may be identified or addressed through careful analysis of reports and landings, increased observer coverage, or video monitoring (e.g. Mawani 2009, Grinnell 2010). Information on species composition and levels of discards and landings may also be improved by increasing at-sea observer coverage, training observers to distinguish among species, and taking genetic samples of landings (Shivji 2010). Management strategy evaluation becomes a particularly important tool when data are limited and likely biased, by providing the means to evaluate whether application of reference point estimators for management is robust to the suite of known data problems. However, MSE itself can be difficult if there is insufficient knowledge to specify plausible operational or observation models.

Apart from data limitations, fisheries sustainability assessments may be confounded for populations also affected by non-fisheries anthropogenic impacts, leading to elevated risk of an overall negative population outcome when fishing mortality is assessed as sustainable. To the extent that non-fisheries impacts can be quantified, they can be accounted for in sustainability assessments of fisheries. However, while direct sources of human-caused mortality are theoretically straightforward to address, the same is not true indirect impacts such as sublethal fisheries interactions, anthropogenic reductions in carrying capacity (Moore, submitted), and the importance of social behavior (Jacoby et al. 2011), spatial complexity (Cardinale et al. 2011), and social relationships (Mills and Ryan 2005) to marine vertebrate demography.

Operational challenges

At the operational level, a primary issue is the lack of clearly defined management objectives for non-target species in the majority of management contexts, both domestic and international. Vague language in the United Nations Fish Stocks Agreement of 1995 refers to “maintaining or restoring populations of such species above levels at which their reproduction may become seriously threatened," but provides no further guidance on defining the bounds of acceptable population status and risk as weighed against incentives or imperatives to maximize value (e.g., economic, employment, or protein) from fisheries. Domestic legislation and regulations rarely define more specific objectives. Collaboration among decision-makers, scientists, managers, and other stakeholders is therefore required to define population-relevant management objectives and accepted risk levels for non-target species.

The process of pre-defining management objectives, establishing rules for actions linked to status of performance measures, and performance testing the management procedure via MSE can be difficult and may be resisted by different stakeholder groups for a variety of reasons. Besides obvious disagreement about acceptable trade-offs in conservation vs. utilization objectives, reluctance to the process can stem from lost flexibility for managers, difficulty for non-scientists of understanding the MSE process, disagreement about how to handle uncertainty, concerns about data used to estimate indicators and reference points, and an overwhelming number of species to consider (Smith et al. 1999, Hilborn 2007, Kurota et al. 2010, Hobday et al. 2011). However, well-developed management models and assessment frameworks can streamline the assessment process, build consensus among stakeholders in the management process and generating incentives to reduce uncertainty (e.g., Smith et al. 2009, Hobday et al. 2011), while accessible, global databases for demographic and catch information on non-target taxa can expedite data gathering and management.

Compliance is another major challenge to successful implementation of management procedures based on use of reference points (refs). At the international level, lack of compliance with agreements regarding data provision is beginning to be addressed through measures such as an agreement by International Commission for the Conservation of Atlantic Tuna (ICCAT) to make continued landing of sharks by member States contingent on provision of shark landings and discard data to the Commission (Rec. 11-15). Increased at-sea observer coverage or video monitoring can improve compliance as well as data quality for indicator assessment, and individual limits and accountability by vessel or license holder for impact on non-target species has proven successful in both domestic and international contexts (e.g., Waugh et al. 1998, Mawani 2009). However, both technical and operational challenges related to data collection, monitoring, and enforcement are complicated by varying capacity among fishing nations to augment cost-intensive programs. Cost-effectiveness and simplicity of management programs should be maximized, and negative impacts on fisheries (in terms of profit, landings, or other prioritized values) minimized.

Implementing management procedures requires regular feedback and updating based on performance and new information (refs). These and other needs, such as monitoring and stakeholder involvement, can be met through a standard framework for ecological risk assessment (ERA; USEPA 1998) with an inclusive planning process and regular monitoring and feedback of results to enable adaptive management. For example, the Commission for the Conservation of Antarctic Marine Living Resources, which coordinates science and management for fisheries in the Southern Ocean, applies an ERA framework for seabird bycatch management that incorporates elements of vessel-specific accountability, spatial management based on risk, and rigorous monitoring up to 100%, and continues to improve through annual monitoring, review, and revision (Waugh et al. 2008). Management regimes should also be set up to encourage further data collection or acquisition to improve knowledge of the system (e.g., Taylor et al. 2000).

Political challenges

Progress on incorporating assessment and management of marine megafauna into fisheries governance regimes has been slow. Non-target species are generally a lower priority than target species, and in the case of elasmobranchs, their landed value provides a further barrier to potentially reducing catches. Conflicts between fisheries management agencies and wildlife protection agencies in many countries generally result in lack of coordination and agreement, with the outcome taxon-dependent (refs). At the international level, differing values among countries with respect to conservation complicate international agreement on appropriate measures. Cumulative mortality across domestic and international fisheries and from non-fishing human impacts can derail discussions of limiting bycatch of highly migratory species, and uncertainty is also often used as a reason for inaction (refs).

The Food and Agriculture Organization's Code of Conduct for Responsible Fisheries (FAO 1995) calls for a precautionary approach to management of living marine resources, which requires that uncertainty may not be used to delay management action, which should instead be based on the best available information. Several technical guidelines and plans of action agreed under the FAO Code of Conduct, other international and domestic instruments (REFS), and trade regulation and incentives all provide means for forward progress on management of fisheries impacts on non-target marine megafauna. Champion agencies or governments, or partnerships among them, can have considerable influence on domestic and international policy, and would thus provide an important means forward in instituting reference points as a basis for bycatch management for marine megafauna (REF). Where indicators for individual fisheries are being evaluated against reference points to manage a species that is impacted by other fisheries or non-fishery human impacts or in other jurisdictions, management may initially focus on proportional impact of the fishery relative to other fisheries affecting that population until another basis for relative responsibility for population declines can be agreed upon by all actors affecting a population. However, potential unintended consequences of unilateral management action may include effort displacement to another area or fishery with potentially negative ecosystem consequences (REF).

Conclusions

The need to evaluate the potential impact of fisheries and other mortality sources on non-target marine populations is great, as fisheries bycatch alone has been demonstrated to have serious population-level effects across a range of taxa. Despite their limitations, reference points are widely viewed as essential components of sustainable fisheries management frameworks for target fish stocks. Reference point estimators designed for non-target data-poor populations similarly offer similar potential to help manage impacts of fisheries (Hall and Mainprize 2004). The use of PBR for marine mammals in the U.S. was an early regulatory example of this. Since then, there have been noteworthy changes in domestic fisheries policies to implement management procedures for assessing incidental impacts of fishing based on use of reference point estimators. A few examples include requirements under the U.S. Magnuson Stevens Act to use reference points for target and non-target species, New Zealand's' draft seabird policy that employs a PBR-based reference point, and implementation by the Australian Fisheries Management Authority of ecological risk assessments using the SAFE model.

Management Strategy Evaluation is an essential component for ensuring effective performance of a reference point estimator for use as part of a management procedure. MSE-type approaches have become relatively common practice in the management of target fish species. However, their use to date for evaluating management procedures for non-target populations based on data-poor reference point estimators is rare. The PBR management model and the IWC Revised Management Procedure (Cooke et al. 2012) remain the best examples of policy implementation. The recently developed MGD tool by Curtis and Moore (submitted) provides the only other example we know of developed for another non-target taxonomic group. However, without MSE validation, there can be little assurance that the management approach will be sufficiently precautionary or be robust to uncertainties in its ability to help achieve management objectives.

There are numerous challenges to implementing management based on the use of simulation-tested reference point models. The most daunting include information limitations that can prevent use or evaluation of management models with even the simplest assumptions and minimal data needs, coupled with operational challenges of defining management objectives and control-feedback procedures that participants in the management process can agree to. In the case of international fisheries, problems are exacerbated by inconsistent levels of funding, compliance, commitment to conservation objectives and international agreements, and complexities of management in the context of cumulative fishery and non-fishery impacts rather than only fishery-specific impacts.

Despite these challenges, management strategies based on MSE-tested reference points for data-poor species provide a powerful tool for identifying sustainable levels of mortality or take for many non-target populations consistent with a precautionary approach. Recently developed risk assessment frameworks also can help overcome challenges by establishing a template for building consensus around objectives, efficiently dealing with uncertainty, and by providing a context for applying reference point models to large numbers of species. New examples are emerging that illustrate the utility of these complementary sets of tools to practically improve fisheries management (Smith et al. 2007b, Smith et al. 2009, Hobday et al. 2011, Zhou et al. 2011). Increased uptake of these approaches by domestic and international fishery management institutions is needed to ensure sustainability of fishing impacts on vulnerable populations of marine megafauna.

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Table 1. Summary of reference point estimators

Tool	PBR (Potential biological removal)	Modified PBR for seabirds
Taxonomic application to date	marine mammals	seabirds
Management objective(s)	• Minimise time to recovery • Sustain population near carrying-capacity • Sustain population at some fraction of carrying-capacity • Risk assessment from one or more fisheries	• Minimise time to recovery • Sustain population near carrying-capacity • Sustain population at some fraction of carrying-capacity • Risk assessment from one or more fisheries
Reference point	Potential biological removal (PBR)	Potential biological removal (PBR)
Performance indicator	Mortality estimate (number of individuals)	Mortality estimate (number of individuals)
Data requirements	• Population size (20th percentile estimate)	• Number of breeding pairs (or total population size)
Other required parameters (can use defaults)	• Maximum growth rate • Management recovery factot	• Optimal adult survival rate • Age at first breeding • Management recovery factor
Advantages	• Minimal analysis effort • Simple and transparent • Accomodates multiple management objectives	• Minimal analysis effort • Simple and transparent • Accomodates multiple management objectives
Disadvantages	• Can be overly conservative • Indirect anthropogenic effects not accounted for	• Can be overly conservative • Standard PBR does not account for age- or sex-bias in bycatch • May have to be adapted to account for terrestrial threats • Estimates total anthropogenic mortality that can be sustained, which may be difficult/impossible to assess fully

Table 1 (cont.). Summary of reference point estimators

Tool	MGD (Maximum growth decrement)	SAFE (sustainability assessment for fisheries effects)
Taxonomic application to date	sea turtles	elasmobranchs, finfish, sea snakes
Management objective(s)	• Minimise time to recovery • Sustain population near carrying-capacity • Sustain population at some fraction of carrying-capacity • Risk assessment from one or more fisheries	• Sustain population at maximum sustained fishing mortality (MSM), equivalent to maximum sustained yield • Minimize extinction risk
Reference point	Maximum growth decrement (MGD)	• Fishery mortality rate relevant to management objective (u_{MSM} or u_{crash})
Performance indicator	Size-classified mortality estimates (number of individuals)	Fishing-induced mortality (derived from effort and other data)
Data requirements	• Number of breeding females	• Population survey data • Fishing effort data (with location & gear type)
Other required parameters (can use defaults)	• Life history (slow or fast growing species) • Approximate population model • Management recovery factor	• Catch rate • Escapement rate • Natural mortality
Advantages	• Accounts for size-structure of mortality • Adjusts PBR model for unequal reproductive value of mortality	• Availability of required data for some species • Can use values for related species when data are missing
Disadvantages	• Can be overly conservative • May have to be adapted to account for terrestrial threats • Estimates total anthropogenic mortality that can be sustained, which may be difficult/impossible to assess fully • More complex than standard PBR	• May not be appropriate for aggregating species

Table 1 (cont.). Summary of reference point estimators

Tool	DCAC (Depletion-corrected average catch)	(DB-SRA) Depletion-based stock reduction analysis
Taxonomic application to date	fishes	fishes
Management objective(s)	Sustain population at current levels	Sustain population at MSY or other reference point
Reference point	Ysust = sustainable yield	• Fishery mortality rate relevant to management objective (*u*MSM or *u*crash)
Performance indicator	Average catch	Average catch
Data requirements	Long time series of catch	Long time series of catch
Other required parameters (can use defaults)	• ratio of current biomass level to K • fraction of K where MSY occurs • natural mortality rate M • the ratio (c) of fishing mortality (F) to M that will maintain MSY (i.e., FMSY = cM)	same as DCAC, plus age at maturity and specification of a productivity model
Advantages	• Robust to some parameter biases • Adjusts PBR model for unequal reproductive value of mortality	• Robust to some parameter biases • Can address various management objectives
Disadvantages	• Sensitive to specification of B(t)/K • Can address limited management objectives • Data requirement may be difficult to meet	• Sensitive to specification of B(t)/K • Input requirements may be difficult to meet

Figure 1. Management Strategy Evaluation components for a typical framework used to advise bycatch management.