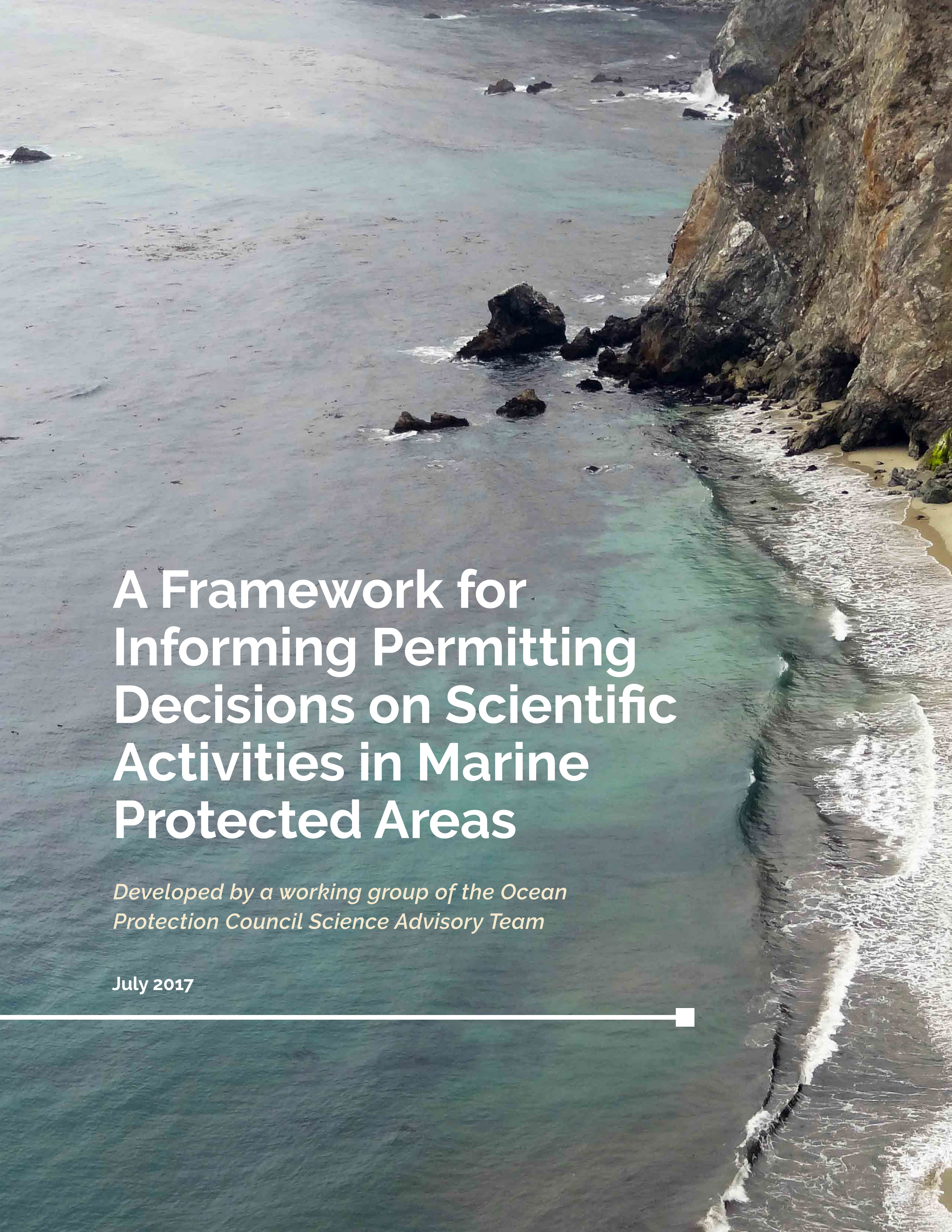

A Framework for Informing Permitting Decisions on Scientific Activities in Marine Protected Areas

Developed by a working group of the Ocean Protection Council Science Advisory Team

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About the Ocean Protection Council Science Advisory Team

This report has been endorsed by the Ocean Protection Council Science Advisory Team (OPC-SAT). The purpose of the OPC-SAT is to provide scientific advice to California's ocean and coastal state agencies and entities. The work of the Ocean Protection Council Science Advisory Team is managed by California Ocean Science Trust and supported by the California Ocean Protection Council.

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Executive Summary

Marine protected areas (MPAs) and MPA networks are important management tools that often have multiple goals and must balance potentially conflicting activities, one of which is scientific research. MPAs provide unique and important research opportunities because their ecosystems are subject to minimal human disturbance. Moreover, research is essential for evaluating MPA performance, and thus is an integral component of MPA management. However, scientific research may also impact the biota and habitats being studied. Hence, MPA managers must understand and weigh the ecological costs and benefits of proposed research activities to determine whether they can be permitted within MPA boundaries without compromising the effectiveness of the MPA or the integrity of an MPA network.

At the request of the Department of Fish and Wildlife (Department), we propose a quantitative, ecologically-based decision framework to estimate the impacts of scientific research with the goal of facilitating scientific permitting decisions in California's newly established network of MPAs. The framework identifies the ecological consequences of a diversity of scientific research activities and provides an unbiased, transparent, and objective means to make informed permitting decisions. This approach consists of four steps:

1. **Exclude projects that don't need to be conducted in MPAs** – This "MPA relevance" component considers whether or not an MPA is essential for meeting the objectives of the research project (e.g., does the project require a protected population or community or are non-MPA locations inappropriate for the study). The Department has been employing a similar criterion for reviewing permits since 2008.
2. **Quantify ecological impacts of the project** – This model-based element uses scientific principles to assess the proportionate impacts within an MPA to: a) the population of any targeted species, b) four major marine ecological assemblages (macrophytes, sessile invertebrates, mobile invertebrates, and fishes), and c) the physical habitat that supports MPA biota. The model quantitatively estimates the ecological impacts of scientific activities, including consideration of the vulnerability of targeted species, assemblages, and habitats, based on their recovery time and the ecological significance of affected biota.
3. **Quantify the cumulative impacts to species, assemblages, and habitat affected by the proposed project and all other on-going projects in the MPA** – This analysis allows each research project to be evaluated independently while also determining its contribution to the cumulative impacts of all research activities in the MPA.
4. **Compare the estimated cumulative impacts of all projects with policy-based acceptable impact thresholds for species, assemblages, and habitats** - This outcome will lead to decisions to accept, deny or request modification and resubmittal of proposed projects.

The core of the framework is a suite of quantitative models that estimate the ecological impacts for the many methods commonly used in scientific research projects. Ecological impact is expressed as a proportion of the population, assemblage, and habitat within an MPA that will be affected by the proposed research. The models take into account direct impacts (e.g., activities resulting in immediate mortality or habitat damage), as well as indirect impacts (e.g., activities that generate incidental or

unintentional effects on other species, assemblages, or habitat). Impacts are calculated separately for individual species, ecological assemblages, and habitats. These proportionate impacts are then adjusted to account for vulnerability of the species, assemblage or habitat, based on their estimated time for recovery and the ecological significance of the affected biota.

Determining an acceptable level of ecological impact is a policy decision that may vary among species, ecosystems and MPAs. As a starting point, we propose an overall (i.e., cumulative) impact limit of 0.1 to any population, assemblage, or habitat, as a level beyond which the conservation value of an MPA may be compromised. The ecological impacts calculated in the framework are then compared with the impact threshold to determine if any individual project, or the cumulative impact of multiple projects, exceeds the acceptable threshold. The ecological impacts are compared to the acceptable impact thresholds, both individually and cumulatively for each targeted species, each of the four assemblages (macrophytes, sessile invertebrates, mobile invertebrates, and fishes), and the habitat. If any of these exceed the threshold, the approach outlined in the framework indicates that the proposed research should be revised to reduce its impact or permission to proceed should be denied.

While we propose an overall impact threshold of 0.1, we also recommend that allowable impact be linked to the anticipated benefits of the research. The Department should allow projects with small direct management value to consume only a small fraction of the available impact threshold, leaving room for future research envisioned to be of greater scientific value, or critical to informing MPA management. Moreover, we propose that no individual project should consume more than 1/5th of the available threshold for any population, assemblage, or habitat without the likelihood of generating equivalent benefits as determined by permitting staff.

The proposed approach identifies the ecological impacts of proposed scientific procedures and estimates their effects on species, communities, and habitats within each MPA and compares the individual and cumulative impacts of scientific projects against Department-determined thresholds. This objective and transparent method for making decisions to permit scientific research in MPAs can be consistently applied across staff and over time and facilitate interactions between managers and researchers so that modifications to study designs can be made before or after permit submission. Applicants will benefit because this approach should expedite permitting decisions for most projects. It will also provide managers and researchers with information on the state of species, assemblages and habitats within an MPA targeted for study. An additional advantage of using this framework is that high-impact projects can be readily identified and staff resources can be focused on projects of greatest concern to achieving MPA conservation goals.

Introduction

Marine protected areas (MPAs) and MPA networks[1] are important management tools for protecting species, habitats, ecosystems, and biodiversity (Claudet et al. 2008, Lester et al. 2009, Fenberg et al. 2012, Edgar et al. 2014, Guidetti et al. 2014, Caselle et al. 2015). Consequently, the number and cumulative area set aside in MPAs has grown rapidly over the past few decades (Wood et al. 2008, Lester et al. 2009, Lubchenco and Grorud-Colvert 2015). Because MPAs often have multiple objectives, including conservation, research, fisheries management, and public enjoyment, managers must balance potentially conflicting activities to ensure that MPA goals are met. Besides their conservation or other goals, MPAs also provide unique and important scientific research and educational opportunities because their ecosystems are usually subject to minimal human disturbance. For example, scientific study designs can require biota and habitat within MPAs to serve as important references for understanding the effects of human activities on the structure and functioning of ecological communities (Dayton et al. 2000, Sainsbury and Sumaila 2003, Carr et al. 2011, Sala et al. 2012), or provide valuable information about populations and life history parameters in the absence of fishing (Garrison et al. 2011). In addition, scientific information on the status and dynamics of populations and communities is essential for MPA managers to monitor and evaluate individual MPA and MPA network performance (Grorud-Colvert et al. 2010, Babcock and MacCall 2011, Carr et al. 2011, McGilliard et al. 2011, Starr et al. 2015). Thus, issuing permits for scientific activities is an integral component of MPA management (Stab and Henle 2009).

Scientific activities have the potential to impact the abundances, demographic structure, or behavior of species and modify their habitat depending on the specific procedures used, and the spatial extent and frequency of their application. Thus, scientific activities could alter the structure and functional processes of biological communities and potentially compromise the effectiveness of an MPA or the integrity of an MPA network. To ensure that MPA goals are met, managers must understand the likely ecological impacts of proposed scientific work in order to determine whether these activities should be permitted within MPA boundaries, and if so, with what parameters, controls, conditions or constraints to advance the science without compromising the MPA goals. Much attention has been given to determining the ecological impacts of fishing and fishing gear and other forms of human activity on marine populations, communities, and habitat (Pauly et al. 1998, Jackson et al. 2001, Myers and Worm 2003, Worm et al. 2006, Myers et al. 2007, Branch et al. 2010), but based on our literature search, studies focused on evaluating the effects of the diverse procedures used in scientific research and monitoring programs—in or outside of protected areas— have been neglected. Nevertheless, in

[1] A **protected area** is a clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long term conservation of nature with associated ecosystem services and cultural values. (**IUCN Definition** 2008)

making permitting decisions, managers need to determine the likely ecological impacts[2] of proposed scientific activities in the context of MPA goals and weigh these impacts against their potential scientific, educational, and management benefits. This need is pervasive among protected area managers and extends far beyond California's recently established MPAs, as evidenced by the permit issuance criteria used by most of the US National Marine Sanctuaries (e.g. Channel Islands National Marine Sanctuary[3], Monterey Bay National Marine Sanctuary[4]). These permit issuance criteria clearly articulate how activities that provide understanding, management, or educational benefits may be considered appropriate to conduct in the sanctuary, but only if adverse effects on sanctuary resources can be minimized.

Unfortunately, too often managers are forced to base permitting decisions on qualitative and incomplete information, in order to make subjective judgments on the expected ecological impacts of scientific projects. Similarly, scientists also don't always understand the direct or indirect effects of their proposed work on their target species or the broader ecosystem. This can lead to unanticipated impacts of scientific research on MPA biota and habitat, produce delays and inconsistencies in permit decision-making, and create difficulties for applicants attempting to understand reasons for permit rejection. Evaluating the ecological impacts of scientific activities, however, can be a daunting task because of the wide range of potential sampling or collection methods that might be proposed. These can range from minimally-intrusive visual or photographic surveys, to the placement of intrusive experimental structures, the manipulation or collection of organisms, or the complete clearing of biota from an area. Moreover, scientific activities can be lethal or non-lethal, have inadvertent effects on non-targeted species or communities, and produce impacts that extend throughout communities, particularly if a study affects species with important ecological roles, such as ecosystem engineers (Jones et al. 1994), dominant species (Grime 1987), keystone predators (Paine 1966), or other foundation species (*sensu* Dayton 1972).

Our purpose is to present a quantitative, ecologically-based decision-support tool that facilitates the ability of managers to more consistently and objectively estimate the ecological impacts of proposed scientific activities on macrobiota in MPAs. The proposed decision-support framework first breaks down a proposed project into its individual components and then for each project procedure estimates the proximate and ultimate impacts on an MPA's populations, assemblages, and physical habitats. Proximate impacts are calculated as proportionate impacts to populations, assemblages, and habitats directly resulting from the

[2] We define "**impact**" as any predicted ecological change relevant to management and attributable to a proposed research activity. Impacts may have positive or negative ecological consequences and vary across different levels of ecological organization (i.e. individuals, populations, communities, ecosystems).
[3] United States Code of Federal Regulations CFR 15 §922.74
[4] CFR 15 §922.133

scientific activity, whereas ultimate impacts are extended through the ecosystem and over time, accounting for impacts on strong ecological interactors that can indirectly affect community structure, as well as the estimated time needed for populations, assemblages and habitats to recover. The estimated ecological impacts of each individual scientific project, as measured by the ultimate impact assessment, are added to those of other projects to measure the cumulative effects of scientific work being performed or proposed for an MPA. These impacts are then compared with policy-based impact thresholds for MPA macrobiota and habitats that are established by managers. Impact thresholds are expected to generally be consistent across groups within an MPA, but may vary among MPAs depending on MPA regulations and goals, or may be modified to account for external sources of ecosystem vulnerability (e.g. oil spills, disease outbreaks, ocean acidification, or environmental change).

Although the decision-support framework presented here has been constructed with MPAs in mind, it is based upon established ecological principles and should also be applicable to any spatial or ecosystem-based approach to managing extractive or non-extractive activities in terrestrial, freshwater, and marine ecosystems. Importantly, this framework takes a precautionary approach and attempts to acknowledge uncertainty and account for data limitation. It is not designed to be prescriptive, but rather to provide a structured and quantitative framework for managers to employ when making decisions about issuing permits for scientific activities in MPAs.

The Decision Support Framework

Our suggested approach to determine whether scientific activities can be accommodated within a protected area, draws from our familiarity with scientific work and permitting taking place in the network of 124 MPAs recently established along the coast of California, USA[5]. We employed California's MPA network (Gleason et al. 2013) to inform our approach because the network contains numerous protected areas with diverse conservation goals, there is a relatively rich body of habitat and ecological information available, and intense research activity in some MPAs is leading to management concerns. Descriptions of the goals and types of MPAs represented in this network are presented in several publications (Gleason et al. 2013, Kirlin et al. 2013, Saarman et al. 2013). Case studies using the equations and models described herein are provided in Appendix A and are based on data gathered during MPA establishment and from on-going research programs taking place in California MPAs.

The framework consists of four components that constitute steps in a sequence for making permitting decisions for studies involving macrobiota (Figure 1). The first is an "MPA

[5] www.wildlife.ca.gov/Conservation/Marine/MPAs

Appropriateness" component that considers whether the proposed scientific activity is appropriate to conduct in an MPA. Appropriateness depends on several considerations related to the match between an MPA and a study's scientific goals. If the project is deemed appropriate for an MPA, the permitting decision is then informed by the "Ecological Impact Assessment" component of the framework. This component, which includes estimates of both proximate and ultimate impacts, is designed to estimate the ecological consequences of proposed scientific activities at three levels: the population, the assemblages that constitute MPA communities, and the habitat. Next, the ecological impacts of the proposed project are added to those determined for on-going or simultaneously proposed scientific activities in the same MPA to assess cumulative impacts. The second and third components of the framework allow each proposed project to not only be evaluated independently at three levels but also provide an estimate of the cumulative impacts of all potential and on-going scientific work in the MPA. The fourth component of the framework is the "impact threshold comparison", which weighs the cumulative ecological consequences of a proposed project plus all other proposed or permitted scientific activities against a policy-based impact threshold established for the MPA. If the cumulative ecological impacts of all the scientific activities in the MPA, including the impacts of the proposed project, are less than the impact thresholds for affected populations, assemblages, and habitats, then a favorable permitting decision is recommended. Here, we focus on the last three components of this decision-support framework, the individual and cumulative "ecological impact assessments" and the "impact threshold comparison" components.

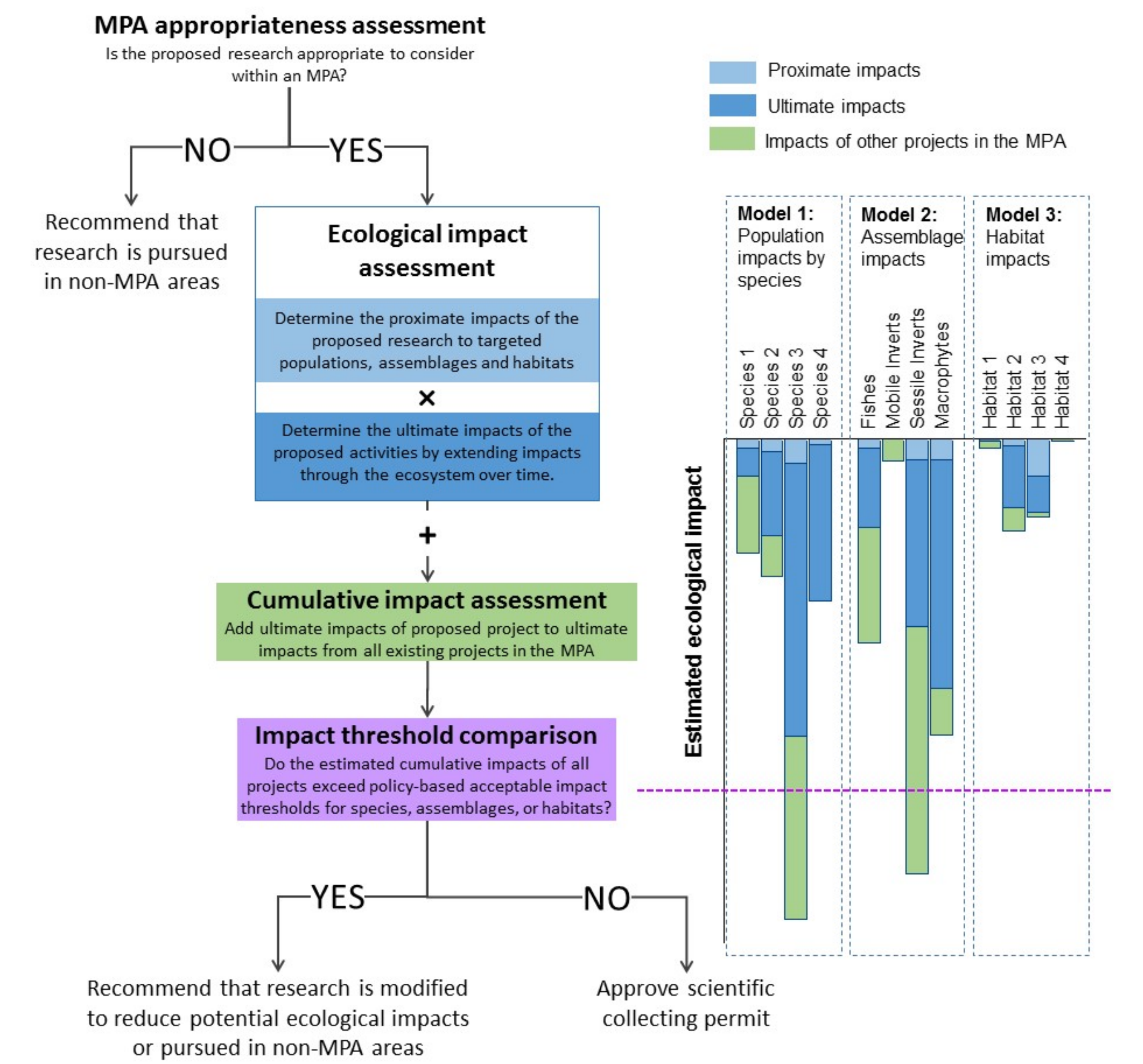

Figure 1. Decision-support framework for consideration of proposed research activities in marine protected areas, including the four key assessment elements: MPA appropriateness, ecological impacts, cumulative impacts, and comparison to thresholds of acceptable impact for each MPA. The final result of this decision framework is a recommendation that the proposed research be approved or modified to reduce impacts to levels below the impact thresholds for affected populations, assemblages, and habitat.

MPA Appropriateness Component

The first step of the proposed framework is determining whether or not the proposed project, including all of its scientific activities, is appropriate to consider permitting within an MPA. In general, scientific activities are only deemed appropriate within an MPA if they are relevant to the MPA's protections, needed to maintain the integrity of long-term monitoring programs, not feasible to conduct elsewhere, or important and of sufficiently low impact to not interfere with MPA goals (Table 1). These criteria are in close alignment with the permit issuance criteria used by the Channel Islands National Marine Sanctuary[6], and other National Marine Sanctuaries. There are many reasons why a scientific activity might require the ecological protection afforded by an MPA. For example, the MPA could be essential to the proposed research design because of its designation (i.e. the project requires a protected population) or location (i.e. the project requires an organism or habitat not readily available outside the MPA). The need to monitor MPA performance or continue established long-term sampling programs that meet regulatory requirements or inform resource management may also serve to justify performing work in an MPA. Potential conflicts between MPA establishment and on-going fisheries and other survey and assessment programs (Field et al. 2006, Mcgilliard et al. 2015, Ono et al. 2015) highlight the need to assess the impacts of such research and make informed decisions about their continuation. In addition, low-impact educational activities can be considered as scientific activities when these occur in an MPA located near an educational institution or scientific facility or when they cannot readily be conducted elsewhere because of logistical constraints.

Table 1. Examples of reasons why proposed scientific research and educational activities might be appropriate within an MPA.

- Research is consistent with and facilitates MPA goals (i.e. necessary for application of MPA as a management tool).
- Research is being done to evaluate the effectiveness of an MPA in achieving management objectives and to inform management.
- Focus of research is on the ecological or socio-economic effects of MPAs separate from their management objectives.
- Research requires a protected population or ecosystem.
- Target species, assemblage, or ecosystem is locally rare, and not readily found outside of local MPAs.
- Research is the continuation of a long-term monitoring program or research project, particularly if the program precedes protected area establishment.
- Protected area has unique accessibility, for example co-location with a research facility or other research infrastructure, and is important to institutional scientific and educational work.

Ecological Impact Component

For projects deemed appropriate to conduct in MPAs, we estimate the ecological impacts of scientific activities using three ecological models (Figure 1). These models address effects of

[6] United States Code of Federal Regulations CFR 15 §922.74

proposed projects on an MPA's: 1) population(s) of targeted species or, if necessary, taxonomic or ecologically-meaningful groups of macrobiota (e.g. *sensu* Steneck and Dethier 1994, Wilson 1999); 2) ecological assemblages of macrobiota; and 3) physical habitat(s). In all three models, proximate impact is expressed as a proportion of the available population, assemblage, or habitat located within a protected area's boundaries. We assess the proximate impacts of all scientific activities proportionately because MPAs vary widely in the size and composition of species, assemblages and habitats. Thus, our approach allows for individualized impact assessment because it is based on the actual physical and biological composition of each MPA.

In addition to calculating the proximate impacts of proposed scientific activities, we also calculate the ultimate impacts extended through the ecosystem and over time by taking into account: 1) effects on species with important ecological roles—e.g. keystone predators or foundational species; and, 2) the recovery times of impacted populations and habitats. Thus, each of the three ecological impact models generates two outputs: the proximate and ultimate impacts. Reporting the proximate impacts, which are strictly proportionate, helps maintain transparency in the models and aids interpretation of results, but the ultimate impacts, which are modified proportions and thus best represented as unitless numbers, are used to assess cumulative impacts, compare effects of proposed scientific activities against the impact thresholds, and inform permitting decisions. The three ecological impact models also address direct and indirect effects of proposed scientific activities. This is important because often scientific activities have not only direct effects on an MPA's populations, communities, and habitat but also unintended or incidental and indirect effects that must be taken into account.

The population model (**Equations 1.1 and 1.2**) addresses direct impacts to the population(s) of targeted macrobiotic species or groups and is only used in cases where the scientific activity identifies a specific target. In cases where no target species or group is identified, the population model is omitted, and all impacts are estimated using the assemblage (second) and habitat (third) models.

The assemblage model (**Equations 2.1 and 2.2**) accounts for direct and indirect (i.e. incidental) effects, depending on its application. Examples of indirect effects include the unintended catch of other fishes with non-selective sampling methods (e.g. hook and line, nets) and incidental mortality or dislodgement of non-targeted sessile organisms, including epifauna, while collecting targeted sessile species with hand tools. The assemblage model also assesses direct impacts in cases where no target is identified, and study procedures are instead designed to affect multiple species or sample a cross-section of the community (e.g. beach seining to sample the fish assemblage or clearing plots of all organisms in the rocky intertidal to investigate succession). The assemblage model considers the effects on four assemblages that constitute communities of macro-organisms in coastal habitats: macrophytes, sessile invertebrates, mobile invertebrates, and fishes. These impacts are computed and evaluated independently for each assemblage and not combined, reflecting the inherent difficulties in modelling ecological impacts using a single community parameter. Thus, when assessing cumulative impacts, the impacts of the proposed and all existing projects are summed within each assemblage, but not across assemblages.

The third model, the habitat model (**Equations 3.1 and 3.2**), assesses the direct and indirect impacts to physical habitat and is applied to all proposed studies. In addition to impacting MPA macrobiota, scientific activities also can create short and long-term impacts on physical habitat, which are captured by the habitat model.

Ecological Impact Models

To evaluate a proposed research project using the ecological impact models, the project must first be broken into its component procedures, including the numbers of organisms to be collected, the species or groups targeted, and the methods used. Some projects may include a number of different targets and methods, each of which should be evaluated independently, and the cumulative impacts of all the project components considered against the impact thresholds. In cases of uncertainty (e.g. the researcher will attempt to capture organisms using several methods, but doesn't know how many will be captured with each method), the models should be parameterized conservatively (e.g. using the most impactful realistic combination of methods from those proposed).

Impacts on populations of targeted species. The population model is used when researchers target one or more particular species and consists of two different impact estimates. The first, the proximate impact assessment (**Equation 1.1**) makes a quantitative estimate of the impact of the proposed scientific activity on the targeted species considering lethal effects of the proposed sampling method(s), handling effects on affected organisms during and following sampling, and the efficacy of the sampling method in collecting targeted organisms. These effects are then placed into proportional context by considering the quantity of individual organisms impacted relative to the estimated size of the population within the MPA (**Equation 1.1**). Once calculated, the estimate of proximate impacts is then extended through the ecosystem and over time by accounting for the ecological role of the targeted organisms and their recovery times to derive an estimate of ultimate impacts (**Equation 1.2**). The population model does not estimate unintentional or incidental impacts of targeted take on the community and is not applied to study methods that are designed to sample multiple species or entire assemblages; these impacts are considered in the impacts on assemblages model.

The proximate impact of proposed research activities on a targeted species or group ($PI_{targ\ i}$) is generated as:

(Equation 1.1)

$$PI_{targ\ i} = \left\{\{M_{meth\ i} + [(1 - M_{meth\ i}) \times M_{hand\ targ\ i}]\} \times \left(\frac{1}{Eff_{meth\ i}}\right)\right\} \times \frac{N_{targ\ i}}{Dens\ or\ \%\ cover_{targ\ i} \times A_{MPA\ hab\ i}}$$

Where,

$M_{meth\ i}$ is the proportionate mortality of the targeted species or group *i* subjected to study method *i*.

1-$M_{meth\ i}$ is the proportion of individuals of the targeted species or group *i* subjected to but not killed by method *i*.

$M_{hand\ targ\ i}$ is the proportionate mortality caused by handling the targeted species or group *i* subsequent to capture.

$Eff_{meth\ i}$ is the proportionate success of the study method in collecting the proposed number of individuals (i.e. $N_{targ\ i}$ / total number collected) of the targeted species or group.

$N_{targ\ i}$ is the proposed number of individuals or percent cover of the targeted species or group *i* collected with method *i*.

$Dens_{\ targ\ i}$ *or % cover*$_{targ\ i}$ is the density (individuals per unit area) or area-based percent cover of the targeted species or group *i* in its appropriate habitat within the MPA.

$A_{MPA\ hab\ i}$ is the area of appropriate habitat for the targeted species or group *i* within the MPA.

To calculate the ultimate impacts to targeted populations as they extend through the ecosystem and over time, the proximate impact $PI_{targ\ i}$ from **Equation 1.1** is used in our model to calculate the ultimate impact $UI_{targ\ i}$ using **Equation 1.2**

The ultimate impact to each target species ($UI_{targ\ i}$) is calculated as:

$$UI_{targ\ i} = PI_{targ\ i} \times \frac{RT_{targ\ i}}{2} \times Interaction_{targ\ i}$$ (**Equation 1.2**)

Where,

$PI_{targ\ i}$ is the estimated proximate impact to the population of target species *i* in the MPA from **Equation 1.1**.

$RT_{targ\ i}$ is the estimated recovery time for target species *i*. Recovery time is estimated for each species based on life history parameters and is not determined by the extent of the impact.

$Interaction_{targ\ i}$ is an index of the ecological importance of target species *i*. By default, any species not identified as a strong interactor receives an interaction index equal to one.

<u>*Impacts on assemblages.*</u> The assemblage model assesses the community-wide impacts of the proposed scientific activities, including the incidental impacts of studies targeting individual species, and the impacts of study procedures that are designed to affect multiple species or sample a cross-section of the community. The assemblage model also consists of proximate and ultimate impact estimations, which are computed independently for each of the four assemblage groups—macrophytes, sessile invertebrates, mobile invertebrates, and fishes. The proximate impact assessment (**Equation 2.1**) makes a quantitative estimate of the impacts of the proposed scientific activity on each assemblage, considering the susceptibility of assemblage-members to the proposed sampling methods, the lethal effects of those sampling methods, and effects of subsequent handling of targeted and non-targeted organisms. The model assumes that each assemblage is distributed evenly throughout the area of appropriate habitat within the MPA, thus the proportion of each assemblage encountered by the proposed sampling method is equal to the proportion of available habitat sampled. Once calculated, the proximate impacts for each assemblage are then extended by incorporating the ecological roles of species within the assemblages and the assemblage recovery times to derive an estimate of ultimate impacts (**Equation 2.2**).

The proximate impact of proposed research activities on each assemblage ($PI_{assemb\ i}$) is generated as:

(Equation 2.1)

$$PI_{assemb\ i} = \left\{M_{meth\ i} + \left[(1 - M_{meth\ i}) \times M_{hand\ non-targ}\right]\right\} \times (Suscep_{meth\ i}) \times \frac{A_{samp\ hab\ i}}{A_{MPA\ hab\ i}}$$

Where,

$M_{meth\ i}$ is the proportionate mortality of assemblage *i* subjected to method *i*.

1- $M_{meth\ i}$ is the proportion of assemblage *i* subjected to but not killed by method *i*.

$M_{hand\ non-targ}$ is the proportionate mortality caused by handling non-target species within assemblage *i*. In most cases, this is simply the mortality associated with catch and release.

$Suscep_{meth\ i}$ is the proportion of an assemblage within the sampling area that is susceptible to take by method *i*.

$A_{samp\ hab\ i}$ is the area of habitat *i* subject to sampling method *i*. This area may be proposed by the applicant (for area-based or community-wide studies) or inferred from the density of targeted species or groups.

$A_{MPA\ hab\ i}$ is the area of appropriate habitat for assemblage *i* within the MPA.

Estimating the area impacted by proposed scientific activities ($A_{samp\ hab\ i}$) can be very straightforward when the study uses an explicit spatial design. For example, if a study samples ten 1.0 m^2 plots in a rocky intertidal habitat, then $A_{samp\ hab\ i}$ is 10 m^2. If this same sampling is to occur four times per year with new plots during each sampling period, $A_{samp\ hab\ i}$ is 40.0 m^2. If the identical plots or areas are to be sampled during each site visit, $A_{samp\ hab\ i}$ would be 10.0 m^2 because the actual amount of affected habitat is not increased by repetitive sampling of the same location.

For studies that don't use an explicit spatial design, particularly those that target a particular species, an investigator may have difficulty estimating how much habitat will be sampled to obtain the required number of organisms. For example, if 25 individuals of a fish species are to be taken by hook and line on three occasions during the year, how much habitat will need to be sampled? In such cases, $A_{samp\ hab\ i}$ is calculated based on the number of individuals targeted ($N_{targ\ i}$), the abundance of the target species (*Dens or % cover*$_{targ\ i}$), and an ad-hoc scalar to account for sampling inefficiencies, as shown in **Equation 2.1a**.

(Equation 2.1a)

$$A_{samp\ hab\ i} = \frac{N_{targ\ i}}{Dens\ or\ \%\ cover_{targ\ i}} \times 5$$

In our example, $N_{targ\ i}$ is 75 (i.e. 25 fish, three times per year) and $Dens_{targ\ i}$ is the density of the target fish in the sampled habitat, in this case 0.1/m^2. Thus, the 75 fish targeted are likely to occupy an area of at least 750 m^2. However, the investigator will likely have to fish more than 750 m^2 of habitat to obtain his samples due to sampling inefficiencies. In the absence of better information from the literature, we used an ad-hoc scalar of five to represent these sampling inefficiencies. Thus, in this example, the area sampled would be 3,750 m^2 (i.e. 750 m^2 × 5). The inefficiency multiplier of five produces a conservative but reasonable magnification effect for most targeted sampling methods, but could readily be modified if better information is available.

The ultimate impact to each assemblage ($UI_{assemb\ i}$) that constitutes the community is calculated via **Equation 2.2** using the proximate impact ($PI_{assemb\ i}$) from **Equation 2.1.**

$$UI_{assemb\ i} = PI_{assemb\ i} \times \frac{RT_{assemb\ i}}{2} \times Interaction_{assemb\ i} \qquad \textbf{(Equation 2.2)}$$

Where,

$PI_{assemb\ i}$ is the estimated proportionate impact to the assemblage in the MPA from **Equation 2.1.**

$RT_{assemb\ i}$ is the recovery time in years of assemblage *i*.

$Interaction_{assemb\ i}$ is an index of the ecological importance of assemblage *i*.

Impacts to habitats. The habitat model assesses impacts of scientific research activities on the physical structure of a habitat and also incorporates proximate and ultimate impacts. Proximate impacts to the habitat ($PI_{hab\ i}$) are estimated considering the probability that a scientific sampling method will alter the physical habitat and the proportion of the available habitat that will be sampled (**Equation 3.1**). These estimated proximate impacts are then extended over time based on the recovery time of the impacted physical habitat (**Equation 3.2**).

The proximate impact of the proposed scientific activities on the physical habitat ($PI_{hab\ i}$) is generated as:

(Equation 3.1)

$$PI_{hab\ i} = P_{alt\ hab\ i\ meth\ i} \times \frac{A_{samp\ hab\ i}}{A_{MPA\ hab\ i}}$$

Where,

$P_{alt\ hab\ i\ meth\ i}$ is the probability (0 to 1) that sampling method *i* will alter habitat *i*.

$A_{samp\ hab\ i}$ is the area of the habitat *i* subject to sampling method *i*. As in **Equation 2.1**, this area may be proposed by the applicant (for area-based or community-wide studies) or inferred from the density of targeted species or groups.

$A_{MPA\ Hab\ i}$ is the area of habitat *i* within an MPA.

As described in the section on impacts to assemblages, $A_{samp\ Hab\ i}$ may either be provided by the applicant for area or community-based studies, or inferred from the number and density of target organisms as described in **Equation 2.1a**.

The ultimate impact to each habitat ($UI_{hab\ i}$) is calculated as:

(Equation 3.2)

$$UI_{Hab\ i} = PI_{hab\ i} \times \frac{RT_{hab\ i}}{2}$$

Where,

$PI_{hab\ i}$ is the estimated proportionate impact to the habitat in the MPA from **Equation 3.1**.

$RT_{hab\ i}$ is the recovery time of the physical habitat in years.

We elected not to represent the ecological importance of physical habitats with an interaction index, because all physical habitats are of vital importance to their inhabitants, and we felt that attempting to differentiate more and less important habitats would be meaningless, thus the ultimate impact is modified by recovery time only.

Model Parameters. Parameterizing the three ecological impact models requires inputs on: 1) impacts of study methods; 2) macrobiota abundance; 3) habitat abundance; 4) species with important ecological roles; and 5) recovery times for species, assemblages, and habitats. Whereas the impacts of study methods, ecological roles, and recovery times are likely to be relatively consistent inside and outside of protected areas, species and habitat abundances are specific to each MPA, and should be estimated for every MPA where proposed scientific work is to be undertaken in order to determine the proportionate impacts on which our models are based.

Because of the importance of maintaining MPA protection, we consistently used a precautionary approach in developing and parameterizing the ecological impact models. This precautionary philosophy frequently conflicted with the need for simplicity and generalization in the face of limited information. For example, precisely estimating method-related mortality for each potential target species was neither feasible nor supported by the current body of scientific knowledge; however, it was important not to dramatically underestimate mortality for any species. Hence, we used a suite of approaches described in Appendix B, including grouping organisms and study methods and assigning categorical values to these groups using expert judgment approaches.

Impacts of study methods. Scientists use a large variety of methods in performing their studies and these methods can have impacts on macrobiota and habitat depending on the nature of the project and the particular species, assemblage, or habitat being studied. In the three models, the impacts of study methods are expressed as a probability of mortality for organisms, and probability of alteration for habitats. Sublethal effects and minor habitat alterations are not explicitly addressed, except as a low probability of mortality or alteration. For the purposes of these models, study methods are defined as all means of performing scientific work, including observation, capture, handling or manipulation, relocation, and sacrifice of organisms. Habitat alterations, both intentional and unintentional, are also considered, including addition of artificial structures, removal or reconfiguration of physical habitat and alteration of bottom habitat through contact with sampling gear (e.g. dredges, trawl nets, hand tools).

The impacts of study methods on organisms are articulated as a function of four factors. First, the mortality caused by the sampling method itself (M_{meth}); in the case of purely observational studies this mortality is zero or near-zero. Second, the mortality caused subsequent to collection due to handling (M_{hand}); for example, tagging captured fish prior to release. Third, any mortality caused by limited sampling efficacy (Eff_{meth}); for example, if a study required only females for gamete analysis, but sex was impossible to determine without harming the organisms, sampling efficacy could be 0.5. And fourth, the susceptibility of organisms to the particular sampling method employed ($Suscep_{meth}$); this factor determines how sampling and handling mortality should be applied to non-target organisms in the community. $Suscep_{meth}$ is defined as the proportion of an assemblage that is susceptible to take by a particular sampling method. For example, a susceptibility value of 0.25 for the fish assemblage indicates that 25% of fish are vulnerable to incidental capture by the sampling method, thus the mortality associated with the sampling method (M_{meth}) is applied to 25% of the fish assemblage in the sampling area.

The impacts of study methods on habitats are articulated simply as the probability of altering the physical habitat ($P_{alt\ hab\ meth}$). Scientific activities may intentionally or unintentionally alter the physical or chemical characteristics of an ecosystem, however, the most common effects of scientific activities on the abiotic environment result are changes to the structure of the physical habitat. Hence, for simplicity, our framework focuses exclusively on the potential impacts of scientific work resulting in modifications to physical features of the environment; chemical effects of scientific projects are not treated in our model and will require separate consideration if proposed. We considered scientific procedures such as bottom trawling that scar bottom habitat, and the addition, removal, or reconfiguration of physical habitat, which alters the availability of surfaces, cracks, and crevices for species to populate.

To parameterize the models with information about the impacts of study methods, we relied extensively on expert judgment, because data and literature were unavailable for quantifying the impacts of scientific research methods on most organisms. This reflects a pragmatic response to data limitation, not an essential element of the framework. For the sake of simplicity, we estimated the per capita mortality rates of particular scientific procedures for large groups of organisms, not individual species. Our groupings closely mirror the assemblages used throughout the models: macrophytes, mobile invertebrates, sessile invertebrates, and fish, with a further subdivision of the fish assemblage to account for pressure-related mortality in fish with swim bladders. Rather than attempting to precisely estimate mortality rates, we assigned categorical mortality values for each method-group combination, and attempted to be conservative in these assignments. In most cases, the categorical assignments (e.g. “high” mortality) were translated to a range of values (e.g. 33-66%), and the conservative end of that range was then used as the model parameter. Examples of mortality estimates for scientific study methods and a description of our categorization approach are described in Appendices B and C.

As with the other parameters that reflect the impacts of scientific study methods, there is very little literature that can be used to calculate the probability of habitat alteration associated with study methods ($P_{alt\ hab\ meth}$). Thus, we also used an expert judgment approach to assign categorical probabilities of altering the physical (not biogenic) habitat. These categories were then translated to ranges of values, and the conservative end of the range was used in the models (see Appendix C for more details).

Species abundance. Estimating the impacts of scientific study procedures in our model requires density or percent cover data ($Dens_{targ}$ *or %* $cover_{targ}$) on species abundances within an MPA in order to calculate the proportionate effects of the project. Ideally, estimates of density or percent cover of a species or taxonomic group will be available for an MPA. However, if existing data are unavailable, limited, or likely inaccurate, the best available abundance estimates for the MPA should be obtained either empirically through non-destructive pilot surveys, from the literature, or from data taken from surveys performed in nearby, comparable habitat.

In some MPAs, such as many in California's MPA network, species abundance estimates are available from multiple sampling events that include spatial and temporal components. In keeping with our conservative approach, we used a nonparametric bootstrapping approach with estimates of density or percent cover across all spatial and temporal sampling events each year, and used the lower quartile of the bootstrapped results to provide abundance estimates for model input. This method generates a conservative density or cover value based on all of available empirical data, albeit with two important limitations. First, this method does not account for temporal trends in density or percent cover, thus abundance estimates obtained in this way should be used with caution when there is evidence of temporal trends. Second, abundance estimates of zero can often occur for a number of species-MPA combinations, which can result either from the failure of the sampling methods to detect low densities of a targeted species or its true absence from the MPA. In cases where the best available species density or cover estimates are zero, the applicant may be asked to provide an empirical abundance estimate using non-destructive means to inform the impact assessment models.

Habitat abundance. Habitat abundance data (area, $A_{MPA\ hab}$) are also needed to populate the impact assessment model and to extrapolate organism and assemblage abundances. We extrapolate species abundances using habitat-specific density or cover estimates, and assume that assemblages are habitat-specific and uniformly distributed across the habitat.

To estimate habitat abundance, we first categorized habitat types using three features known to strongly influence the distribution and abundance of marine populations and communities: geomorphology, depth, and proximity to the sea floor. The quality and quantity of data available for estimating habitat area varied from MPA to MPA in California's MPA network and was constrained by available mapping data so we employed a simple binary classification of sediment or rock. Sediment habitats include mud, sand, and gravel substrata, whereas rock habitats include bedrock, boulder, and cobble. The selected depth categories used in our model reflect ecologically meaningful categories (i.e. intertidal, 0-30 m, 30-100 m, > 100 m) and parallel those used in the design of the California's MPA network (Allen et al. 2006, MLPA Science Advisory Team 2008, 2009, 2010, CDFW 2016). We also used proximity to seafloor, a feature that distinguishes pelagic habitat from demersal or benthic habitat. However, because of the strong interaction between pelagic and benthic ecosystems in shallower depths, pelagic habitats were considered distinct from their underlying benthic habitats only at depths greater than 30 m. When combined, these features collectively generated ten distinct habitat categories (Table 2).

Table 2. Coastal marine habitat categories.

Depth (m)	Rock	Sediment	Water column
Intertidal	rocky intertidal	sandy beaches; marsh and mudflats	NA
0-30	shallow reef and kelp forests	estuaries; open coast soft-bottom	NA
30-100	mid-depth rocky reefs	mid-depth soft-bottom	shallow pelagic
> 100	deep rocky reefs	deep soft-bottom	deep pelagic

The habitat data collected and compiled in association with MPA establishment (MLPA Science Advisory Team 2008, 2009, 2010, Golden 2013, Young and Carr 2015a) served as a model for estimating habitat abundance (area) in California's MPAs. For offshore locations, habitat areas were obtained using high-resolution digital elevation models, raster datasets that consist of depth values at regularly spaced intervals (e.g. 2m and 5m), produced by the California Seafloor Mapping Project (Golden 2013). Along the shoreline (including intertidal habitats), the best habitat data available for California MPAs was represented by a linear shoreline feature obtained from National Oceanic and Atmospheric Administration (NOAA) Environmental Sensitivity Index maps. This linear feature was classified into four simple categories (rocky intertidal, beach, estuarine mud flats, and salt marsh) and used as a linear measure of habitat availability or converted to area using the mean width of the intertidal zone multiplied by shoreline length. However, even for California MPAs, mapping gaps exist, most notably a narrow nearshore habitat band extending the entire length of the coastline where substrate data are difficult to collect because of navigation hazards (shallow water, kelp, wave action) that preclude vessel-based mapping. To ensure that species and assemblage abundance estimates were as accurate as possible, we did not ignore substrate availability in this zone, but estimated it by interpolation using substrate information from the adjacent shoreline and offshore zones (Saarman et al. 2015).

Species with important ecological roles. A primary goal of most protected areas is to protect not just individual species but the structure and function of entire ecosystems. Because each species plays a distinct ecological role, it is important to consider all the species potentially affected when estimating the ecological impacts of proposed scientific activities, and particularly those known to strongly affect community structure through their interactions with other species. We addressed this consideration in our ecological impact assessment models through the calculation of ultimate impacts, which take into account effects on species with important ecological roles. This approach is consistent with a fundamental tenet of ecosystem-based management—to adopt measures that ensure the ecological functions of species are sustained (Grumbine 1994, Pikitch et al. 2004, McLeod and Leslie 2009, Belgrano and Fowler 2011). Examples of species with important ecological roles (Table 3) include structural species and ecosystem engineers (*sensu* Jones et al. 1994) that form or influence biogenic habitat and alter the physical environment (e.g. mussel beds, kelp forests, corals, seagrass beds). Some of these species, including keystone species, have ecosystem-wide effects that are disproportionate to their abundance (Paine 1966, Holling 1992, Power et al. 1996).

The functional roles of foundation species are largely manifest through interactions with other species and the strength of these interactions varies markedly. Our assessment of ultimate impacts includes an estimation of the strength of these interactions for species likely to be impacted by proposed scientific work. Some species are strong interactors whose interactions (predation, competition, facilitation) result in cascading effects that extend throughout much of the ecosystem. To ensure that important species interactions are accounted for in assessing ultimate impacts. Our approach was to (i) identify important species interactors in the MPA from the literature, (ii) categorize potential strong interactors by their interaction types (see Table 3), (iii) qualitatively assign strengths for each interaction type, (iv) sum the total interaction scores across all categories and, (v) translate these scores to an appropriate scale, termed the "interaction index" ($Interaction_{targ\ i}$). Because the list of strong interactors within each assemblage-habitat combination is small (typically less than 10), determining if any are likely to be susceptible to a specific method is feasible. In keeping with our precautionary approach, the interaction index used for each assemblage is equal to the highest interaction index of any species in the assemblage that may be susceptible to the study methods employed. In situations of uncertainty, we conservatively assumed susceptibility of all species in the assemblage and used the strongest interaction score. Our procedures for treating interaction strength are described and estimates are provided for several common species and species groups in Appendix D.

Table 3. Important species interactions for macrobiota that should be accounted for when estimating ultimate impacts. Descriptions of many of these categories are adopted from Ellison et al. (2005). Examples are drawn from coastal marine ecosystems.

Interaction	**Description and examples (coastal marine)**
Keystone predators	Species whose ecological effects are disproportionately large relative to its abundance, manifest by the preferential consumption of ecologically significant species (e.g. foundation species, ecosystem engineers) with ramifications to the state of an ecosystem (Paine 1966, 1969, Power et al. 1996). Examples include the intertidal sea star, *Pisaster ochraceus*, the subtidal sea star, *Pycnopodia helianthoides*, the sea otter, *Enhydra lutris*.
Structural species (biogenic habitat)	Species whose growth form produces habitat used by other species. Distinct from autogenic engineers in that the influence of structural species is generally confined to their 3-dimensional footprint. Examples include most macroalgae, mussels, corals, tubeworm colonies, seagrasses whose physical structure is inhabited by other species (invertebrates, fishes, epiphytic algae).
Ecosystem engineers (autogenic)	Species whose physical structure influences other species by modifying the physical or chemical environment beyond their 3-dimensional footprint (*sensu* Jones et al. 1994). Examples include kelps and corals that modify water movement or light attenuation in the subtidal, or temperature and desiccation in the intertidal.
Ecosystem engineers (allogenic)	Species that alter their environment through action on another organism (*sensu* Jones et al. 1994). Examples include sea urchins that influence the abundance of algae as sources of biogenic habitat, or modify coral and rocky reef structure, limpets that create mosaics of open space in mussel beds, parrotfishes that alter coral structure and generate sand.
Facilitators (other than biogenic habitat)	Species whose interactions with others are either mutualistic or commensalisms, benefiting at least one of the participants and causing harm to neither (Stachowicz 2001). These positive interactions extend beyond those directly linked to the structural influence of the species. For example, coralline algae generate settlement cues for many invertebrates, algae reduce stressful environmental conditions in the rocky intertidal,
Dominant species (competitors)	Species that competitively exclude subordinate species (Grime 1987), garner a disproportionate share of resources and modify the structure and functional processes in ecosystems. Examples include mussels in the rocky intertidal, colonial anemones, surface forming or sub-canopy kelps that out-compete shorter stature algae.
Trophic importance (food-chain support)	Species that create important links in trophic pathways, thereby influencing how nutrients and energy are incorporated into and pass through food webs. Examples include abundant planktivores and detritivores that create plankton and detrital-based trophic pathways, abundant herbivores that make primary production available to higher trophic levels. Examples include large schools of planktivorous fishes, and herbivorous crustaceans that are preyed on by fishes.

Recovery time for species and assemblages. The duration of impacts from scientific activities will vary greatly depending on the rate at which affected species and assemblages are able to recover their abundances and ecological roles. For example, impacts on long-lived species or those with low reproductive rates or infrequent larval recruitment events are likely to have long-lasting ecological effects compared with impacts on short-lived species with high reproductive rates and frequent larval recruitment events. Not only will the ecological impacts last longer, but populations with long recovery times are likely to be more vulnerable to small population perturbations. We incorporated impact duration into our model (RT_{targ}) by examining the time to recovery in years for species and assemblages affected by scientific study procedures. Because recovery of affected populations is likely to be incremental, we incorporate recovery time into the model by multiplying the proportionate impact by one half of the recovery time ($RT_{targ}/2$). This approach assumes a linear recovery from the time of the impact to the end of the recovery time.

Our working definition of recovery time for populations and assemblages was replacement of the abundance (density or percent cover) and size-structure of individuals removed, to reflect the lost density- and size-dependent functional roles of impacted species. We considered only lethal impacts in estimating effects on organisms and assemblages. Recovery at the local scale could involve immigration of older life-stages, vegetative encroachment, or the recruitment, growth, and survival of propagules. We did not consider replacement by immigration of older life-stages of mobile organisms or vegetative encroachment as recovery, because net loss to the population or assemblage in the MPA would still occur if replacement occurs at the local scale. Rates of recovery by propagules depend on a complex combination of factors, and generic estimates are available only for a handful of species. Hence, we used a suite of alternative approaches for estimating recovery time based on the natural mortality rates of individual species using the equations developed by Hoenig (1983) to estimate natural mortality based on other life history parameters. In keeping with our precautionary approach, we assumed that the recovery time of an assemblage (RT_{assemb}) was equal to that of the slowest-recovering organism in that assemblage. The details of these procedures and examples of estimates of recovery time for a variety of species and assemblages are described in Appendix E.

Recovery time for physical habitat. Like populations and assemblages, impacted physical habitat will take some period of time to recover (RT_{hab}). The rate at which the habitat returns to pre-perturbed conditions, will vary with the composition of the habitat and the nature and spatial extent of the scientific activity just as the biotic recovery time will be species dependent. For example, trawling on soft bottom (e.g. mud, sand, or gravel) will likely modify bottom habitat only temporarily (Lindholm et al. 2004), whereas trawling on hard, rocky surfaces (e.g. cobble, boulder or contiguous rock reef) can modify a habitat more permanently (Tissot et al. 2008). Like recovery of populations, habitat recovery is likely to be incremental as physical forces (e.g. waves, currents) gradually restructure habitats, so we incorporate habitat recovery time into the model by multiplying the proportionate impact by one half of the recovery time ($RT_{hab}/2$).

Habitat recovery durations were estimated as a continuous variable (in years) by experts familiar with each habitat type (see Appendix E for details). For some types of habitats (i.e. rock substrates), the habitat alterations are likely to be longstanding or even permanent unless actively reversed. However, for pragmatic considerations we capped RT_{hab} at 20 years in our model, but recognize that the cumulative impacts in such cases may last much longer and, therefore, should trigger additional scrutiny. This approach and estimates of the recovery time for a variety of habitats and scientific procedures are described in Appendix E.

Impact Threshold Comparison

Determining an acceptable level of ecological impact is a policy decision that may vary among species, ecosystems and MPAs, but it is only by comparing estimated impacts to this threshold, that the decision support framework provides permitting guidance. Impact thresholds should be set by managers and take into account, among other things, the goals of the MPA, effects of large-scale forces like ENSO events, and any known extractive activities allowed in the MPA (accounting for illegal extraction, i.e. poaching, is problematic). In cases and areas where poachers are caught and the illegal amount of take known this should be accounted for in future allocation of take. The design of the framework, however, allows managers to set a single threshold that applies to all the populations, assemblages, and habitats within the MPA. This is possible because the relevant biological and ecological factors (e.g. recovery time and ecological role) that might influence such a threshold are already incorporated into the estimation of ultimate impacts. Although the setting of impact thresholds will be a challenge for any marine system, as a starting point we suggest that managers limit the cumulative impacts of scientific activities in an MPA (as estimated by the cumulative ultimate impacts in the three models) to no more than 0.1, for any population, assemblage, or habitat. Although it is tempting to refer to the ultimate impacts and impact thresholds as proportions or percentages, the inclusion of recovery time and ecological role make this characterization misleading, thus we refer to ultimate impacts and their corresponding threshold as a unitless number.

Our framework was modeled in part on previous risk assessment frameworks implemented to allow for *de minimus* mortality of vulnerable populations. In recognition of the need to allow for minimal incidental mortality of marine mammals in fisheries and other marine activities, the National Marine Fisheries Service developed the concept of potential biological removal (PBR) as a maximum mortality threshold to be implemented with the recognition that mortality was to be avoided and minimized to the extent practicable. The PBR threshold was developed based on a the minimum population size estimate for a given stock (the 20th percentile of abundance estimates was used in light of uncertainty), the maximum population growth rate, and a recovery factor that accounts for additional sources of uncertainty and bias. In development of our models and threshold guidance, we borrowed several aspects of the PBR approach: 1) our conservative estimates of species abundance (lower quartile of bootstrapped distribution of annual means) was derived from the use of minimum population size, 2) the recovery times used in calculating ultimate impacts function similarly to the population growth rates, and 3) we used the PBR framework to put the potential impact thresholds in context. Using the PBR approach, Wade (1998) generated values for a variety of pinnipeds and cetaceans and these values range from 6% of the minimum population estimate removed annually for relatively abundant species of concern (sea lions, elephant seals, harbor porpoises) to 0.01% for rare cetaceans (blue whale). Given this range of PBR values for species with slow growth rates relative to fishes, invertebrates, and algae, we view an ultimate impact threshold of 0.1 (which could be realized through extraction of a maximum of 10% of the population of a short-lived species or as little as 0.13% of the population of a long-lived species with a strong ecological role), to be a conservative starting point for setting impact threshold levels.

Sensitivity Analyses

To visualize the relationships between input variables and output values in our models, we graphed a series of relationships to show how estimated proximate impacts and their corresponding ultimate impacts respond to varied parameter inputs (Figure 2). Each input variable, illustrated by a separate line, was varied between its minimum and maximum possible value (x-axes), while all other input variables were held constant and the resultant output value was plotted on the y-axis. In addition to plotting the effects of individual input variables, we also plotted the combined effect of varying all input variables simultaneously. In the case of the proximate impact equations (**Equations 1.1, 2.1, 3.1**), any input values held constant were set to the median from the distribution of actual values and the proportion of the population targeted was set to a constant of 5% to ensure that output values were within a realistic range. In the case of the ultimate impact equations (**Equations 1.2, 2.2, 3.2**), we used a constant proximate impact of 1% as the input. The shape of each relationship illustrates the sensitivity of the output value to that input parameter, with steeper slopes indicating greater sensitivity.

For proximate impacts at the population level (Figure 2a), method and handling mortalities ($M_{meth\ i}$ and $M_{hand\ targ\ i}$, respectively) exhibit linear relationships and efficacy ($Eff_{meth\ i}$) a curvilinear relationship to the output value. Of the variables with linear relationships, $M_{meth\ i}$ most strongly affects the output value; however, the curvilinear relationship to $Eff_{meth\ i}$ surpasses method mortality at low levels of efficacy. Thus, the proximate calculated impact to the population is most sensitive to method mortality except at low levels of sampling efficacy. Since most common scientific study techniques have relatively high efficacy and there are multiple factors that discourage ineffective sampling, these results suggest that accurate estimates of method mortality are of particular importance for estimating impacts at the population level. In contrast, when the ultimate impacts to populations are calculated, incorporating recovery time and species ecological roles (Figure 2d), the ultimate impact at the population level is most sensitive to recovery time ($RT_{targ\ i}$).

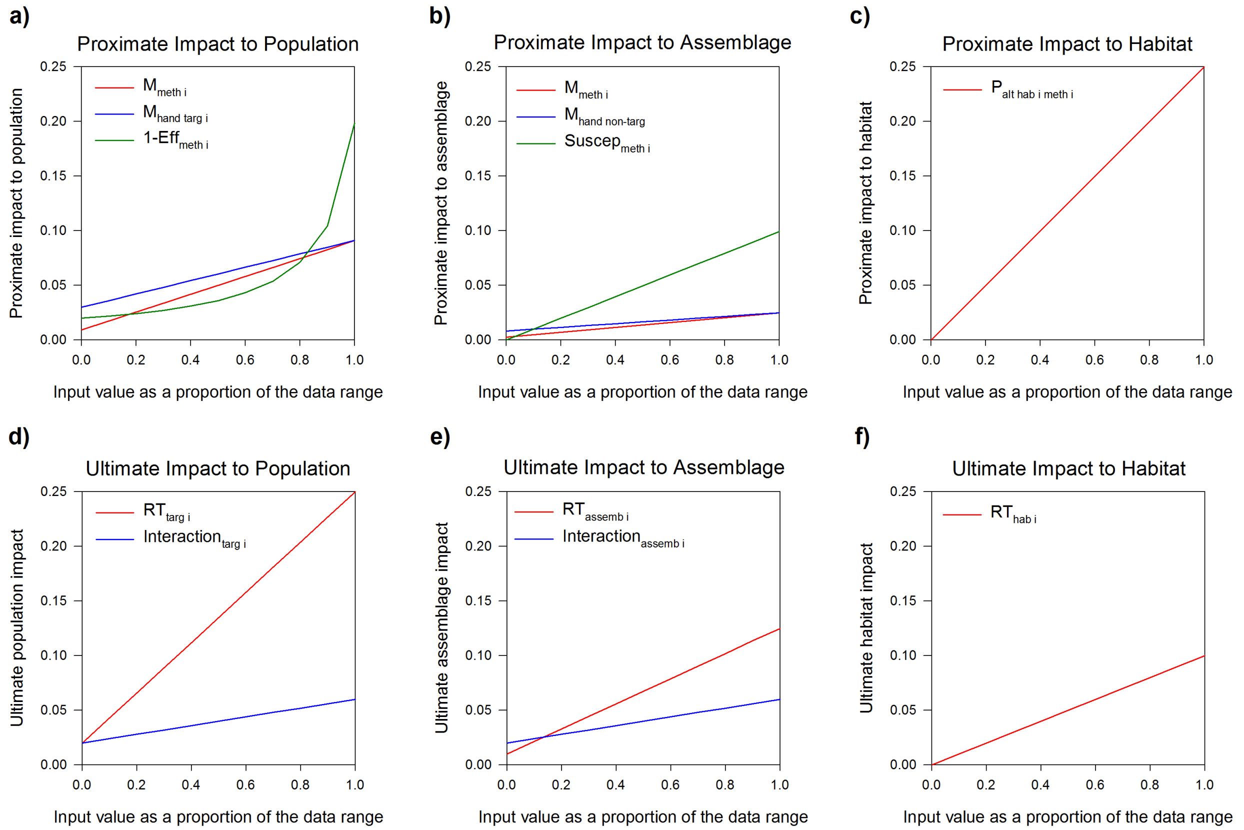

Figure 2. Relative sensitivity of estimated impacts to populations, assemblages, and habitats to variation in key input parameters. Sensitivity is expressed as the rate of change in estimated impact (vertical axis) caused by change in the parameter value (horizontal axis). Input values are standardized by the range of possible values, and plotted as a proportion of that range (horizontal axes), while all other inputs are held constant. To ensure that the impacts plotted are realistic, constants were set at the median of real world values and the proportion of the population, assemblage, or habitat targeted was set to 5% for the proximate impacts (top panels A, B, and C), and the proximate impact to the population, assemblage or habitat was set to 1% for calculation of the ultimate impacts (bottom panels (D, E, and F). **(A)** Relative sensitivity of estimated proximate population impact caused by variation in mortality associated with sampling method ($M_{meth\ i}$), handling effects ($M_{hand\ targ\ i}$), and effectiveness of the sampling method ($Eff_{meth\ i}$). **(B)** Sensitivity of estimated proximate assemblage impact caused by variation in mortality associated with sampling method ($M_{meth\ i}$), handling effects on non-targeted species ($M_{hand\ non\text{-}\ targ}$), and susceptibility of non-target species to the sampling method ($Suscep_{meth\ i}$). **(C)** Sensitivity of estimated proximate habitat impact associated with variation in sampling methods ($P_{alt\ hab\ i\ meth\ i}$). **(D)** Sensitivity of ultimate population impact to variation in population recovery time ($RT_{targ\ i}$) and species interaction index ($Interaction_{targ\ i}$). **(E)** Sensitivity of the ultimate assemblage impact to variation in assemblage recovery time ($RT_{assemb\ i}$) and species interaction indices within the assemblage ($Interaction_{assemb\ i}$), and **(F)** sensitivity of ultimate habitat impacts to variation in habitat recovery time ($RT_{hab\ i}$).

At the assemblage level (Figure 2b) , all three input parameters have linear relationships to the proximate impact value, but susceptibility ($Suscep_{meth\ i}$) has the steepest slope; thus, the proximate impact is most sensitive to method susceptibility meaning that obtaining accurate estimates of susceptibility to common sampling methods is paramount to making good estimates of assemblage level impacts. Similar to analyses of population level impacts, the ultimate impacts at the assemblage level are most sensitive to recovery time ($RT_{assemb\ i}$) (Figure 2e). Thus, recovery time (Figure 2d,e) played an important role in estimates of both population and assemblage impacts. Finally, both the proximate and ultimate impacts to habitat are influenced by a single parameter: the probability of habitat alteration resulting from the method ($P_{alt\ hab\ i\ meth\ i}$) in the proximate impact calculation (Figure 2c), and the recovery time of the habitat ($RT_{hab\ i}$) for ultimate impacts (Figure 2f).

Discussion

The decision-support framework presented here fills a need for an evidence-based permitting approval process for MPAs by providing a quantitative approach for estimating the ecological impacts of scientific activities. This approach offers advantages for both permit granters and applicants; Scientists proposing projects and managers permitting projects will benefit because the review process is transparent, unbiased, scientifically credible, and consistent across staff and over time. Because likely impacts of proposed projects can be readily identified, permitting decisions, particularly for low-impact projects, will likely be expedited. Since the framework quantifies the potential impacts of proposed studies, it provides information about where to make study design modifications to reduce project impacts. MPA managers will benefit because interactions between managers and permit applicants can be focused on those scientific activities of greatest concern. In addition, because managers will understand the anticipated impacts of proposed research projects during the permitting decision process, they will be able to better accommodate and prioritize studies with greater management or scientific value.

Granting permission to perform scientific research in protected areas has long been a management responsibility, because scientific collecting and other study procedures can impact protected species populations and ecosystems, and particularly rare taxa and habitats (Minteer et al. 2014, Henen 2016). However, assessing the potential impacts of scientific activities can be challenging, and as a consequence permitting decisions must often be based on qualitative information and judgments made by management officials who are unlikely to be intimately familiar with both the research methods (Finley 1988) and the taxa or ecosystems being studied (Henen 2016). Many scientists feel a responsibility to minimize the impacts of their research (Henen 2016), but this feeling is insufficient to address management and stakeholder concerns, not least because individual researchers are unlikely to consider the cumulative impacts of multiple research projects. During the establishment of California's MPAs, the potential for scientific research activities to impact biota was raised by fishermen and others restricted from extractive activities in MPAs (Gleason et al. 2010, Fox et al. 2013a, 2013b, Saarman and Carr 2013, Sayce et al. 2013). Since scientific studies are often the only explicitly extractive activities allowed in MPAs, assessing the impacts of those activities is especially important. Moreover, public support of scientific work depends on trusting scientists as well as their scientific integrity (Shamoo and Resnik 2015). Thus, before permits are issued, the objective and transparent understanding of the anticipated impacts of proposed scientific research activities are not only important for MPA managers but also for scientists seeking to maintain public support for their work while leaving a minimal footprint on the systems they study.

While our permitting decision-support framework provides an unbiased method for estimating the ecological impacts of a research project, the success of this approach depends on the quantity and quality of the data used to populate the models. For example, the framework requires abundance data for species and groups as well as habitat availability for each specific MPA where scientific work is to be undertaken; it also requires knowledge of species that play important ecological roles and that have long recovery times. The recent establishment of California's MPA network greatly expanded the availability of biotic and abiotic data throughout the state[7] (Young et al. In Press., MLPA Science Advisory Team 2009, 2010, Young and Carr 2015a, 2015b). Nevertheless, data describing species abundances are more likely to be available from MPAs that have previously supported considerable scientific work and less available for MPAs that have received little scientific attention. Our approach attempts to mitigate issues of data limitation and acknowledges uncertainty in several ways. First, we simplify the biotic effects of scientific sampling procedures by focusing only on extraction and mortality, the most impactful results of a research activity. Second, we conservatively apply parameter values by generalizing likelihoods of mortality to the assemblage level, using the high end of categorical ranges instead of precise numerical values for most parameters, and using conservative estimates of species abundance to populate our models. Third, although we use empirical data from the scientific literature when available, we often rely on expert judgment to make working estimates of model parameters including mortality rates, habitat impacts, species interactions, and recovery times (Appendices B-E). We expect that these estimates will be enhanced and sharpened with future input from the scientific community and as new knowledge becomes available.

We view the available data for species abundances and habitat availability generated during the California MLPA planning process (MLPA Science Advisory Team 2008, 2009, 2010) and by the baseline characterizations of ecosystems sponsored by the California Ocean Protection Council following establishment of the network[7] to be informative for many of California's MPAs, and believe our expert judgment approach provides a strong starting point for estimating the impacts of scientific research procedures. However, we recognize that more information is needed to improve model predictions. Hence, the accuracy of our models can be improved over

[7] oceanspaces.org/monitoring/regions

time as new data are generated from scientific studies performed within California's MPAs and elsewhere. In addition, this decision framework affords opportunities for scientists proposing studies to obtain the data necessary to populate the model. This is particularly important for MPA-specific data where in many cases the applicant will likely be highly knowledgeable about the species and system being studied and have access to the best available information. This presents both a challenge and an opportunity for the permitting agency. It places a burden on the permit granter to determine that the data provided by the applicant are both appropriate and the best available, a decision that might require consultation. However, it also provides an opportunity for managers during the application process to obtain and compile more and better data for future permit judgments, thereby generating the information needed to improve model accuracy over time.

Estimates of research impacts generated by this decision-support framework go beyond the proportion of a species or assemblage affected by a proposed study, which is captured with the estimated proximate impacts. The ultimate impacts, which are used for decision-making, additionally incorporate the ecological importance of a species or assemblage by evaluating its ecological role (the interaction index) and the duration of the impact by accounting for the recovery time of the affected species, assemblage or habitat. By incorporating these two factors into the estimates of ultimate impacts, we have generated a framework that can, with a single MPA-wide impact threshold, provide conservative protection for even sensitive species, assemblages, and habitats. We acknowledge that better understanding of the effects of species interactions and better predictions of the time required for functional recovery of ecological roles could improve the accuracy of our ecological impact predictions, but believe that the approach errs on the side of conservatism wherever possible. Additionally, because recovery time may exceed the lifetime of the permit itself, the framework retains the information from a permitted activity so it may be included in the cumulative impact assessment until the recovery time for that project has been exceeded.

Few studies have quantified the strength of interactions among species, especially those interactions that extend through a community (e.g. trophic cascades). Yet, because of the strong roles played by these species in organizing and structuring communities (Ellison et al. 2005), understanding the impacts of research activities on foundation species (*sensu* Dayton 1972) is particularly important as reflected in our sensitivity analyses. As more knowledge is accrued, the ability to quantify species interactions will improve and the values needed to populate our model will become more refined. This reinforces the importance of conducting studies in protected ecosystems where natural species interactions can more readily be quantified.

Recovery time in marine populations is also difficult to quantify and predict because, in addition to life history characteristics such as fecundity and age at maturity, recovery is dependent on larval and spore recruitment for most species, events that occur over different distances and temporal scales for different species types (Morgan 2000, Reed et al. 2000, Kinlan and Gaines 2003, Menge et al. 2003, Shanks et al. 2003, Shanks and Eckert 2005, Carr and Syms 2006). This means that the availability of adult source populations outside MPAs and not their abundances within MPA boundaries may be a limiting factor in population recovery for species whose propagules disperse long distances (Kinlan and Gaines 2003, Palumbi 2003, Shanks et al. 2003, Mace and Morgan 2006a, 2006b). Moreover, recruitment of many marine organisms is highly irregular and episodic (Caley et al. 1996, Eckert 2003, Wing et al. 2003, Carr and Syms 2006, Shanks and Roegner 2007), and ocean conditions among other parameters increase temporal uncertainty when it comes to recruitment cycles (e.g. Wing et al. 1995, Broitman et al. 2008, Morgan et al. 2009a, 2009b, Caselle et al. 2010, Ralston et al. 2013). Despite the stochastic nature of recruitment events that fuel population productivity for most species, recovery time plays a critical role in our models as evidenced by the sensitivity analyses which identify it as the most important parameter affecting the ultimate ecological impacts of scientific research activities. This underscores the need for more information that can improve predictions on the rate at which species and assemblages recover their ecological roles following the kinds of perturbations that might be associated with scientific activities.

Permitting a scientific research project to go forward in our approach relies not only on estimates of its individual ecological impacts and its contribution to the cumulative impacts of all other scientific projects, but also the impact level that can be sustained in an MPA without compromising its management and conservation goals. Setting an acceptable level for ecological impacts resulting from scientific research or any other forms of human activity is a policy decision. This task is especially challenging because unlike regulatory policies that set thresholds in other areas, for example water quality where studies have provided more direct evidence of links between problematic perturbations and biotic responses, it is much more daunting to determine impact levels below which the structure, functioning, and provision of ecosystem services are sustained in MPAs or for that matter most aquatic or terrestrial systems. The design of the impact framework, however, facilitates setting simple MPA-wide thresholds because the calculations of ultimate impacts already consider the most relevant factors (recovery time and ecological role) that could influence impact thresholds for individual species, assemblages, or habitats. Thus, a single policy-based impact threshold set for an MPA should apply and confer similar protections to any species, assemblage or habitat.

The acceptable level of impacts resulting from scientific research activities will vary from MPA to MPA based on their conservation goals and allowed activities. For example, in California some MPAs (State Marine Reserves – SMRs) prohibit any commercial or recreational take while others (State Marine Conservation Areas -SMCAs and State Marine Parks - SMPs) allow fishing and other human activities that can impact marine biota and physical habitat (Gleason et al. 2013, Saarman et al. 2013, CDFW 2016). Ultimately, effects of impactful activities besides scientific research will need to be assessed to ensure that MPA conservation and management goals can be met, particularly in MPAs similar to California's SMCAs and SMPs. Adding the ecological impacts of other extractive activities, which are measureable in the same currencies used to assess effects of scientific activities, can be accommodated in our approach if the required data are available. However, our model does not address effects of other stressors likely to be operative in an MPA such as water-borne pollutants or changing ocean conditions, including ocean acidification. As a result, acceptable impact levels must not only be set in the context of MPA goals and regulations, but also regularly re-assessed in consideration of the effects of other stressors.

Although our permitting framework can estimate and contrast the individual and cumulative ecological impacts of scientific activities in MPAs, it is designed to serve only as a guide, not as a prescription, for decision-making. Ultimately, the permit granting agency must decide not only on the impact levels that can be sustained by an MPA without compromising its goals, but also which research projects to allow when the cumulative impacts of scientific activities threaten to exceed acceptable thresholds. In MPAs subject to intense scientific activity, applications may need to be prioritized to derive the greatest management or scientific benefit from research without exceeding MPA impact thresholds. As a starting place, we suggest that no single project should consume more than one fifth of the acceptable impact threshold (e.g. if the impact threshold is set at 0.1 then the ultimate impact to any population, assemblage, or habitat should not exceed 0.02) without a clear justification of the benefits or value of the proposed scientific research. We hope this rule of thumb will ensure that no single project precludes other research in an MPA, except under extraordinary circumstances. Exactly how the benefits of scientific activities will be weighed against their ecological costs, is ultimately a management decision, but we think the greatest scientific benefit will be derived from those research projects that require MPAs to advance scientific understanding or that meet MPA management needs (e.g. monitoring programs that evaluate the status of MPA populations and communities). There also is a recognized need to continue established surveys and the collection of time series data to inform resource management in and outside of MPAs, such as fisheries surveys that inform stock assessments (Field et al. 2006, Mcgilliard et al. 2015), as well as studies required to meet mandates of governmental agencies, such as water quality monitoring required by the State, and also to afford appropriate, low-impact educational opportunities that help train the next generation of scientists and lead to greater public understanding of the value of protected ecosystems.

While our permitting decision-support framework is designed to address the approval process for scientific research within California's system of MPAs, the approach used is transferable to protected terrestrial and freshwater systems or other habitats where spatial or ecosystem-based approaches are used to manage extractive activities. This is because the framework is based on established ecological principles that apply across habitat types and requires only site-specific data and the ability to estimate the effects of study procedures. Although our decision-support framework is designed to facilitate the ability of protected area managers to evaluate the likely impacts of proposed scientific projects, it does not address all permitting problems for either managers or scientists proposing studies. For example, research involving certain species (e.g. endangered or otherwise specially protected species) can be much more complicated and involve multiple permitting agencies and, as pointed out by Paul and Sikes (2013), researchers must often navigate a maze of requirements and wait for months to obtain needed permits. In California, permission to perform scientific work in most MPAs falls under the regulatory authority of the Department of Fish and Wildlife. However species and ecosystems within MPAs can also fall under other regulatory authorities. For example the Point Reyes SMR located along the southern coast of Point Reyes overlaps with the Central California Coast Biosphere Reserve, the Gulf of Farallones National Marine Sanctuary, the Point Reyes National Seashore, the Point Reyes Headlands Extension Area of Special Biological Significance (ASBS), and the Point Reyes Headlands National Research Natural Area (McArdle 1997). Collectively, this area is managed by no less than two federal and two state agencies, each of which requires their own permitting processes. If permitting agencies converge on a common permitting decision-support framework, like the one generated here, the permitting process could be greatly improved and expedited. These issues require attention if collecting the scientific information needed to manage and conserve populations and ecosystems of all kinds is to be facilitated and appropriately regulated in MPAs and other protected environments.

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Appendix A: Case studies of ecological impact assessments using the decision framework

To test the decision framework, we frequently ran hypothetical example projects through the framework and examined the resulting values to see if they seemed reasonable. These hypothetical examples helped to refine the models and their parameterization and proved invaluable for understanding how the results may be useful for informing management decision making.

Below are the results of the ecological impact assessments for four hypothetical projects (Table A1). The similarities and differences between the four projects illustrate key elements of the models.

Table A1. Proximate and ultimate impacts calculated for each of four hypothetical projects. Proximate impacts are represented as a percentages, while ultimate impacts are unitless. Ultimate impacts are color coded where green font indicates impacts are less than 0.02 for the population, assemblage or habitat (i.e. recommendation to approve project), and orange is used for impacts between 0.02 and 0.05 (i.e. recommendation to revise project).

Project	Impact type	Impact on pop'n	Impact on assemblage				Impact on habitat
			Fishes	Mobile inverts	Sessile inverts	Macro-phytes	
1: Target 200 purple urchins using hand tools on 0-30m depth rock in Pt. Lobos SMR. Target urchins will be sacrificed for gonad analysis, any other organisms will be released.	proximate	0.216%	0.000%	0.001%	0.011%	0.011%	0.001%
	ultimate	**0.01298**	**0.0000**	**0.0002**	**0.0007**	**0.0010**	**0.0001**
2: Target 10 red urchins using hand tools on 0-30m depth rock in Pt. Lobos SMR. Target urchins will be sacrificed for gonad analysis, any other organisms will be released.	proximate	0.118%	0.000%	0.001%	0.006%	0.006%	0.001%
	ultimate	**0.03889**	**0.0000**	**0.0001**	**0.0004**	**0.0005**	**0.0001**
3: Target 80 lingcod using hook and line gear in 0-30m depth rock in Pt.	proximate	0.265%	0.190%	0.001%	0.007%	0.007%	0.007%

Project	Impact	Impact	Impact on assemblage				Impac
Lobos SMR. Target lingcod will be tagged and released and any other organisms will be released.	**ultimate**	**0.01061**	**0.0190**	**0.0001**	**0.0004**	**0.0006**	**0.0007**
4: Fifty 1 m²plots in the rocky intertidal will be cleared of all organisms using hand tools in Pt. Lobos SMR. Mobile organisms will be released.	proximat e	N/A	0.005%	0.025%	0.227%	0.227%	0.002%
	ultimate	**N/A**	**0.0002**	**0.0022**	**0.0205**	**0.0273**	**0.0002**

Projects 1 and 2 are identical projects performed on two different species of urchins. In the case of Project 1, removal and sacrifice of 200 purple urchins only results in a proximate impact on the urchin population of 0.2%, which when scaled to ultimate impacts by a 4-year recovery time and interaction index of 3, yields an ultimate impact to the population of ~0.013. In contrast, just 10 of the much less numerous red urchins constitutes 0.1% of the population, and when that is scaled to ultimate impacts by a 22-year recovery time and interaction index of 3, it yields an ultimate impact to the population of ~0.039. This comparison illustrates the importance of abundance and recovery time in determining impact levels. Both projects 1 and 2 have low levels of incidental impact on assemblages and habitats due to the use of hand tools that result in little incidental take or habitat damage.

Project 3 illustrates how higher levels of susceptibility to a study method distribute the impacts of the study through susceptible assemblages. In project 3, 80 lingcod are proposed to be taken by hook and line. The fish assemblage is considered to be moderately susceptible to hook and line gear, thus many other fish within the assemblage are likely to be impacted by the study method. With respect to handling mortality, differences between handling of the targeted lingcod (tag and release), and non-targeted fishes (catch and release) is small. As a result of these two factors, the proximate impacts to both the target species and the fish assemblage as a whole are quite similar (0.27% vs. 0.19%) and remain quite similar when scaled to ultimate impacts by the recovery time and interaction index. Impacts to less susceptible assemblages and physical habitat remain low.

Project 4 illustrates an example of a community-wide study in which no target is declared. In this project, impacts are not calculated at the population level, but are instead reflected at the assemblage and habitat level. In this example, the interplay of susceptibility and mortality determine where the maximum impacts are observed. For the study method, which is clearing with hand tools, sessile invertebrates and macrophytes are most susceptible and also likely to sustain the highest mortality when removed from the substrate. These factors translate to the highest proximate impacts on these two groups (~0.2% on each). This pattern holds when proximate impacts are scaled to ultimate impacts by recovery time and interaction index.

Appendix B: Overview of model parameterization methods

The decision framework that we developed for evaluating proposed research activities in MPAs requires substantial information to provide useful results. Specifically, we need inputs for each of the 14 parameters used in the ecological impact equations, only two of which are likely to be provided by the applicant (number of target organisms or area sampled) and two of which could be gleaned from location-specific empirical data (density of target species and habitat availability). To meet these informational needs, the group developed a series of data tables with model parameters for a wide variety of potential research activities. The parameter tables are not exhaustive, but they provide values that are applicable to most common study methods, species, assemblages, and habitats.

The 10 parameters for which we developed tables differed in the availability of peer reviewed literature and empirical data, thus we employed different approaches to populate the data tables. Our approaches fell into three main categories, 1) literature search and compilation, 2) expert judgement, and 3) expert judgement or literature search with the aid of a decision guide. All approaches involved frequent internal review and consensus-building within the workgroup, as well as testing of resulting parameter values by running examples through the ecological impact equations. Because of the importance of maintaining MPA protection, we maintained a precautionary philosophy throughout development of the parameter tables.

Literature review: With all parameters, we first reviewed the literature to determine if parameter values could be gleaned from published science. While the primary literature is a preferred information source, there were few parameters for which we could find estimates, and those we *could* find were often more specific than needed (e.g. mortality estimate associated with hook and line gear for an individual fish species when we needed a more general estimate of mortality for an entire fish assemblage). Due to these limitations, we found that values from the literature served primarily as anchors for the parameter values, but the majority of values were derived through the expert judgement and guided decision-making approaches described below.

Expert judgement approaches: When we employed expert judgement approaches, we initially assigned parameter values as qualitative categories (e.g. low, high, etc.). This enabled the experts to avoid getting caught up in the details and instead to focus on broad similarities and differences between the parameters they were evaluating. Those qualitative categories were later translated into ranges of values, and the conservative ends of the ranges were used as the input parameters in the equations. The step of translation from qualitative to quantitative values proved to be an iterative process requiring workgroup consensus and repeated example runs to ensure that the impact model outputs seemed reasonable. This process was also informed by the sensitivity analyses in some cases.

How we deployed the expert judgement approach varied based on the complexity of the parameter tables and number of values that needed to be generated. For some of the simpler tables, all members of the work group provided qualitative categorical values for the whole table. These survey-style responses were then compiled and the median value was used in the parameter table. Where there was a great deal of variation in the responses by different work group members, the group reconvened to discuss the source of the differences and develop a better consensus. Differences were frequently attributable to differing conceptual approaches to the question and quickly resolved through discussion.

For some of the more complex parameter tables, we assigned each workgroup member a portion of the table to fill out, based on their experience and expertise. We then reviewed the completed parameter table as a group, ensuring that all workgroup members used similar criteria for assigning categorical values and that values were comparable and yielded reasonable results when applied to the impact equations.

Guided decision-making: Some parameters required weighing of multiple factors or use of a variety of different information sources from the literature, depending on what information was available. In these cases, we laid out the conceptual elements of parameter assignment as a decision guide and then followed the steps in that guide to derive the final values. For example, recovery time estimates can come from a variety of different sources depending on the type of organism and the availability of life history information. We formalized how each potential information source should be used and how to apply expert judgement if no information could be found in the literature in a decision guide, which is described in greater detail in Appendix E. Similarly, we developed a decision guide to aid in identification of strong ecological interactors, which uses expert judgement to identify the different types of interactions associated with each candidate species and then helps to compile that information to derive an overall interaction index.

Specifics on the approach used to develop each parameter table as well as example values can be found in appendices C-E. Appendix C describes the five parameters related to the impacts of study methods on organisms and habitats: method-related mortality (M_{meth}), handling-related mortality (M_{hand}), method efficacy (Eff_{meth}), susceptibility to the study method ($Sucep_{meth}$), and the probability of habitat alteration resulting from the study method ($P_{alt\ hab\ meth}$). Due to a paucity of literature on the impacts of study methods, these five tables were developed using an expert judgement approach. Appendix D describes the development of interaction indices ($Interaction_{targ}$ and $Interaction_{assemb}$) to characterize ecological interactions for species and assemblages. These interaction values were developed using a combination of decision guide and expert judgment. Finally, Appendix E describes recovery times for species, assemblages, and habitats (RT_{hab}, RT_{assemb}, and RT_{hab}). These values were developed using a decision guide, literature search, and expert judgment in combination.

Appendix C: Estimating the impacts of study methods on organisms and habitats

The ecological impact models contain five parameters related to the impacts of study methods on organisms and habitats: method-related mortality (M_{meth}), handling-related mortality (M_{hand}), method efficacy (Eff_{meth}), susceptibility to the study method ($Sucep_{meth}$), and the probability of habitat alteration ($P_{alt\ hab\ meth}$) which are highlighted where they appear in the proximate impact equations below in yellow, green, blue, pink, and gray respectively.

$$PI_{targ\ i} = \left\{ \{M_{meth\ i} + [(1 - M_{meth\ i}) \times M_{hand\ targ\ i}]\} \times \left(\frac{1}{Eff_{meth\ i}}\right) \right\} \times \frac{N_{targ\ i}}{Dens\ or\ \%\ cover_{targ\ i} \times A_{MPA\ hab\ i}}$$

$$PI_{assemb\ i} = \{M_{meth\ i} + [(1 - M_{meth\ i}) \times M_{hand\ non-targ}]\} \times (Suscep_{meth\ i}) \times \frac{A_{samp\ hab\ i}}{A_{MPA\ hab\ i}}$$

$$PI_{hab\ i} = P_{alt\ hab\ i\ meth\ i} \times \frac{A_{samp\ hab\ i}}{A_{MPA\ hab\ i}}$$

For these five parameters, the availability of literature or empirical data was severely limited, thus we defaulted to expert judgment approaches to fill out the tables. For those few methods that closely mirror common commercial fishing techniques, we were able to find some literature on mortality and incidental take (related to susceptibility), and this information was used to anchor expert-derived estimates.

For these parameters, we started with commonly used scientific techniques and created categories of study and handling methods (e.g. hand nets) that were designed to encompass multiple methods with similar impacts (e.g. A-frame nets, dip nets), but also to differentiate between apparently similar methods where the types of organisms impacted or the magnitude of the impacts were likely to be substantially different (e.g. fish vs. invertebrate traps or midwater vs. bottom trawl over different substrates). The resulting categories should be applicable to most future proposed projects.

Similarly, generating precise mortality or susceptibility estimates for individual species was not feasible using the expert judgment approach, so we grouped organisms into categories that closely mirrored the assemblages used throughout the models: macrophytes, sessile invertebrates, mobile invertebrates, and fishes, with an additional division of fishes into those from deeper and shallower water to account for barotrauma effects for some parameters. This division of the fish assemblage by depth was due to the special consideration of barotrauma in fish with swim bladders captured at depths greater than 50m, where the likelihood of mortal barotrauma is higher than in shallower waters. Thus, for each study method, mortality estimates were assigned to each of the following five assemblage groups: macrophytes, sessile invertebrates, mobile invertebrates, fish <50m depth, and fish >50m depth.

All of this categorization greatly reduced the number of values necessary to comprehensively describe the impacts of the most commonly used study methods. However, categorization also made it impossible to precisely estimate parameter values because, for example, different fish species are likely to sustain slightly different mortality rates from the same gear due to inherent characteristics of the species. To account for this uncertainty, the workgroup members (experts) conservatively assigned qualitative categories of mortality (e.g. minimal, very low, low, etc.) to each method-group combination and these categories were subsequently translated to ranges of values. In keeping with our conservative approach, the most conservative end of each range was used as the actual parameter input for the purposes of calculation. The number of categories and translation to values differed for each parameter because of the unique characteristics of each parameter and the sensitivity of the models. These categorical divisions are not an essential element of the models, but represent a balance between the simplicity of assigning categories instead of precise value and the perceived nuances of the differences between study methods.

Estimates of mortality directly resulting from the study method (M_{meth})

Study method mortality (M_{meth}) is defined as the proportionate mortality of organisms subjected to the study method. The study method is defined as the method used to capture or observe organisms and applies to all organisms, whether targeted or incidentally captured. For example, in a hypothetical fish tagging study that uses hook and line gear to capture the fish, all captured organisms would be subject to mortality from the hook and line study method, whereas only fish of the target species would be subjected to the handling mortality associated with tagging (M_{hand} described below). All proposed projects will use some kind of study method, including observational studies that don't require capture or contact with the organisms—in these cases the study method is "visual observation" with zero or near-zero probability of mortality. Note that the method mortality estimates represent the mortality expected for individuals actually captured or sampled by the method, not the probability of capture. For example, beach seines may not be likely to capture macrophytes, but any macrophytes collected by beach seines are highly likely to perish. The likelihood of capture is represented separately as susceptibility ($Suscep_{meth}$) and described in the following section.

To estimate the probability of mortality associated with a proposed sampling method (M_{meth}, highlighted in yellow in the equations above), the working group applied the expert judgement approach described in Appendix B. As this parameter table proved to be one of the more lengthy tables, different experts worked on different sections of the table to assign mortality categories to each method-assemblage combination (i.e. hand nets for mobile invertebrates). Six categories of mortality were applied (Table 1), ranging from zero to one (e.g., intended sacrifice of the organism). Reflecting a precautionary application of these uncertain estimates, the highest value within a category was used as the parameter value in the equations.

Table C1. Qualitative categories used to assess mortality associated with study methods and the quantitative ranges and parameter values associated with each category.

Mortality Category	Probability of Mortality	Paramete r Value	Definitions and examples
Very High	0.67< - 1.00	1	Includes highly impactful methods (e.g. trawl gear on sessile invertebrates) and intentionally lethal methods (e.g. spear-fishing).
High	0.33< - 0.67	0.67	Includes impactful methods (e.g. trawl gear on mobile species) and vulnerable groups (e.g. fish from >50m depth)
Moderate	0.10< - 0.33	0.33	This category was applied rarely, mostly to vulnerable groups taken by methods that are considered low impact (e.g. hook and line for fish from >50m depth), or to resilient groups taken by methods considered to be high impact (e.g. some trawl gear on fish from <50m depth)
Low	0.01< - 0.10	0.10	Includes study methods that are generally considered to be low impact, but are known to have mortality (e.g. hook and line in shallow water).
Very Low	0< - 0.01	0.01	Includes methods that have very low, but arguably non-zero impacts.
Zero	0	0	This category was only used for methods with likelihoods of mortality so low that the group felt comfortable calling them zero.
Not applicable	N/A	0	This category was used where the study method could not realistically impact the group in question (e.g. hand nets for fish from depths >50m).

Table C2. Examples of common sampling methods and their estimated probability of mortality for the five assemblage groups using the scoring categories described in Table C1.

Method	Probability of mortality by assemblage					Scoring notes
	Fish <50 meters	Fish >50 meters	Mobile Inverts	Sessile Inverts	Macrophytes	
Beach seine	0.10	N/A	0.01	1	1	Sessile inverts and macrophytes are assumed to perish if removed from their (potentially rocky) substrate.
Cast net	0.33	N/A	0.10	1	1	High mortality for fishes reflects vulnerable anchovies and silversides
Hand net	0.10	N/A	0.01	1	1	Sessile inverts and macrophytes are assumed to perish if removed from their (potentially rocky) substrate.
Hand tools	0.01	N/A	0.10	1	1	Sessile inverts and macrophytes are assumed to perish if removed from their (potentially rocky) substrate.
Hook and line from the surface	0.10	0.33	0.10	1	1	Sessile inverts and macrophytes are assumed to perish if removed from their (potentially rocky) substrate.
Trawl, soft bottom, small mesh	0.10	0.66	0.10	0.10	1	Relatively low mortality for inverts and shallow fishes reflects the slow speeds and short tows characteristic of small mesh research trawls. Sessile inverts, such as clams, are likely to survive from soft bottom as they are not attached.
Visual observation	0	0	0	0.01	0.01	Non-zero mortality for macrophytes and sessile inverts represents potential trampling effects

Estimates of community susceptibility to sampling methods ($Suscep_{meth}$)

Susceptibility is the parameter that accounts for the likelihood that an organisms will be captured with a particular study method. $Suscep_{meth}$ is defined as the proportion of an assemblage within the sampling area that is susceptible to inadvertent take by a particular study method, and it determines how sampling and handling mortality should be applied to non-target organisms in the community. For example, a susceptibility value of 0.25 for the fish assemblage indicates that 25% of fish are vulnerable to incidental capture by the study method, thus the mortality associated with the sampling method (M_{meth}) is applied to 25% of the fish assemblage in the sampling area. If an assemblage has low susceptibility to a particular method, that means one of two things: either it is possible to be very selective with the method and capture only targeted organisms, or the method is ill-suited to capturing organisms in that assemblage.

To estimate susceptibility ($Suscep_{meth}$, highlighted in pink in the equation above) the working group applied the expert judgement approach described in Appendix B. As this parameter table proved to be one of the more lengthy tables, different experts worked on different sections of the table to assigned susceptibility categories to each method-assemblage combination (i.e. hand nets for mobile invertebrates). Eight categories of susceptibility were used (Table C3) with value ranges ranging from zero to one. As with other parameter values, we used the conservative end of each range as the actual parameter input. Examples of these estimates of susceptibility to common sampling methods for each four assemblages are presented in Table C4.

Table C3. Qualitative categories used to assess the susceptibility of assemblages to particular study methods and the quantitative ranges and parameter values associated with each category.

Susceptibility Category	Proportion of Assemblage Susceptible	Parameter Value	Definitions and examples
Very High	0.75< - 1.00	1	A relatively indiscriminate method that is principally used to sample an entire community (e.g. trawling), and would likely not be appropriate to use for targeting a particular species unless mortality or is very low or sampling area very small.
High	0.5< - 0.75	0.75	Most of the assemblage is susceptible to this method and substantial incidental take is likely in targeted studies.
Moderate	0.25< - 0.5	0.50	Up to half of the assemblage is susceptible to incidental take by this method (e.g. beach seine for fishes), thus it may not be a good choice for targeted studies.
Moderate-low	0.1< - 0.25	0.25	The assemblage is only moderately susceptible to incidental take by this method, or it is possible to target a species within this assemblage with moderate accuracy (e.g. hook and line for fishes).
Low	0.01< - 0.1	0.1	The assemblage is not very susceptible to incidental take by this method, or it is possible to accurately target species in this assemblage.
Very low	0.001< - 0.01	0.01	The assemblage is not at all susceptible to incidental take by this method, or this method can very precisely target a species with little impact on the rest of the assemblage.
Minimal	0< - 0.001	0.001	The chances of incidental take of this assemblage are minimal.
Zero	0	0	This assemblage cannot be taken incidentally by this method.

Table C4. Examples of common sampling methods and estimates of the susceptibility of species in four assemblage groups using the scoring categories described in Table C3.

Method	Probability of mortality by assemblage				Scoring notes
	Fish	Mobile Inverts	Sessile Inverts	Macrophytes	
Beach seine	0.50	0.10	0.01	1.00	Many fishes and most invertebrates readily evade a beach seine. Especially in estuaries, macrophytes can be inadvertently removed by this method.
Cast net	0.25	0.01	0.01	0.01	There is some ability to target fishes, but incidental take is likely. Other assemblages are not very susceptible.
Hand net	0.01	0.01	0.001	0.001	Can be used to precisely target fish and mobile invertebrates with little incidental take. Sessile invertebrates and macrophytes would only be taken via trampling effects.
Hand tools, sessile organisms on rocky substrate	0.01	0.25	0.25	0.25	Sessile organisms often form habitat for other organisms that are likely to be incidentally removed or damaged with hand tools.
Hand tools, mobile organisms on rocky substrate	0.001	0.01	0.01	0.01	Mobile organisms can usually be removed using hand tools with little incidental damage to other organisms, regardless of assemblage.
Hand tools, mobile organisms on rocky substrate	0.001	0.01	0.01	0.01	Mobile organisms can usually be removed using hand tools with little incidental damage to other organisms, regardless of assemblage.
Hook and line from the surface	0.25	0.001	0.001	0.001	Some ability to select a target within the fish assemblage. Other assemblages are not very susceptible.
Trawl, soft bottom, small mesh	0.75	0.75	0.75	1	All assemblages are quite susceptible, may not be appropriate for targeted studies.
Visual observation	1	1	1	1	All organisms are susceptible, but mortality is zero or near-zero

Estimates of mortality from handling (M_{hand})

Handling is defined as anything that organisms are subjected to subsequent to capture. In the case of observational studies where organisms are never captured, handling is visual with no contact and handling mortality is equal to zero. For all study methods that include capture, there is some subsequent handling of organisms, and this handling often differs for targeted versus non-target organisms. For example, in a hypothetical fish tagging study using hook and line gear as the study method, all captured organisms are subject to mortality from the study method (hook and line). However, only fish of the target species would be subjected to the handling mortality associated with tagging, while fish of other species would only be subject to the handling mortality associated with catch and release.

To estimate the probability of mortality associated with a proposed handling method (M_{hand} highlighted in green in the equations above), the working group applied the expert judgement approach described in Appendix B. The list of handling methods for which values were assigned is relatively short because we categorized handling methods into broad categories. Researchers are likely to have carefully considered, and perhaps even studied, the mortality associated with their handling methods, and thus can help refine this table in cases where our broad handling categories with conservatively mortality assessments don't fit.

Seven categories of handling mortality were applied (Table C5), ranging from zero to one probability of mortality (e.g., intended sacrifice of the organism). Reflecting a precautionary application of these uncertain estimates, the highest value within a category was used in the equation.

Table C5. Qualitative categories used to assess mortality associated with handling methods and the quantitative ranges and parameter values associated with each category.

Mortality category	Probability of mortality	Parameter value	Definitions and examples
Very High	0.67< - 1.00	1	Includes highly impactful and intentionally lethal methods (e.g. sacrifice).
High	0.33< - 0.67	0.67	Includes handling methods with a high likelihood of mortality
Moderate	0.10< - 0.33	0.33	Includes handling methods with a moderate likelihood of mortality (e.g. stomach lavage, delayed release)
Low	0.01< - 0.10	0.10	Includes handling methods that are generally considered to be low impact, but are known to have mortality (e.g. tag and release).
Very Low	0< - 0.01	0.01	Includes handling methods that have very low, but arguably non-zero impacts (e.g. catch and release).
Zero	0	0	This category was only used for methods with likelihoods of mortality so low that the group felt comfortable calling them zero.
Not applicable	N/A	0	Handling method does not apply to the assemblage.

Table C6. Examples of common handling techniques and their estimated probabilities of mortality for the four assemblages using the scoring categories described in Table C5.

Method	Probability of mortality by assemblage				Scoring notes
	Fish	Mobile Inverts	Sessile Inverts	Macrophytes	
Catch and Release	0.01	0.01	0.01	0.01	With catch and release, most of the mortality is due to the method of capture, not the act of release.
Catch and release, mass	0.33	0.33	0.33	0.33	This method applies to situations where release may be delayed due to large numbers of organisms caught at once (i.e. with nets)
Gamete Harvesting	0.33	0.10	N/A	N/A	This is non-lethal gamete harvesting.
Sacrifice	1	1	1	1	Refers to intended sacrifice of organisms for a variety of purposes
Stomach Lavage	0.10	0.10	N/A	N/A	Causing an organism to regurgitate its stomach contents for diet analysis, although not lethal, may cause mortality of fishes and mobile invertebrates
Visual observation	0	0	0	0	This handling method is used in cases where there is no explicit handling because there is no capture.

Estimates of study method efficacy (Eff_{meth})

The efficacy term (Eff_{meth}) is defined as the proportionate success of the study method in collecting the proposed number of individuals (i.e. number of useable samples divided by the total number of organisms sampled). This term is designed to account for situations in which study and handling methods may be applied to an organism (along with their attendant probabilities of mortality) without yielding a sample that is useable for the study. Because there are strong financial incentives for study methods to be both efficient and effective, we anticipate that most proposed studies will have an efficacy of one (i.e. every organism sampled will yield a useful sample). However, we can readily envision several scenarios in which efficacy would be less than one. For example, if a study required samples from 10 males of a fish species, but sex was impossible to determine without lethal dissection, efficacy would be reduced to 0.5 as likely only half of the fish sampled would end up being males and thus contribute useable samples.

To estimate the efficacy parameter (Eff_{meth}, highlighted in blue in the equation above) we envisioned a short list of potential sampling scenarios in which only a portion of the targeted organisms would yield useful samples, and used an expert judgement to assign efficacy to those scenarios. With this parameter table in particular, we anticipate that specific study proposals will be examined on a case by case basis and efficacy values determined based on the specifics of the study. This seems feasible because efficacy can be determined by answering a simple question: "What proportion of the organisms sampled (i.e. subject to both study method and target handling) are likely to yield useful samples (i.e. count toward the number of organisms targeted for the study)?"

Table C7. Examples of efficacy parameters for hypothetical sampling scenarios

Sampling scenario	Efficacy	Notes
All individuals can contribute to the study	1.00	The default scenario. The vast majority of studies should fit this category
Only one sex can contribute to the study but sex cannot be determined non-lethally	0.50	Roughly half of sampled individuals would be unusable for the study
Only individuals over a particular age can contribute to the study, but cannot determine age non-lethally	depends	This would depend on the age distribution of the population and the minimum age required.

Estimates of the probability of habitat alteration resulting from the study method ($P_{alt\ hab\ meth}$)

In addition to impacting organisms through mortality, study methods may intentionally or unintentionally impact the physical or chemical characteristics of the habitat on which organisms depend. For the sake of simplicity we defined the probability of habitat alteration as applying only to the physical structure of the habitat because these are the most common effects likely to result from the study methods considered. We also defined habitat as the abiotic habitat to avoid double-counting impacts to organisms that form biogenic habitat (e.g. kelps, seagrasses, corals, sponges, etc.). We did not distinguish different degrees of habitat impact except as lower or higher probabilities of habitat alteration. The different probabilities of habitat alteration can also be thought of as the proportion of the habitat in the sampling area that is likely to be altered. The duration of these impacts are addressed in the recovery time portion of the models (see Appendix E). The probability of habitat alteration assigned to a particular method is intended to be applied across the entire sampling area.

To estimate the probability that a sampling method will alter a particular physical habitat ($P_{alt\ hab\ meth}$, highlighted in gray in the equations above) the working group used a survey-style expert judgement approach (Appendix B). We used six categories of habitat impact that equate to parameter values ranging from zero to one (Table C8) and applied these categories to the study method categories used throughout the models. We differentiated probabilities of habitat impact for rock and sediment substrates, but did not distinguish habitats more finely (Table C9). Reflecting a precautionary application of these uncertain estimates, the highest value within a category was used as the parameter value.

Table C8. Qualitative categories used to assess the probability of habitat alteration and the quantitative range and parameter values associated with each category.

Habitat alteration category	Probability of habitat alteration	Parameter Value	Definitions and examples
Very High	0.67< - 1.00	1.0	Method is very likely or certain to alter the habitat in the study area (e.g. permanent anchors, experimental structures)
High	0.33< - 0.67	0.67	Method is likely to alter most of the habitat in the study area (e.g. most trawl methods)
Moderate	0.10< - 0.33	0.33	Method is moderately likely to alter the habitat in the study area
Low	0.01< - 0.10	0.10	Method is likely to modify no more than 10% of the habitat in the study area (e.g. beach seine)
Very low	0.001< - 0.01	0.01	Method has a very low probability of habitat alteration (e.g. clearing rocky substrate with hand tools)
Minimal	0< - 0.001	0.001	The chances of habitat alteration are minimal with this method, but not zero (e.g. hook and line methods)
Zero	0	0	The method would not alter the habitat.
Not applicable	N/A	0	Method would not be used in the habitat.

Table C9. Examples of common sampling methods and their estimated probability of habitat alteration using the scoring categories described in Table C8.

Methods	Probability of habitat alteration by substrate type		Scoring notes
	Rock	Sediment	
Beach seine	NA	0.10	Should not be used on rock substrate, contact with sediment will cause some alteration
Cast net	0.01	0.01	May cause some habitat alteration through contact with the substrate and entanglement
Experimental structure	1	1	Designed to alter habitat and will certainly do so within its footprint (i.e. the study area)
Hand net	0.001	0.001	May cause some very minor habitat alteration through contact with the substrate and entanglement
Hand tools, sessile organisms on rocky substrate	0.01	N/A	May alter substrate through scraping/chipping, but likely only a very small fraction of the study area
Hook and line	0.001	0.001	May cause some very minor habitat alteration through contact with the substrate and entanglement
Trawl, soft bottom, small mesh	NA	0.67	Is likely to contact and thus alter most of the sediment habitat within its path.
Visual observation	0	0	Very unlikely to alter physical habitat.

Appendix D: Estimating the strength of ecological interactions

The ultimate impact portions of the ecological impact models contain two parameters related to the strength of ecological interactions between species ($\text{Interaction}_{\text{targ}}$) and assemblages ($\text{Interaction}_{\text{assemb}}$) highlighted in yellow and green respectively in the ultimate impact equations below.

$$UI_{targ\ i} = PI_{targ\ i} \times \frac{RT_{targ\ i}}{2} \times Interaction_{targ\ i}$$

$$UI_{assemb\ i} = PI_{assemb\ i} \times \frac{RT_{assemb\ i}}{2} \times Interaction_{assemb\ i}$$

A primary goal of most protected areas is to protect not just individual species but the structure and function of entire ecosystems. Because each species plays a distinct ecological role, it is important to consider all the species potentially affected when estimating the ecological impacts of proposed scientific activities, and particularly those known to strongly affect community structure through their interactions with other species. Some species are strong interactors whose interactions (predation, competition, facilitation) result in cascading effects that ramify throughout much of the ecosystem. Our goal in estimating the two interaction index parameters was to identify those species with especially strong interactions and ensure those interactions were considered in assessing ultimate impacts.

To estimate the relative strength of interaction among species ($\text{Interaction}_{\text{targ}}$), we used a guided expert judgement approach (see Appendix B) to characterize the types and strengths of interactions for a suite of potential strong interactor candidates. Potential strong interactors were identified for each habitat by experts familiar with the habitat, and then each candidate was assigned a qualitative interaction strength (ranging from zero to four) for each of seven interaction types shown in Table D1 and defined in greater detail in Table 3 in the main body of the manuscript. These qualitative scores were then summed across all interaction types and translated to an interaction index. Summed scores could potentially vary from zero to 24 (a score of four in six of the seven categories—allogenic and autogenic engineers are mutually exclusive). Summed interaction scores were then translated to an interaction index scale from one to three, such that total scores from zero to three, four to 7, and greater than 7 were scaled as whole integers from one to three, respectively (Table D1). The group elected to scale the final interaction index from one, for a species with ecological interactions proportionate to its abundance, to three for a species with strong ecological interactions disproportionate to abundance such that a small change in population could have ramifications throughout the ecosystem.

To estimate the interaction index for an assemblage ($Interaction_{assemb}$) we applied the precautionary principle and used the highest interaction index for any species in the assemblage. In some cases, it may be obvious that the strongest interactor in an assemblage is not susceptible to the proposed study method, and in those cases it is appropriate to use the interaction strength from the strongest interactor that's susceptible to the study method. Because the list of strong interactors within each assemblage-habitat combination is small (typically less than 10), determining if any are likely to be susceptible to a specific method is feasible on a case by case basis.

Table D2. List of some potential strong interactors with interaction index scoring for shallow rocky reef habitat.

Species or group	Keystone species	Ecosystem engineer - allogenic	Ecosystem engineer - autogenic	Structural - Biogenic habitat	Facilitative interactions (not biogenic hab.)	Dominant (competitive and abundant)	Trophic importance (food-chain support)	Relative interaction strength	Interaction index
Giant kelp	-	-	4	4	4	-	4	16	**3**
Southern sea otter[1]	4	-	-	-	-	-	4	8	**3**
Bull kelp	-	-	4	3	4	-	4	15	**3**
Encrusting coralline algae	-	-	-	1	3	-	1	5	**2**
Erect coralline algae	-	-	-	2	-	-	1	3	**1**
Red urchins	-	4	-	2	3	-	2	11	**3**
Purple urchins	-	4	-	2	3	-	2	11	**3**
Lobster	2	-	-	-	-	-	4	6	**2**
Sheephead	3	-	-	-	-	-	4	7	**2**
Lingcod	-	-	-	-	-	-	4	4	**2**
Large barnacles (Balanus nubilus)	-	-	-	3	3	-	1	7	**2**

[1] Southern sea otter is in this table for comparative purposes only. Otters are federally protected and studies that impact them would not be determined using this framework.

Appendix E: Estimating recovery times for populations, assemblages, and habitats

The ultimate impact portions of the ecological impact models each contain a parameter that reflects recovery time: recovery time for populations (RT_{targ}), assemblages (RT_{assemb}), and habitats (RT_{hab}) highlighted in yellow, green, and blue respectively in the equations below.

$$UI_{targ\ i} = PI_{targ\ i} \times \frac{RT_{targ\ i}}{2} \times Interaction_{targ\ i}$$

$$UI_{assemb\ i} = PI_{assemb\ i} \times \frac{RT_{assemb\ i}}{2} \times Interaction_{assemb\ i}$$

$$UI_{hab\ i} = PI_{hab\ i} \times \frac{RT_{hab\ i}}{2}$$

The duration of impacts from scientific activities will vary greatly depending on the rate at which affected species and assemblages are able to recover their abundances and ecological roles and the rate at which habitats return to their unaltered state. For example, impacts on long-lived species or those with low reproductive rates or infrequent larval recruitment events are likely to have long-lasting ecological effects compared with impacts on short-lived species with high reproductive rates and frequent larval recruitment events. Similarly, impacts on more static habitats, such as rocky reefs, are likely to have long-lasting effects compared to impacts on more dynamic sedimentary habitats. Because recovery of affected populations and habitats is likely to be incremental, we incorporate recovery time into the model by multiplying the proportionate impact by one half of the recovery time (*RT*/2) for all three ultimate impact equations. This approach assumes a linear recovery from the time of the impact to the end of the recovery time.

Our working definition of recovery time for populations and assemblages was replacement of the abundance (density or percent cover) and size-structure of individuals removed, to reflect the lost density- and size-dependent functional roles of impacted species. We did not consider recovery at the local scale through immigration of older life-stages or vegetative encroachment because this type of recovery still represents a net reduction of the population or assemblage in the MPA. Instead, we chose to estimate recovery time as a function of the life history characteristics of the organisms. Specifically, we defined recovery time as the inverse of natural mortality because natural mortality should reflect the variety of life history characteristics that influence the recovery of populations: fecundity, age at maturity, life span, episodic recruitment, etc. However, natural mortality estimates are generally only available for those species with stock assessments, so we had to rely on other life history characteristics for some species, and expert judgement for many others. For habitat recovery, we had to rely almost entirely on expert judgement with limited information from the literature to help ground our estimates.

To estimate the three recovery time parameters, we used a combination of decision guide, literature search, and expert judgement (see Appendix B for descriptions) and varied the approach based on the assemblage in question and the availability of information from the literature. Details of how we estimated each recovery parameter along with examples are below. Recovery time estimates are in units of years. Because recovery time is incorporated into the ultimate impact equations as RT/2, we defined the minimum recovery time for a species, assemblage, or habitat as two years to ensure the entire recovery time term would never be less than one. The longest species recovery time we estimated was 26 years for yelloweye rockfish (calculated from a maximum age of 118 years), thus we capped the maximum recovery time at 25 years for species, assemblages, and habitats. To avoid the illusion of excessive precision, we rounded all calculated recovery times up to the nearest year.

Estimates of recovery for populations (RT_{targ})

As described above, defining recovery time as the inverse of natural mortality, is a conceptually simple and attractive solution, however, estimates of natural mortality are typically only available in stock assessments and for managed species. To expand the usefulness of this conceptual approach, we used an equation from Hoenig (1983) that estimates natural mortality from a much more readily available parameter—maximum age. This equation takes the form $\ln Z = 1.44 - 0.982 \ln t_{max}$ where Z equals total mortality, which can be used as a proxy for natural mortality in the absence of fishing and t_{max} equals maximum age. This 1983 equation was chosen because it is the most universally applicable equation of its type, being derived from a combination of mollusk, fish, and cetacean data.

To further expand the usefulness of the natural mortality approach, we sought out another alternative estimate of total mortality from ecological modeling and stock assessments—the ratio of productivity to biomass, which is often used to estimate total mortality (i.e. P/B=Z). This approach yielded some additional recovery time estimates but we found that we were still lacking estimates for many fish and invertebrates (especially colonial invertebrates) and nearly all macrophytes. For these species we used one of two approaches, if a recovery time estimate was available for a similar or closely related species, we used that as a proxy. If not, as was the case for most macrophytes and colonial invertebrates, we developed a decision tree to estimate recovery time based on life history characteristics and growth forms. When all else fails we can default to a conservative RT of five years (e.g. equivalent to a maximum age of 22 years).

Table E1. Recovery time estimates for some fishes and solitary invertebrates derived from maximum age.

Common Name	T_{max} (years)	Calculated Recovery Time	Recovery time parameter (years)
Pacific sea nettle jelly	1	0.24	2
tidewater goby	1	0.24	2
crenate barnacle	2	0.47	2
brown rock crab	6	1.38	2
Pacific sanddab	10	2.27	2
red abalone	20	4.90	5
kelp bass	34	7.56	8
Pismo clam	53	11.69	12
bronzespotted rockfish	89	19.44	20
yelloweye rockfish	118	25.66	25

To estimate recovery times for invertebrates and macrophytes for which natural mortality-based estimates were unavailable, we used expert judgement to classify organisms by the life history characteristics, growth forms, and growth rates that are relevant to recovery times. We then used expert judgement to assign default recovery times for these categories.

Table E2. Example organisms defined by the characteristics that influence their recovery time and the resultant qualitative recovery time assessment and recovery time parameter derived through expert judgement.

	Characteristics					
Example organisms	**Longevity (annual/ perennial)**	**Means of reproduction (sexual/ asexual)**	**Replenishment (propagules/ vegetative)**	**Growth form (solitary/ colonial)**	**Qualitative recovery time estimate**	**Recovery time parameter (years)**
colonial ascidians and bryzoans	perennial	both	both, can brood embryos	colonial	short	2
boring sponges	perennial	both	both	colonial	short	2
encrusting sponges	perennial	both	both	colonial	moderate	3

solitary ascidians	perennial	sexual	propagules	solitary	longer	4
ball or vase sponges	perennial	both	both	solitary	longer	4
encrusting red coralline algae	perennial	sexual	both	solitary	longer	4
winged kelp (Alaria)	annual	sexual	propagules	solitary	short	2
giant kelp (*Macrocystis*)	perennial	sexual	both	solitary	longer	4

Estimates of recovery times for assemblages (RT_{assemb})

In keeping with the precautionary approach that we use throughout the impact evaluations, we defined the recovery time for an assemblage should be equal to the maximum recovery time of any organism in that assemblage. Because the four assemblages used throughout the evaluations are also habitat-specific, a recovery time is assigned to each assemblage-habitat combination. However to avoid under-estimating recovery times for assemblages that are not well represented in the literature, or for which not many recovery times have been estimated, we assigned default assemblage recovery times using a survey-style expert judgement approach (Appendix B). In application of the models, these default values would be used unless a longer recovery time was documented for a species in the assemblage, then the larger value would be used.

To estimate default values for RT_{assemb} all members of the workgroup assigned recovery categories from one to five to each habitat-assemblage combination. The categories were not intended to represent specific lengths of time, but to represent qualitative assessments of recovery time from short to long. The results of the survey were compared and the median values of the responses were calculated. These values were then reviewed by the group and translated into recovery time estimates by group consensus. Final estimates range from two to 14 years and are shown in Table 1.

Table E3. Default recovery times for assemblages as derived via the expert judgement survey approach described above.

Habitat	Fish	Mobile Invertebrates	Sessile Invertebrates	Macrophytes
Beach	NA	2	2	NA
0-30m soft	10	4	6	2
30-100m soft	10	6	8	NA
>100m soft	10	6	8	NA
rocky intertidal	10	6	6	8
0-30m rock	10	10	6	6
30-100m rock	10	10	6	8
>100m rock	10	10	14	NA
Pelagic	10	4	NA	NA
Estuary	6	4	4	8
Marsh	4	4	4	4

Estimates of recovery times for habitats (RT_{hab})

Similar to the process for estimating assemblage recovery times, habitat recovery times (RT_{hab}) were estimated using an expert judgement approach (see Appendix B). Throughout the decision framework, habitat refers to abiotic habitat only, thus the workgroup was estimating the time for altered physical habitat to return to its original state. For relatively static rock habitats, recovery times are likely to be long enough that habitat alterations are effectively permanent. However, for more dynamic sediment habitats, especially those in shallower depths, the actions of waves and currents are likely to gradually restore the habitat to its original state.

To estimate habitat recovery time, members of the workgroup assigned each habitat a recovery time category from one to five to reflect the relative length of recovery time. Scores were then compiled and translated to actual recovery times in years using group consensus approach Final habitat recovery values ranged from two to 20. Although some of the more dynamic habitats are likely to recover in days or weeks rather than years, allowing the value of the recovery time term to fall below one would cause the ultimate impact to habitats equation to effectively discount the impacts to habitats. As this outcome was not deemed conservative, we used a minimum habitat recovery value of 2 years, and capped habitat recovery times at 20 years.

Table E4. Recovery time estimates in years for physical habitats.

Habitat	Recovery Time (Years)
Beach	2
0-30m Soft	2
30-100m Soft	2
>100m Soft	2
>300m Soft	5
Rocky Intertidal	20
0-30m Rock	20
30-100m Rock	20
>100m Rock	20
Estuary	2
Marsh	2