NOAA Technical Memorandum NMFS-NWFSC-144

Marine Protected Resources on the U.S. West Coast:
Current Management and Opportunities for Applying Economic Analysis

https://doi.org/10.25923/vprp-1507

November 2018

U.S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Northwest Fisheries Science Center

NOAA Technical Memorandum NMFS-NWFSC-144

Marine Protected Resources on the U.S. West Coast:

Current Management and Opportunities for Applying Economic Analysis

https://doi.org/10.25923/vprp-1507

Robert Fonner[1] and Amanda Warlick[2]

[1]Conservation Biology Division
Northwest Fisheries Science Center
2725 Montlake Boulevard East
Seattle, Washington 98112

[2]Fishery Resource Analysis and Monitoring Division
Northwest Fisheries Science Center
2725 Montlake Boulevard East
Seattle, Washington 98112

November 2018

U.S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Northwest Fisheries Science Center
https://www.nwfsc.noaa.gov/index.cfm

NOAA Technical Memorandum NMFS-NWFSC Series

The Northwest Fisheries Science Center of NOAA's National Marine Fisheries Service uses the NOAA Technical Memorandum NMFS-NWFSC series to issue scientific and technical publications. Manuscripts have been peer-reviewed and edited. Publications in this series can be cited in the scientific and technical literature. Technical editing services at NWFSC are provided by Al Brown.

The Northwest Fisheries Science Center's NOAA Technical Memorandum NMFS-NWFSC series continues the NMFS-F/NWC series established in 1970 by the Northwest and Alaska Fisheries Science Center, which subsequently was divided into the Northwest Fisheries Science Center and the Alaska Fisheries Science Center. The latter uses the NOAA Technical Memorandum NMFS-AFSC series.

NOAA Technical Memorandums NMFS-NWFSC are available at the Northwest Fisheries Science Center website, https://www.nwfsc.noaa.gov/index.cfm.

Reference this document as follows:

Fonner, R., and A. Warlick. 2018. Marine Protected Resources on the U.S. West Coast: Current Management and Opportunities for Applying Economic Analysis. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-144. https://doi.org/10.25923/vprp-1507

Contents

Figures

Tables

Abstract

The role of economics marine protected resources (MPR) management is currently limited. However, economic tools may help address some of the challenges facing MPR managers. With a focus on NOAA's West Coast Region, this technical memorandum addresses two objectives. First, it identifies MPR management challenges and surveys economic tools and relevant literature for meeting those challenges. Second, it reviews the management status of individual west coast MPR species and identifies opportunities for applying economic analysis to inform management.

Acknowledgments

We thank the following reviewers for their helpful comments: Kelly Andrews, Manuel Bellanger, Bryant Chesney, Mike Ford, Rick Gustafson, Chris Harvey, Dan Holland, Rick Meyers, Mary Moser, Dawn Noren, Jameal Samhouri, and Jeff Seminoff.

Introduction

The National Marine Fisheries Service (NMFS), also called NOAA Fisheries, manages both the listing and recovery of marine species under the Endangered Species Act (ESA) and the protection of marine mammals under the Marine Mammal Protection Act (MMPA). This report focuses on NOAA's West Coast Region, which includes marine resources in California, Oregon, Washington, and Idaho. The region is host to 46 ESA-listed species, or distinct population segments (DPSs),[1] and over 30 species of marine mammals protected by the MMPA. Salmon and steelhead (salmonids, *Oncorhynchus* spp.) are at the forefront of marine protected resources (MPR) management in the West Coast Region, where 28 evolutionarily significant units (ESUs) of ESA-listed salmonids from six species are listed under the ESA. Other ESA-listed species in the region include killer whales (*Orcinus orca*), eulachon (*Thaleichthys pacificus*), yelloweye rockfish (*Sebastes ruberrimus*), bocaccio (*Sebastes paucispinis*), and green sturgeon (*Acipenser medirostris*).

Economic analysis currently plays a limited role in the recovery of ESA-listed species and management decisions. The ESA listing process does not permit economic analysis, but economics is leveraged to inform other ESA recovery planning and management decisions. For example, economic analyses are utilized in the process of designating critical habitat for ESA-listed species. Critical habitat designations are a mechanism designed to ensure that activities implemented, funded, or authorized by the federal government do not destroy or adversely impact habitat areas that listed species rely on for foraging, reproduction, and survival. Amendments to the ESA allow for exclusion of proposed critical habitat if a) the economic benefits of exclusion are sufficiently large, and b) the exclusion will not lead to species extinction. Government researchers estimate the potential economic costs of proposed critical habitat designations (CHD) to inform habitat exclusion decisions. Economic analysis also plays a role in evaluating federal actions. Federal management actions affecting MPR are subject to regulatory reviews under the National Environmental Policy Act (NEPA), and may involve consideration of ESA-related costs and benefits. Specifically, MPR managers can also be compelled to employ economic analysis when significant federal actions impact listed species. These cases involve a net present value comparison of the costs and benefits of an action in accordance with Executive Orders 12866 and 13563.

With regard to threats, the ESA directs scientists to assess five threat "factors" and determine whether species being considered for listing are *threatened* or *endangered* by any of the five factors. The threat assessment considers the severity, certainty, and geographic range of the threat, as well as the potential for threat reduction. The five factors include: 1) the present or threatened destruction, modification, or curtailment of its habitat or range; 2) overutilization for commercial, recreational, scientific, or educational purposes; 3) disease or predation; 4) inadequacy of existing regulatory mechanisms; and 5) other natural or human-made factors affecting continued existence of the populations.

[1] Under the U.S. Endangered Species Act, the listing unit for invertebrates is the taxonomic species; however, the listing unit for vertebrates includes species, subspecies, or distinct population segment. For Pacific salmon, the DPS equivalent is the evolutionarily significant unit, as described by Waples (1991).

NOAA economists held a workshop in 2014 to initiate the process of identifying national social science research needs and best practices related to MPR. A primary recommendation from the workshop was to conduct a high-level analysis assessing species threats and reviewing the policy instruments used to reduce species threats. The workshop further recommended that the analysis identify information needs for conducting threat evaluations and research opportunities for MPR economics research (NMFS 2014a). Our report implements the Protected Resources Economics Workshop recommendations for NOAA's West Coast Region. In accordance with workshop guidance, the analytical approach of this report is to assess the threats, recovery actions, and information/research needs of protected resources on the U.S. West Coast, one species at a time. The overall goal of this report is to identify opportunities for applying economic analysis to inform MPR management in the West Coast Region. To that end, the next chapter of this report synthesizes the species-specific analyses to identify common management challenges and roles for applying economic analysis to inform MPR management and research that exists across species. The remaining chapters present species-specific analyses. For each listed West Coast species,[2] these descriptions enumerate the threats facing species, summarize the scientific understanding of these threats, and identify current and potentially promising actions and policies for recovering listed species.

[2] Salmonids, rockfish, eulachon, cetaceans, pinnipeds, sea turtles, and abalone are discussed in separate chapters.

A Role for Economics in MPR Conservation and Recovery

NOAA's West Coast Region supports MPR species representing a wide variety of habitats, life histories, and threats. Moreover, the scientific understanding of species' threats, limiting factors, and the prospects for recovery vary significantly across listed species in the West Coast Region. This chapter synthesizes the species-specific analyses presented in subsequent chapters, including a summary of the threats facing protected species and opportunities for applying economic analysis to inform MPR management. Table 1 summarizes the specific threats affecting species reviewed in this report and aligns them with current and potential instruments of recovery designed to mitigate species threats.

This chapter proceeds in three sections. The next section synthesizes the species-specific analyses to identify common challenges for managing MPR in the West Coast Region. The second section discusses the current role of economics in MPR management and identifies opportunities for applying economic analysis to address MPR management challenges. The final section integrates our findings with the findings of prior reviews related to MPR economics.

MPR Management Challenges

Several common management challenges emerged across the species-specific chapters in this report. These challenges include: 1) allocating scarce recovery resources, 2) addressing scientific uncertainty, and 3) developing institutional arrangements that promote conservation and recovery. The MPR literature and past management outcomes in the West Coast Region point to three broad strategies for addressing these management challenges. These strategies involve utilizing economic analysis to a) inform the allocation of scarce recovery and conservation resources, b) prioritize research, monitoring, and data collection efforts, and c) design policy alternatives and predict their effects.

The resources available for implementing MPR conservation and recovery are increasingly scarce. Cumulative ESA listings have grown steadily since 1974, and, as of late 2016, almost 2,400 species were listed (Langpap, Kerkvliet, and Shogren 2018). Annual listings have averaged 35 species per year, and ranged from no new listings in 1974 to 129 new listings in 1994. Meanwhile, only 47 species have been delisted since the enactment of the ESA (Langpap, Kerkvliet, and Shogren 2018), resulting in a significant net increase in listed species over time. In sum, expenditures on species recovery efforts totaled at least $30 billion over the past 25 years (Langpap, Kerkvliet, and Shogren 2018). The funding available for species conservation and recovery is one of the best predictors of successful species recovery in an ESA context (Gerber 2016), and for reducing biodiversity loss internationally (Waldron et al. 2017). However, less than a quarter of the funds required for implementing ESA recovery plans are actually allocated to species recovery (Gerber 2016). Further, 24% of ESA-listed species were in decline and had underfunded recovery plans, while 12% of listed species were receiving recovery funding at or above levels specified in their recovery plans (Gerber 2016). These figures suggest that the current ESA system is under pressure from increasing listings, and that current funding levels are not adequate to meet recovery plan goals.

Table 1. Summary of threats and recovery instruments for MPR in NOAA's West Coast Region.

Species/ Species group	Major threat(s)	Current and potential recovery actions
Pacific salmonids (*Oncorhynchus* spp.)	Culverts, dams and other blockages to upstream habitat	Culvert/dam removal
	Warming ocean conditions	Global climate change mitigation
	High freshwater temperatures	Riparian vegetation restoration, increased flow
	Sedimentation from roads, urbanization, etc.	Road decommissioning
	Non-point-source pollution from agriculture	Water quality trading
	Point-source pollution from industry, storm water, sewage overflow	Technology or mitigation upgrades Discharge reductions
	Commercial and recreational harvest	Harvest reductions, seasonal closures
	Commercial bycatch	Bycatch caps, quotas, and risk pools Dynamic ocean management
	Hydropower dams (smolt mortality and migration impedance of adults and juveniles)	Strategically timed dam spillover Habitat restoration to mitigate for hydropower-based mortality
	Genetic change from hatchery fish	Hatchery release reductions Establishment of wild "gene banks"
	Competition from hatchery fish	Hatchery release reductions Portfolio-based hatchery management
	Predation (pinnipeds, pikeminnow, walleye, etc)	Predator removal programs, biological controls, angling derbies
	Destruction of spawning and juvenile rearing habitat	Wetland mitigation banks Side channel construction Floodplain restoration
Eulachon (*Thaleichthys pacificus*)	Changing ocean conditions from climate change	Global climate change mitigation
	Changing freshwater conditions from climate change	Global climate change mitigation
	Commercial fishing bycatch	LED gear modification[a]
	Predation	Predator management

[a] LED lights are attached to the footrope of the trawl net. LEDs change the behavior of the fish (hopefully) so that they do not enter the net. The existing metal rigid-grate excluder device is at the rear of the trawl net and acts to funnel fish out of a hole at the top of the net, after they have entered the net.

Table 1 (continued). Summary of threats and recovery instruments for MPR in NOAA's West Coast Region.

Species/ Species group	**Major threat(s)**	**Current and potential recovery actions**
Yelloweye rockfish (*Sebastes ruberrimus*) and bocaccio (*S. paucispinis*)	Derelict fishing gear	Detection and removal
	Commercial and recreational fishing bycatch	Education and outreach, science partnerships Dissemination of descending devices to anglers to mitigate barotrauma Enhanced enforcement
	Nearshore habitat disruption	Native kelp and eelgrass restoration
	Chemical contamination	Periodic assessments of sediment disposal Contaminated sediment cleanup
	Oil spills	Removal, remediation, restoration of spill damage
	Ocean acidification	Research related to kelp and eelgrass mediating effects
	Hypoxia/nutrient addition	Research related to environmental effects on movement behavior
Green sturgeon (*Acipenser medirostris*)	Restricted access to spawning habitat due to stream impoundments	Dam removal
	Bycatch in commercial and tribal fisheries	—
	Illegal take in recreational fisheries	Enforcement of prohibition on angling
	Exposure to contaminants	Remediation and monitoring
	Entrainment in water diversions, ocean energy projects, and vessel strikes	—
Black abalone (*Haliotis cracherodii*)	Mortality due to withering syndrome	Mapping habitats and baseline densities Population monitoring
White abalone (*H. sorenseni*)	Low densities inhibit successful reproduction	Captive breeding Transplantation and monitoring of captive-bred individuals
	Illegal harvest	Enforcement of harvest moratorium
	Reduced genetic diversity	—
Sea turtles (Chelonioidea)	Habitat modifications from climate change	—
	Ingestion of or entanglement in marine debris	Reduce ocean pollution
	Ingestion of or entanglement in derelict fishing gear	Gear tending and labeling requirements Gear removal efforts
	Bycatch in commercial fisheries	Gear modifications and time/area closures Real-time bycatch reporting and dynamic ocean management

Table 1 (continued). Summary of threats and recovery instruments for MPR in NOAA's West Coast Region.

Species/ Species group	**Major threat(s)**	**Current and potential recovery actions**
Sea turtles (continued)	Bycatch in recreational fisheries	Gear modifications and time/area closures
	Bycatch in foreign commercial and artisanal fisheries	Bycatch limits and incentives to modify fishing gear
	Poaching and illegal harvest	Conservation and monitoring of nesting beaches
	Predation of eggs on nesting beaches	Conservation and monitoring of nesting beaches
	Light pollution and other degradation of nesting beach habitat	Conservation and monitoring of nesting beaches
Pinnipeds[b]	Ingestion of or entanglement in marine debris	Reducing ocean pollution Increased research on degradable plastic packing bands
	Entanglement in derelict fishing gear	Gear tending and labeling requirements Gear removal efforts
	Bycatch in commercial fisheries	Gear modifications and time/area closures Real-time bycatch reporting and dynamic ocean management
	Bycatch in recreational fisheries	Gear modifications and time/area closures
	Prey depletion by commercial fisheries	Pinniped foraging requirements included in fisheries management
	Reduced prey availability due to climate change	Mitigation of global climate change
Cetaceans[c]	Entanglement in marine debris	Reducing pollution Development of degradable plastic packing bands
	Entanglement in derelict fishing gear	Gear tending and labeling requirements, gear removal efforts
	Bycatch in commercial fisheries	Gear modifications and time/area closures
	Vessel strikes	Vessel speed regulations Real-time collision reporting and dynamic ocean management
	Underwater noise	Noise-reducing technologies
	Whale watching vessel impacts	Vessel approach regulations Quota for allocating whale watching rights
	Chemical contamination of ocean habitats	Reducing ocean pollution
	Prey depletion by commercial fisheries	Include cetacean foraging requirements in fisheries management

[b] Guadalupe fur seals (*Arctocephalus townsendi*) and the western stock of Steller sea lions (*Eumetopias jubatus*) are listed under the ESA, and northern fur seals (*Callorhinus ursinus*) and the eastern stock of Steller sea lions are listed as depleted under the MMPA.
[c] ESA-listed cetaceans on the U.S. West Coast include: fin whales (*Balaenoptera physalus*), blue whales (*B. musculus*), sei whales (*B. borealis*), sperm whales (*Physeter macrocephalus*), humpback whales (*Megaptera novaeangliae*), North Pacific right whales (*Eubalaena japonica*), and the Southern Resident killer whales (*Orcinus orca*).

If this trend continues, MPR policymakers will increasingly face trade-offs between recovery actions, and potentially between species (Gerber 2016, Langpap, Kerkvliet, and Shogren 2018). Cost-effective resource allocation is particularly important for achieving recovery goals in the current climate of increasing resource scarcity, underscoring a need for economic analysis to inform MPR management.

Scientific and environmental uncertainty pose major challenges for MPR scientists and policymakers. Scientific uncertainty can take many forms. One form is uncertainty related to the threats facing, and biological needs of, protected species. There is also considerable uncertainty in how proposed recovery policies will influence ecosystems, and how these ecosystem changes will influence MPR populations and habitats. A third and related type of uncertainty pertains to the influence of proposed recovery policies on stakeholder behavior, which also influences ecosystem outcomes. Another consideration is environmental uncertainty, or the inherent variation in present environmental conditions and future environmental conditions. Scientific uncertainty can pose a variety of challenges to MPR policymakers. Chiefly, it complicates the development and evaluation of conservation and recovery policy alternatives, whose outcomes are inherently uncertain. Scientific uncertainty can also moderate which economic analyses and policy approaches are appropriate for a given MPR context. Reducing scientific uncertainty is possible through allocating resources for research, data collection, and monitoring resources. Managing MPR conservation uncertainty requires characterizing and managing extinction risk. This task may involve comparing the stochastic distribution of recovery outcomes associated with candidate recovery policies and assessing whether they buffer or exacerbate extinction risks associated with environmental variation or anthropogenic impacts. The ESA listing process involves an assessment of extinction risk, and recovery plans recommend policies to reduce and mitigate that risk. For critically endangered populations, infrequent events such as natural disasters or oil spills may pose significant extinction risks. Managing extinction risk extends beyond population abundance goals to considerations of genetic variability and spatial distribution of populations.

Subsequent discussions of uncertainty in this report focus on prioritizing and adapting research efforts to reduce and manage scientific uncertainty in specific MPR contexts. Notably, however, the economic literature also addresses uncertainty and learning in natural resource management in a broader context. For example, LaRiviere et al. (2017) characterize different types of uncertainty that arise in natural resource management, evaluate general hypotheses in the literature related to uncertainty, and assess how transferable insights regarding uncertainty are among natural resource contexts.

In addition to resource scarcity and scientific uncertainty, institutional arrangements pose challenges to MPR management in NOAA's West Coast Region. Institutions include official laws and processes, cultural practices, and societal norms. Williamson (2000) defined four levels of social analysis corresponding to economic theories and tools for advancing institutional understanding. The first level involves the analysis of norms, traditions, and other informal institutions that guide societal processes and evolve slowly over centuries. The second level is the institutional environment, which includes fundamental governing structures such as constitutions and formal laws, and, from an economic perspective, the assignment of property rights. The ESA and other MPR-related legislation, which generally evolves over decades, reside at this second institutional level. The third level of social analysis involves institutions of governance intended to promote order, reduce conflicts, and facilitate mutual gains. These governance structures can evolve within a decade's time. Analysis of this level with economics largely relates to theories and tools for analyzing transaction costs. The fourth level, which can change continuously, involves processes of resource allocation and employment that can be analyzed with neoclassical economics and agency theory. Institutions across these interacting institutional levels shape the incentives facing MPR stakeholders, thereby influencing their behavior. Moreover, a diverse set of institutions at various levels likely moderate the effectiveness of MPR conservation policies.

Informing MPR Management with Economic Analysis

Current Role and Future Opportunities

Economics' contribution to MPR science and management in the West Coast Region is currently limited. An internal review of protected fish science in the West Coast Region conducted in 2015 noted a "...complete lack of discussion of coordination between the natural sciences and economics and other social sciences," and recommended that "more effort should be placed on integrating the relevant social and natural sciences that bear on recovery" (NMFS 2015a, p. 2). Despite this lack of coordination, NOAA and its contractors regularly conduct several forms of economic analysis to support MPR management. These include analyses to inform critical habitat designations as part of the ESA process, and analyses to evaluate "economically significant regulatory actions" under EO 12866. A national review by NOAA economists identified 72 economic analyses conducted to support MPR regulatory actions over the past 15 years. Of these, one-third were analyses supporting CHDs, and half evaluated existing or proposed regulations on commercial fisheries (NMFS 2014a). The utility of economic analyses conducted to support CHD is unclear. Plantinga et al. (2014) concluded that current methodologies used to estimate the costs of CHD are sound, but inherent uncertainty makes development of useful estimates difficult. Further, the authors found no evidence that prior CHD economic analyses provided useful information for making exclusion decisions.

The degree to which ESA recovery resources are misallocated, and the impact of potential misallocations on conservation outcomes, is not well understood, and likely differs across species and contexts. Langpap and Kerkvliet (2010) evaluated ESA resource allocations for vertebrates and found that, while actual allocations were significantly misaligned with stated priorities, the inconsistencies did not have significant detrimental effects for conservation objectives. The authors also found that increased funding for recovery programs improved ESA outcomes. Finally, the authors found that trade-offs exist between the stated ESA objectives of recovering populations and preventing extinctions. Specifically, the authors find that preventing a single extinction comes at an opportunity cost of improving the status of two other populations.

Guidance is lacking on when and how government researchers should undertake economic analysis to inform MPR management outside of mandated regulatory analysis and CHD economic analyses (NMFS 2014a). A handbook on best practices could provide researchers with guidance on conducting economic analysis for MPR management, including navigating the complexities of extinction risk, environmental uncertainty, stakeholder incentives, and conservation institutions.

Economic Tools for MPR Management

Economics provides researchers with analytical models and empirical methodologies for addressing MPR management challenges and informing MPR policies. When MPR policies influence stakeholder welfare, incentives, and behavior, economics can provide frameworks for understanding these effects. Economic frameworks for understanding such phenomena include theories and models of human incentives, behavior, and strategic interactions. Economics also provides empirical tools for evaluating recovery alternatives, estimating stakeholder preferences, modeling behavioral responses, and understanding the causal effects of recovery policies.

Most economic analyses of MPR build upon some biophysical understanding of threats facing the resource and the direct impact of conservation actions on recovery objectives. Economic tools can then link biophysical outcomes to changes in ecosystem service flows and human welfare.

It is beyond the scope of this report to enumerate all forms of economic analysis that can be applied to MPR conservation and recovery. Instead, we identify economic tools that can help address management problems that can arise when developing MPR management policies. Table 2 aligns common management problems with relevant economic tools and literature that can help address them.

In the following sections, we further explore the economic tools in Table 2. First, we discuss methods for measuring stakeholder preferences and welfare changes associated with MPR policies and outcomes. Next, we review tools for evaluating candidate MPR policies. We conclude the section by surveying methods and literature related to developing MPR policies and understanding their economic, social, and biophysical effects.

Estimating stakeholder preferences and values

Economic tools are capable of estimating the values stakeholders hold for conservation policies and recovery outcomes. The economic costs and benefits of some recovery policies, such as hydroelectric dam spillover, are directly observable in markets. However, the economic values of policies that influence ecosystem service flows are sometimes not revealed by market transactions. Economic values associated with species recovery and conservation outcomes are typically nonuse values not revealed in markets.

Economists utilize stated preference (SP) and revealed preference (RP) methods to estimate the economic value of goods whose values are not revealed in markets (i.e., nonmarket values). Stated preference studies employ survey techniques to elicit preferences over alternative conservation policies and willingness to pay for related nonmarket values (e.g., ecosystem services). These methods are capable of estimating a wide range of nonmarket values, and are particularly useful for measuring nonuse values such as the values stakeholders hold for the ongoing existence of a species. The two most common SP methods are contingent valuation and choice experiments. Contingent valuation methods face respondents with discrete choices across alternate environmental and policy conditions, often in a referendum format. Choice experiments face respondents with choices across alternative environmental outcomes that systematically vary according to specified ecosystem service attributes and their cost to respondents.

Table 2. MPR management problems and associated economic tools and literature. Key: *CEA* = cost-effectiveness analysis, *CBA* = cost–benefit analysis.

Management problem	**Relevant economic tool(s)**	**Use of tool**	**Relevant literature**
Inform allocation of scarce resources for PR recovery and management	Ecosystem services evaluation	Measuring public preferences for species recovery	Wallmo and Lew 2012
	CEA, CBA	Evaluating and prioritizing recovery action alternatives	Newbold and Siikamäki 2009, Sanchirico et al. 2013, Gjertsen et al. 2014
Prioritize research, monitoring, and data collection efforts	CEA, CBA	Evaluating alternatives to reduce scientific uncertainty	Bisack and Magnusson 2014
Identify policy instruments that account for stakeholder incentives	Market-based policy instruments	Incentivizing behavior that advances conservation goals	Bisack and Sutinen 2006, Holland 2010, Innes et al. 2015
Evaluate policy instruments	Causal policy analysis	Understand the effect past policies had on outcomes of interest	Athey and Imbens 2017
	Behavioral modeling	Understand how stakeholders respond to incentives associated with MPR policies	Haynie and Layton 2010
	Bioeconomic modeling	Optimal control of management problems, dynamic policy simulation	Chan and Pan 2016
	Institutional and behavioral economics	Understand how formal and informal institutions shape stakeholder incentives and conservation outcomes; understand contexts where stakeholder behavior departs from rationality assumptions	Ostrom 2002, Abbot and Wilen 2009, Ostrom and Basurto 2011, Barnes-Mauthe et al. 2015, Sawchuk et al. 2015, Li, van den Brink, and Woltjer 2016, Stephenson et al. 2017, Aburto-Oropeza et al. 2018
	Nonmarket valuation	Evaluate stakeholder and public preferences for conservation policies and associated outcomes	Bell, Huppert, and Johnson 2003, Anderson, Lee, and Levin 2013, Forbes et al. 2015, Lew 2015, Dundas, von Haefen, and Mansfield 2018

RP methods value ecosystem services through behavioral observations under utility maximization assumptions. Travel cost models use data on trip-taking behavior and distance traveled to the sites, often recreation sites, to estimate demand for site visits. Hedonic pricing models decompose the sales prices of real estate transactions and other differentiated goods by their attributes (e.g., lot size or square footage), which include environmental amenities.

Ecological production functions are another method for valuing environmental goods. This method models ecosystems as factors of production for valuable goods and services. For example, the production function approach could value water quality in tidal flats by estimating its role as an input to shellfish aquaculture production.

Appropriate valuation of nonmarket goods and services requires significant expertise and financial resources. When estimation of these values is not feasible, rigorous application of benefit transfer (BT)[3] methods can also inform recovery policy evaluation. The economic literature includes numerous applications of nonmarket valuation that measure stakeholder preferences and values for the preservation, protection, and enhancement of MPR. These include SP studies that value species recovery and associated policies using contingent valuation (e.g., Bell, Huppert, and Johnson 2003) and choice experiments (e.g., Wallmo and Lew 2012). Lew (2015) provides a detailed review of past SP studies that measure willingness-to-pay for MPR recovery and conservation. RP studies are also relevant for MPR valuation. For example, Netusil and Summers (2009) used hedonic pricing to decompose payments for instream flows according to the relevant attributes, including whether the stream included ESA-listed species, and Dundas, von Haefen, and Mansfield (2018) used the travel cost method to value the costs of recreational off-road vehicle closures for endangered species protection. Using a production function approach, Barbier (2007) valued coastal mangrove forests as inputs to producing commercial fishing and marketable goods.

Estimates of ecosystem service values and stakeholder preferences can inform the development and evaluation of MPR management policies. Cost–benefit analysis, which is discussed below, can require estimation of nonmarket benefits and costs with the tools discussed above. Likewise, stakeholder preference parameters and welfare measures can serve as inputs to evaluation criteria for recovery action evaluation (e.g., Sanchirico et al. 2013).

Evaluating alternative recovery actions and research efforts

Economic tools can also evaluate alternative research policies and policy portfolios. Cost-effectiveness analysis (CEA) and cost–benefit analysis (CBA) are two common frameworks for evaluating MPR policies. CEA is a decision framework that identifies least-cost alternatives for achieving a specified threshold (e.g., achieving recovery goals). Alternatively, CEA can identify alternatives that yield the most progress towards conservation objectives subject to a specified resource budget. Protected resources policy objectives are typically related to species recovery. Thus, applying CEA in these contexts requires a meaningful understanding of how candidate recovery actions influence progress toward recovery and the costs of candidate actions. A key advantage of CEA is that it does not require estimation of the benefits of species recovery or other nonmarket benefits of conservation alternatives. In one recent application of CEA to MPR management, Gjertsen et al. (2014) used a population model and cost data in Monte Carlo simulations to conduct a stochastic CEA of alternative conservation strategies for leatherback turtles (*Dermochelys coriacea*).

[3] Benefit transfer involves applying values derived from some study site to a different policy site (Weber 2015).

By comparison, CBA compares alternatives based on a comparison of total costs to total benefits. CBA evaluates alternatives based on their net benefits and involves estimation of the costs and benefits associated with alternatives. CBA is used either to evaluate the soundness of a single alternative or to compare multiple alternatives. One MPR-related CBA was completed by the Department of the Interior and partners to evaluate the costs and benefits of removing four hydroelectric dams on the Klamath River in California, in part to improve habitat for imperiled wild salmonid populations (DOI 2012).

Economic impacts, distributional effects, and environmental justice are other factors to consider in addition to economic costs and benefits. Economic impact analyses examine the effects of alternatives on business revenue, wages, and employment in a specified community or area. Distributional effects refer to how the costs and benefits of policies are distributed across the affected population. These effects are important to consider in the context of species recovery, where the benefits are likely to be geographically widespread but the costs may be concentrated in a certain community or region (Boxall et al. 2012, Mansfield et al. 2012, Sanchirico et al. 2013).

The timing of recovery costs and benefits is an important consideration in evaluating policy alternatives. In general, stakeholders prefer current benefits to future benefits and prefer incurring costs in the future to incurring costs now. These time preferences are typically incorporated into CEA and CBA through the discounting of future values; though determining an appropriate discount rate is challenging. The optimal timing of policy instrument implementation is also of interest to economists and policy makers. Speir et al. (2015) considered the optimal timing of dam removal to protect endangered salmonids in California. The authors found that uncertainty in ecosystem costs creates public incentives to delay implementation while irreversibility effects such as extinction create incentives to hasten implementation.

The efficiency and effectiveness of recovery policies may depend on the timing of policy implementation across interacting ecosystem components, including interactions across trophic levels. Samhouri et al. (2017) concluded that the synchronous recovery of predators and prey is usually more expedient than sequential recovery strategies. The authors further found that predator-first recovery strategies, which may in some cases carry additional political salience due to their being endeared to the public (see Lew 2015), are particularly slow. Samhouri et al. (2017) also surveyed real-world management policies and found that in practice, both sequential and synchronous recovery strategies are common, further underscoring the potential for coordinated ecosystem-based recovery strategies to increase the efficiency and effectiveness of recovery policies compared to the status quo. Addressing an economics conference, Adamowicz (2016) highlighted the importance of recovery instrument timing, including stakeholder incentives to delay implementation and the defining recovery horizons that weigh management trade-offs.

Besides serving as a framework to prioritize policy alternatives, CEA and CBA can help address scientific uncertainty challenges. For example, CEA can help inform data collection and monitoring effort allocation decisions when these activities are undertaken for the purpose of increasing the predictive power or parameter precision of a statistical model. Estimating and analyzing the value of the scientific information is another tool for addressing uncertainty with economic analysis. The value of scientific information can include the value of reduced extinction risk associated with improved decision-making or the value to stakeholders of relaxing use regulations. Likewise, CBA can inform evaluation of candidate research efforts based on the economic value of the scientific information that is expected to be obtained from those allocations. For example, increasing the precision of marine mammal population estimates (i.e., reducing uncertainty) could allow for a reduction of the regulatory burden on fishermen. Bisack and Magnusson (2014) examined the economic value of increased precision in marine mammal stock estimates. Their results suggest that for one Atlantic fishery, abundance surveys are a more cost-effective means of reducing uncertainty than increasing observer coverage. The authors also found that the benefits to private industry of increased stock estimate precision more than offset the costs of the data collection. This study represents an application of economic tools to measure the net value of scientific information. In another application related to uncertainty, Tomberlin (2010) addressed the question of whether to undertake costly biological monitoring or allow human use when it is not known whether a species is present in a given habitat area. The study provides an example of using dynamic economic analysis to operationalize adaptive management when learning is possible.

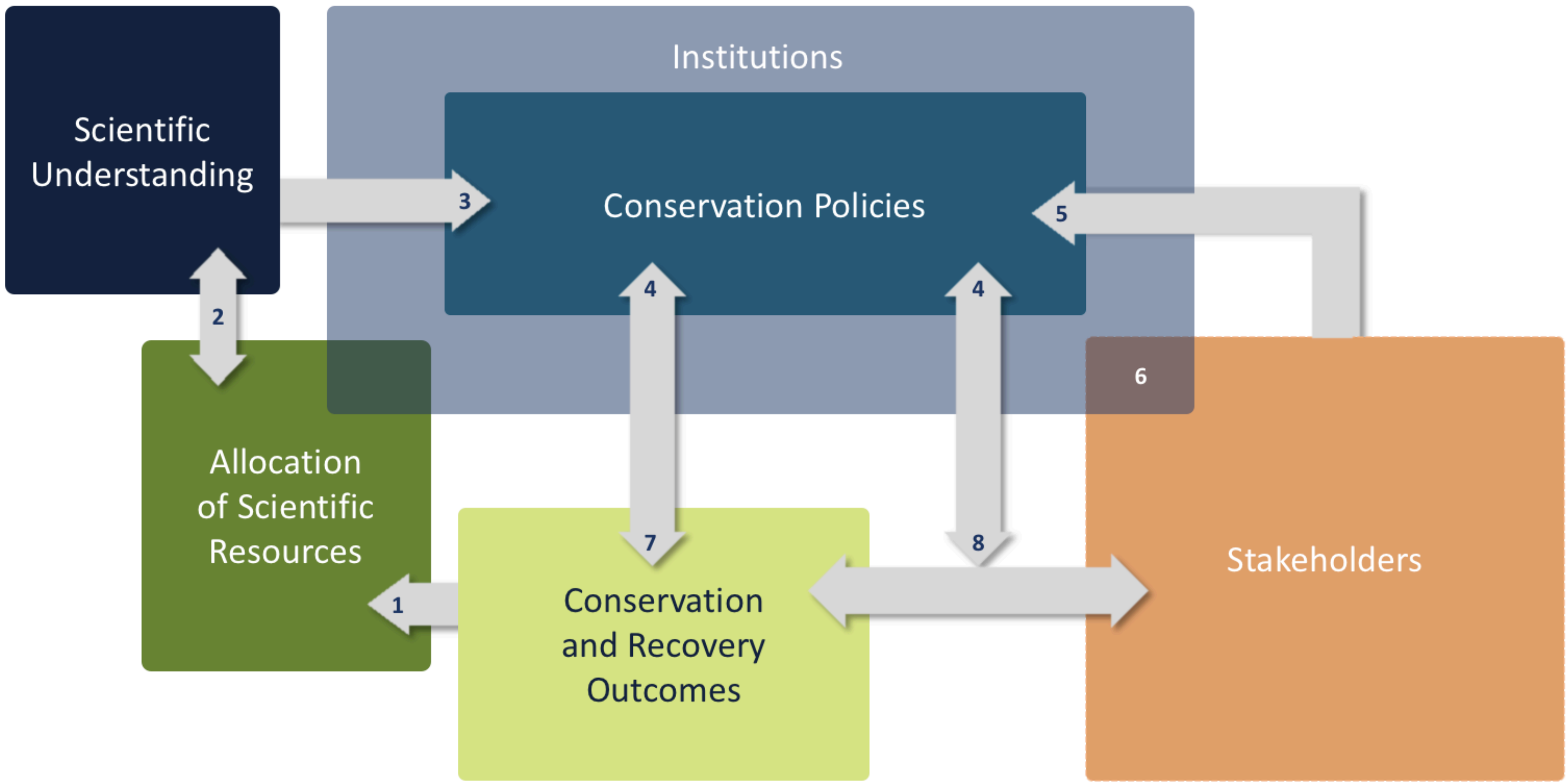
Developing policy instruments

Designing and evaluating policy instruments to promote conservation and recovery is a difficult but essential problem for MPR policymakers. Figure 1 depicts select linkages connecting conservation policies to stakeholder behavior, scientific priorities, and recovery outcomes. While additional linkages likely exist, we focus on those that are highlighted in this report and relevant to developing MPR policy instruments. Developing MPR conservation policies requires assessing the threats facing MPR and proposing policy alternatives for addressing the threats. Policymakers then evaluate alternative policies to determine how financial resources should be allocated to advance conservation objectives. Estimation of stakeholder preferences informs understanding of how conservation policies and conservation outcomes influence stakeholder welfare. In Figure 1, the arrow from conservation policies to stakeholders depicts these relationships. On the other hand, the arrow from stakeholders to conservation policies depicts direct stakeholder involvement in the policymaking process.

A first step in developing alternative recovery instruments is enumerating and assessing the threats facing the MPR of concern, including interactions among related threats. A second step is to identify strategies for reducing threats to achieve conservation objectives. A third step is to assess the uncertainty in threats and expected threat mitigation. Implementing these steps depends heavily on an understanding of relevant baseline conditions and limiting factors. It also emphasizes the importance of understanding the state of underlying biological science, and its uncertainty,[4] when designing and evaluating recovery policy instruments.

MPR policy instruments include actions that reduce threats through direct modification of habitat features (e.g., habitat restoration, derelict fishing gear removal, predator removal) and actions intended to modify stakeholder behavior to reduce threats (e.g., fishing regulations, water quality standards). These instrument types are depicted in Figure 1 (arrows seven and eight, respectively). Designing policy instruments to incentivize environmentally beneficial stakeholder behavior requires an understanding of the dynamics of interacting human and biophysical systems, and how potential policy levers influence stakeholder incentives. Economists employ behavioral and bioeconomic modeling to understand these processes.

[4] See Essington, Sanchirico, and Baskett (2018) for an example of how ecological uncertainty can moderate appropriate management strategies.



Instrument features			Policy instrument types
Adaptive management		Stakeholder involvement	
1. Conservation & recovery outcomes inform research priorities	3. New science informs conservation policies	5. Stakeholder input informs conservation policies	7. Policy instruments that reduce species threats through direct modification of ecosystems & habitat features
2. New science informs the allocation if scientific resources	4. Conservation & management outcomes inform conservation policies	6. Self-governance by stakeholders through formal & informal institutions	8. Policy instruments intended to modify stakeholder behavior to reduce threats

Figure 1. Translating MPR policy instruments into conservation and recovery outcomes. Each component of the process (colored boxes) influences and is influenced by several others in a complex network. Numbers draw attention to features and types of policy instruments (see legend). Arrows represent ways that components of the process influence each other that are relevant to developing MPR policies. The overlapping boxes reflect that scientific resource allocations and stakeholders can form institutions that influence recovery outcomes.

Policy instruments intended to influence stakeholder behavior can be divided into top-down and market-based instruments. Top-down, or "command and control" approaches, directly regulate the behavior of stakeholders. An example of command and control instruments could be the prohibition of certain economic activities through critical habitat designations. Conversely, market-based policies do not regulate uses directly, but instead incentivize behavior that promotes conservation objectives. Market-based approaches include cap-and-trade systems that address common-pool resource problems through establishment of property rights, as well as carrot-and-stick approaches that reward socially desirable behavior and penalize undesirable behavior. The cap-and-trade approach is applicable to pollution control, commercial fisheries target catch (e.g., individual tradable quotas), commercial fisheries bycatch (Lent and Squires 2017), and tourism (Higham et al. 2016). Carrot-and-stick approaches are also applicable to bycatch management and MPR regulations, as evidenced by the recently enacted Chinook salmon (*Oncorhynchus tshawytscha*) bycatch reduction incentive plan in the Bering Sea–Aleutian Islands (Alaska) pollock (*Gadus chalcogrammus*) fishery (USOFR 2016). This policy incentivizes bycatch avoidance by restricting pollock fishing opportunities for vessels with poor bycatch performance while rewarding vessels with good bycatch performance with less restrictive access to fishing grounds. Market-based instruments also include taxes that incentivize environmentally desirable behavior (e.g., a bycatch tax). Finally, risk pools are another market-based policy that provides a means for sharing the risk of unintended MPR take across stakeholders (Holland and Jannot 2012). Lent (2015) and others have advocated for market-based instruments to help address the major threats facing marine mammals. For a review of using market-based instruments to mitigate undesirable impacts, see Innes et al. (2015).

Economists and other social scientists also acknowledge the ability and importance of stakeholders in developing and enforcing conservation policies and institutions (Ostrom 2002, Lennox et al. 2013, Sawchuk et al. 2015, Field et al. 2017, Grand et al. 2017). Likewise, implementation of the ESA on private lands has shifted focus from post-listing regulation to engaging stakeholders and offering prelisting incentives to avoid the costs associated with listing regulations (Langpap and Wu 2017).

Evaluating conservation policies that modify habitats directly is straightforward if the costs, biological response, and uncertainty associated with alternatives are known. By comparison, evaluating conservation policies designed to modify stakeholder behavior requires understanding how policy alternatives influence stakeholder incentives, behavior, and welfare. Economic tools for modeling and analyzing stakeholder incentives and behavior, and analyzing and predicting policy outcomes, can help policymakers understand the potential effects of proposed recovery policies. The influence of alternative policies on stakeholder incentives, behavior, and welfare determines how those policies influence conservation outcomes and public wellbeing. In an application from the literature, Bisack and Sutinen (2006) modeled fisher behavior in the New England gillnet fishery under alternative harbor porpoise (*Phocoena phocoena*) bycatch reduction policies. The authors found that ITQ management of bycatch leads to higher fishing rents compared to management with closures. Economists can also use stated preference to evaluate defined policy alternatives and conservation outcomes. In another application, Forbes et al. (2015) used a choice experiment to estimate consumer preferences over methods to recover endangered Canadian rockfish (*Sebastes* spp.).

Strategic interactions can arise in situations where one stakeholder's preferred behavior depends on the behavior of other stakeholders. Economists use game theory, or the mathematical analysis of strategic behavior between decision-makers, and related economic tools to help explain stakeholder incentives and predict stakeholder behavior in these situations (e.g., Abbott and Wilen 2009, White et al. 2012).

Another important type of stakeholder behavior is the decision of whether to comply with regulations. Economic tools can examine compliance decisions and associated stakeholder incentives (e.g., Bisack and Das 2015, Langpap and Wu 2017).

In addition to evaluating candidate recovery policies, economists also study how past policies have affected conservation outcomes with tools designed to investigate causality between policy treatments and intended effects. Random controlled trials (RCT) are the ideal tool for determining causality, but are often infeasible or unethical in the MPR context. Other tools include instrumental variables, difference-in-differences, regression discontinuity designs, and traditional potential outcome approaches (see Athey and Imbens [2017] for a recent review). In the context of Figure 1, the goal of causal policy analysis is to improve understanding of the linkage depicted by the arrow leading from conservation outcomes to scientific understanding. In other words, causal policy analysis illuminates how past conservation policies influenced outcomes of interest to policymakers.

To evaluate some recovery instruments, economists must model how management actions interact with ecosystem processes and stakeholder preferences to shape stakeholder behaviors and conservation outcomes over time. They accomplish this by modeling how stakeholders respond to incentives, associated institutions, and the natural environment, and how those responses in turn influence biophysical systems and conservation outcomes. These bioeconomic models are used in protected resources contexts for dynamic analysis, optimal control, and dynamic policy simulations. Policy simulations often rely on estimates from biophysical studies as well as estimates of stakeholder preferences, incentives, and responsiveness (e.g., Anderson, Lee, and Levin 2013).

Sometimes recovery and conservation policies cause spillovers, or effects beyond the jurisdiction or boundary of the policy. These effects can occur through a variety of channels, including input reallocation, prices, and ecological–physical linkages (see Pfaff and Robalino [2017] for a review). Economic analysis provides tools for constructing counterfactual scenarios and policy simulations to evaluate spillover effects. For example, Chan and Pan (2016) found that fishery regulations designed to reduce sea turtle (*Chelonioidea* spp.) mortality by closing a U.S. fishery once a bycatch limit had been reached led to a spillover effect of higher fishing effort by non-U.S. vessels.

Finally, economists use institutional analysis to address common-pool resource problems and other market failures associated with formal or informal institutional arrangements. The importance of institutions in shaping conservation outcomes was a key finding of this report. This concept is further emphasized in Figure 1, where stakeholder involvement and informal stakeholder institutions are identified by bubbles five and six. While not shown on the figure, institutions also moderate the effects that conservation policies have on stakeholders, conservation outcomes, and scientific resource allocation. Institutions also moderate how scientific advances influence conservation policies.

Economics provides tools to analyze the role of institutions in MPR management and recovery. For example, Hanna (2008) evaluated institutional requirements for managing healthy and resilient salmonid ecosystems with an emphasis on the role of stakeholder incentives and transaction costs. Ostrom and Basurto (2011) developed tools for evaluating alternative institutional arrangements and Stephenson et al. (2017) offered guidance for integrating institutional and economic considerations into fisheries policy. Li, van den Brink, and Woltjer

(2016) used institutional analysis to develop a governance framework for considering ecosystem services in coastal and marine natural resource management. In addition to formal institutions, stakeholder engagement, education, and involvement are increasingly recognized as important institutional considerations for protected resources management (e.g., Sawchuk et al. 2015, Duvall et al. 2017, Aburto-Oropeza et al. 2018). The development of social norms (Ostrom 2002) and the influence of social capital (e.g., Barnes-Mauthe et al. 2015) on environmental outcomes are other informal institutions of interest to economists and other social scientists.

Summary and Integration with Prior Reviews

This report summarizes the threats, recovery instruments, and scientific understanding associated with each MPR species in NOAA's West Coast Region. Salient points from the species sections were synthesized in the proceeding discussion to assess the challenges facing west coast MPR management and opportunities for addressing the challenges with economic tools. The challenges identified include managing scientific uncertainty, scarce resources, and institutional challenges. The opportunities include utilizing economic tools to inform the allocation of scarce recovery and conservation resources; prioritize research, monitoring, and data collection efforts; and design policy alternatives and predict their effects.

Economists have reviewed the role of economics in protected resources management in other contexts. A seminal review by Shogren et al. (1999) argued that economics matters to endangered species management because: 1) human behavior and economic factors help determine species risk, 2) it is important to take into account the opportunity costs of species protection, and 3) economic incentives are central to shaping human behavior, and, in turn, species recovery. With regard to the Canadian context, Adamowicz (2016) identified the role of CBA, species triage, the timing of species recovery, climate change effects, market-based policy instruments, and equity impacts as some of the important topics where economics can contribute to endangered species management. Finally, a recent article by Langpap, Kerkvliet, and Shogren (2018) surveys new theory and evidence on resource scarcity, the economic costs of species recovery, and the incentives present when endangered species are on private lands. The authors identify stakeholder incentives associated with imperiled but unlisted species, the role of environmental organizations, and the political economy of the ESA process as topics for future research.

Pacific Salmonids

Current Status

Historically, millions of anadromous salmonids (*Oncorhynchus* spp.*)* filled the estuaries, rivers, and streams of the U.S. West Coast. Over the past century, however, salmonid runs in the western U.S. have declined markedly due to a myriad of anthropogenic activities. An estimated 29% of the nearly 1,400 Pacific salmonid populations that once inhabited the Pacific Northwest and California have gone extinct since Euro-American contact. Likewise, between 16% and 30% of all historical ESUs in this area are estimated to have been lost (Gustafson et al. 2007). The remaining populations occupy an estimated 40% of their original range, and half of those salmon populations in the Pacific Northwest are thought be at risk of extinction (Ruckelshaus et al. 2002). Within the Columbia River basin alone, 84 percent of salmonid populations are not considered to be viable (McClure et al. 2003). These declines have led to the listing of portions of five species of Pacific salmonids (Chinook [*O. tshawytscha*], chum [*O. keta*], coho [*O. kisutch*], and sockeye salmon [*O. nerka*], and steelhead [*O. mykiss*]) under the ESA. These listings afford protections for 28 ESUs[5] and DPSs across Washington, Idaho, Oregon, and California. NOAA Fisheries defines an ESU as a stock of salmon that is substantially reproductively isolated from other population units and represents an important component in the evolutionary legacy of the species (Waples 1991). The following sections describe the threats facing west coast salmonid species and synthesize the body of scientific literature that seeks to understand the relative importance of those threats and the prioritization of recovery action alternatives.

Assessment and Evaluation of Threats

Scientists have traditionally categorized the anthropogenic threats that affect Pacific salmon into four categories known as the “four Hs:” Habitat, Hydropower, Hatcheries, and Harvest. This section broadly reviews each threat category and discusses how the four-H classification system does not fully capture the true complexity of the situation facing protected salmonids on the west coast.

Rapid development in the west coast region over the past 150 years led to significant degradation of salmon habitat, the first “H.” The sources of these impacts are diverse, and include: agricultural, urban (e.g., residential, commercial, and industrial), waterway (e.g., dredging, channelization) and natural resource (e.g., forestry, mining) development. Moreover, anadromous salmonids occupy a diverse range of habitats over the course of their life cycle, all of which are subject to modification by human activity. For example, human activity directly affects freshwater habitats that support spawning, rearing of juvenile salmon, and passage to the sea. Essential salmon habitat characteristics subject to anthropogenic impacts include freshwater and ocean temperature and pH, suitable spawning substrate, water quality (oxygen, toxicity, fine sediment), and water quantity (i.e., in-stream flow).

[5] The ESA listing unit for steelhead was initially the ESU, but under the 1996 joint FWS–NMFS DPS policy memorandum (USOFR 1996) this was changed to the distinct population segment (DPS).

The second "H," hydropower, refers to the threats Pacific salmon face in freshwater habitats due to hydropower dams. Dams do provide a number of economic benefits, including hydropower, water storage, transportation facilitation, and flood control. While dams provide services for some stakeholders, they also represent a threat to salmonid sustainability. Dams can impede passage by adult salmon (as well as other migratory fishes) returning to spawn, and by juveniles as they migrate to the sea. Dams also modify freshwater habitat, creating reservoirs where rivers once existed. These modifications can reduce the quality and quantity of essential fish habitats for spawning and juvenile rearing and modify the ecological communities living in those habitats. In many cases, changes in the ecological community can pose additional threats to salmon (e.g., increased predation by introduced and native lacustrine species during the migratory delay). Moreover, as this discussion illustrates, the four Hs are not independent but rather synergistic, overlapping, and interrelated threats; in addition to being a migratory barrier, hydropower is directly responsible for habitat degradation.

The main function of salmon hatcheries, the third "H," is to supplement harvest opportunities in the face of declining wild fish runs. In addition to providing additional fish for harvest, salmon hatchery objectives may include supplementing weak stocks, recolonizing areas where salmon have been extirpated, and providing cultural resources. However, mounting scientific evidence suggests that hatchery fish can interfere with the viability of protected wild stocks through ecological (e.g., competition, increased predation of wild stocks) and genetic interactions (interbreeding between domesticated hatchery fish and locally adapted native fish; Naish et al. 2007). Buhle et al. (2009) found that increased hatchery operations are associated with lower wild run productivity. Moreover, supplementation programs in the Pacific Northwest have largely proven ineffective at providing successful colonists to reestablish viable salmonid runs (Waples et al. 2007). As with hydropower, hatcheries can also influence habitat quality. Specifically, hatchery weirs can block access to historical spawning and rearing habitat, and hatchery effluent can be responsible for water-quality and disease threats.

The fourth "H," harvest of Pacific salmonid species, has long provided value for tribal, recreational, and commercial fishermen. However, by the latter part of the twentieth century, overharvest of Pacific salmon likely contributed to significant population declines for many west coast populations. For example, catches in the troll fishery in Oregon peaked in 1976 with a catch of more than 1.8 million fish before starting a gradual decline. From 1979–92, coho salmon landings averaged 357,000, with a maximum annual catch of 697,000 fish over that period (ODFW 1998). Harvest was one of several factors that contributed to the decline, and Oregon coho were ESA-listed in 1998. Today, commercial, recreational, and tribal harvests are at a fraction of their historical levels. However, because it is difficult for fishermen to avoid catching weak stocks, and because salmon are a bycatch species in some west coast fisheries (e.g., Pacific whiting, *Merluccius productus*), harvest still represents a source of mortality for protected salmonid stocks.

While the four Hs provide a starting point for understanding the threats facing protected Pacific salmonids, these categories have significant overlap and interdependencies. For example, hydropower projects can reduce available salmon habitat and lead to ecological shifts that increase salmon predation. Likewise, hatcheries can affect the quantity of habitat available to wild salmon (through competition) and the threat posed by salmon harvests.

Pacific salmon management continues to evolve, and the focus on threats posed by the four Hs in the academic literature has in part given way to a focus on emerging threats such as climate change (Crozier 2016) and marine mammal predation (Chasco et al. 2017b). However, unlike climate change and other threats with diffuse sources, threats posed by the four Hs are directly connected to salmon management decisions and remain relevant to salmon recovery planning. The threats described by the four Hs are also influenced by dynamic processes such as changes in climate and ocean conditions and trends in human population growth and economic development (Maas-Hebner et al. 2016). Understanding the interdependencies among threats, and how external processes affect the severity of threats, is a challenging but essential question for salmon scientists moving forward. Pacific salmonids face an interconnected and dynamic array of threats. Furthermore, the threats facing salmonids vary significantly across populations.

The following paragraphs describe the criteria NOAA Fisheries uses to determine the viability of salmonid populations and review literature evaluating the relative impacts of threats on viability.

NOAA's units for salmonid endangered ESA listings, ESUs and DPSs, represent reproductively isolated groups of significant populations that share distinct evolutionary histories. Within ESUs, major population groups (MPGs) are identified based on their genetics, spawning and rearing habitat, and life histories (i.e., ocean distribution, juvenile and adult migration timing, age at return). The viability of individual populations within these groups is evaluated by NOAA scientists based on their abundance (i.e., number of spawners over generations), productivity (i.e., population growth rate), spatial structure,[6] and diversity[7] (McElhany et al. 2000). NOAA also considers the influence of resilience to catastrophic risk, facilitation of metapopulation processes, and maintenance of among-population diversity in estimating the viability of an ESU or DPS.

While scientists generally agree on the criteria for determining population viability, there is less consensus regarding how to model viability and how threats influence viability. Many studies in the literature focus on linking a single threat or threat category to the viability of a single population or population group. These studies are important for understanding threats, but they often ignore the spatial heterogeneity and interdependencies of threats across the west coast region. The development of models that consider multiple, interrelated threats and weigh the relative impact of those threats are especially useful for guiding more efficient population recovery plans. However, such models are difficult to develop and require data inputs that are often not available. Moreover, the diffuse nature of climate change, population growth, and other threats facing salmonids make understanding the relative influence of threats on populations especially difficult. Despite these challenges, a number of studies have considered the influence of multiple salmonid threats simultaneously and assessed their relative effects on population viability (e.g., Scheuerell et al. 2006, Hoekstra et al. 2007, Fullerton et al. 2011, Lawrence et al. 2014). The results of these studies are largely context-specific, but they provide methodological foundations for future applications in new areas and contexts.

[6] The geographic distribution of a population and the processes supporting that distribution. Rationale is to have multiple spawning reaches within a population and allow high-production "source" areas to supplement other population segments. This protects against catastrophic events and periods of low survival.
[7] Considers life histories and traits, genetic characteristics, dispersal, and gene flows.

Instruments of Recovery

This section reviews literature that applies economic analysis to recovery action evaluation and prioritization using cost effectiveness analysis and cost–benefit analysis.

A number of studies in the literature conduct cost-effectiveness analysis across habitat restoration alternatives (Paulsen and Wernstedt 1995, Watanabe et al. 2006, Newbold and Siikamäki 2009, Fullerton et al. 2010b, Ogston et al. 2014). The study by Paulsen and Wernstedt (1995) conducted a CEA across habitat-, harvest-, and hatchery-related recovery actions. Watanabe et al. (2006) find that targeting habitat features rather than biological recovery can lead to substantial inefficiencies. Fullerton et al. (2010b) incorporate restoration costs within a simulation framework to capture the economic and biological uncertainty associated with alternative recovery actions. Ogston et al. (2014) show that the cost of improving natural smolt production via cost-effective habitat restoration is similar to that of traditional hatchery smolt production.

The location of restoration activities affects an action's effectiveness. Watersheds are spatially connected; it follows that the impact of habitat restoration varies significantly depending on where the habitat is restored (e.g., Watanabe et al. 2005, 2006, Newbold and Siikamäki 2009, Fullerton et al. 2010a, Null and Lund 2012). Further, Barnas et al. (2015) found that enacted restoration projects in the Pacific Northwest tend to be misaligned with the biological needs of salmon at the subwatershed scale. The implication for managers is that effectiveness is heterogeneous at subwatershed scale within and across restoration actions. Moreover, the spatial distribution of fish within and across river basins is an important component of viability for ESUs (McElhany et al. 2000).

Data Gaps and Opportunities for Applying Economic Analysis

The preceding sections reviewed the threats facing protected Pacific salmon populations, alternative recovery actions, and methods for prioritizing recovery actions. The breadth of issues covered in this review illustrates the complexities involved with recovery of Pacific salmon populations. Sources of this complexity include:

1. The life histories of salmon and the wide range of habitats they occupy.
2. The diverse set of stakeholders and associated values for salmon resources and habitat.
3. The institutional landscape spanning state, federal, and tribal arenas.
4. The interactions among threats facing Pacific salmon populations.
5. The uncertain impact climate change will have on salmon across their diverse habitats.

This section identifies areas where economics and other social sciences may contribute to informing efficient and effective policies for Pacific salmon recovery. We begin by describing an idealized procedure for identifying and prioritizing salmon recovery actions and then identify where economics and other social sciences can contribute to the process.

Understanding how human activities and the natural environment influence the viability of salmon populations is critical for developing effective recovery policies. Models of these processes already exist for specific areas and contexts. Ideally, models will accurately incorporate the myriad of factors influencing population viability, capture the interplay among these factors, and characterize model uncertainty across time and space. Moreover, the spatial scope of models would be expanded to, at a minimum, the extent of habitat utilized by a given ESU or DPS.

A next step in identifying effective instruments for Pacific salmon recovery is constructing scenarios that define how factors affecting population viability evolve from their baseline state over time. These scenarios require the identification of specific alternative futures for climate change and human development. They can then serve as inputs into the population models to construct status-quo population viability predictions under a given set of assumptions.

With baseline scenarios specified, analysts may then construct a set of specific recovery actions expected to influence salmon population viability. Next, sets of recovery actions can be identified that, according to the specified models, will achieve population (ESU) recovery objectives. Potential recovery plans can be winnowed down based on their recovery performance and institutional feasibility. Recovery performance would be judged based on the ability to influence viability (production, diversity, spatial structure) and the robustness of performance across the alternative future scenarios. Additional considerations for the development of these models include incorporating effectiveness interactions among the recovery alternatives, and integrating new scientific information into models as it becomes available (i.e., adaptive management).

Economics, which concerns itself with the efficient allocation of scarce resources, brings tools to bear for comparison of alternative recovery plans. CEA requires information on the costs of alternative recovery actions. If scenarios are constructed as defined above, cost-effectiveness analysis assigns costs to the alternatives, so that the lowest-cost plan for meeting biological goals can be selected. Alternatively, if restoration managers are tasked with allocating a specified budget, cost-effectiveness analysis prioritizes alternatives that generate the largest biological response per restoration dollar spent. CEA is most useful in cases where the costs associated with a given recovery action are easily observed. This is generally the case, for example, when prioritizing alternative habitat restoration projects. There are a number of studies that use CEA to prioritize restoration actions (e.g., Watanabe et al. 2005, 2006, Newbold and Siikamäki 2009, Fullerton et al. 2010b, Null and Lund 2012, Ogston et al. 2014, Barnas et al. 2015); however, these studies consider only a small fraction of the potential areas and actions considered for restoration. Given the significant resources that are spent on Pacific salmon habitat restoration, expanding the use of CEA could substantially improve the effectiveness and efficiency of these restoration efforts. Such an expansion of CEA for habitat restoration would involve scaling up existing biophysical models of salmon population viability across basins.[8] Furthermore, such models would enable restoration managers to map restoration priorities across the landscape and craft appropriate land-acquisition strategies.

[8] Currently these models have been developed for only a limited number of watershed areas.

The number of federally listed species has been steadily growing for decades (Langpap, Kerkvliet, and Shogren 2018). If this trend persists, a day may come when allocation decisions will involve trade-offs between salmonid ESUs and DPSs. With regard to this possibility, scientists have discussed a form of recovery triage (e.g., Levin and Stunz 2005, Gerber 2016), where resources are diverted from ESUs that are "too far gone" to those for which recovery is possible.[9] If this situation arises, economic analysis (e.g., CEA, CBA, nonmarket valuation) could help to compare the expected biological responses achievable across ESUs with the available resources. Moreover, stated preference studies could measure public preferences for species recovery outcomes to further illuminate management trade-offs and inform the allocation of resources across ESUs.

Conflicting protected species mandates can present a further obstacle to endangered salmon recovery. One visible example of this emerges from recent heavy predation of salmon by protected sea lions below Bonneville Dam on the Columbia River. Management alternatives for controlling sea lion populations are limited by the MMPA, despite their threat to listed salmon. A second case of conflicting protected species mandates occurs when Southern Resident killer whales (listed under the MMPA and ESA) prey upon the ESA-listed Puget Sound Chinook salmon population. These and other cases of conflicting and overlapping mandates suggest that more holistic, ecosystem-based approaches could improve the effectiveness of ESA-listed Pacific salmon management.

The literature on Pacific salmon biology, management, and recovery collectively echoes a number of important themes that help define desirable characteristics of Pacific salmon recovery policies. First, recovery policies are more likely to be effective if they are robust to uncertainty, particularly the uncertainty related to climate change. Second, effective recovery policies account for the complex set of interdependent threats facing salmon. Third, effective recovery policies are readily adaptable to emerging science, environmental change, and institutional change. Finally, and most relevant to the current report, socially desirable recovery policies account for the economic trade-offs associated with salmon recovery policies. Economic analysis can contribute to this discussion through estimation of the total economic value, and associated distributional impacts, associated with alternative salmon recovery policies. Economics can further contribute through analysis of the institutions (e.g., regulations, property rights, informal agreements) and stakeholder dynamics associated with salmon recovery management.

[9] The process for writing off a listed species is complex, and involves the "god-squad." The process is to remove a species from the list in order to allocate elsewhere, rather than doing it in a de facto way by withholding resources.

Eulachon

Current Status

Eulachon (*Thaleichthys pacificus*), also known as "Columbia River smelt," "hooligan," or "candlefish," is a small (adults 160–250 mm) anadromous fish from the smelt family that occurs over the continental shelf in the northeastern Pacific Ocean. Eulachon populations range from northern California to the southeastern coast of Alaska in the Bering Sea (Willson et al. 2006, Moody and Pitcher 2010). The fish live their adult lives (2–5 years) at sea before returning to freshwater to spawn from late winter to spring (Gustafson et al. 2010). Eulachon spawning habitat is characterized by cool waters (0–10°C) in the lower reaches of large, snowmelt-fed rivers (Gustafson et al. 2010). Likewise, eulachon thrive in cool, nutrient-rich marine waters. Eulachon have long been an important harvest species for local indigenous peoples who consume them fresh, smoked, dried, and salted, and, north of the Fraser River, rendered into eulachon oil or "grease" (eulachon are 15–20% fat). Historically, large commercial and recreational fisheries also existed for the species in its freshwater habitats (Gustafson et al. 2016).[10]

The process of listing eulachon as a protected species began in 1999 when NMFS was petitioned to list the Columbia River population under the Endangered Species Act. This petition was rejected due to a lack of supporting information. In 2007, the Cowlitz Indian Tribe petitioned to list eulachon, this time for all populations in Washington, Oregon, and California. After analyzing genetic and ecological data, NMFS's biological review team (BRT) found that a DPS exists from northern California to and including the Skeena River in northern British Columbia. This "Southern DPS" can be further divided into four subpopulations: Mainland British Columbia, Fraser River, Columbia River and tributaries, and Klamath River. The BRT further found that the Southern DPS eulachon faced a "moderate risk of extinction throughout all of its range" (Gustafson et al. 2010, p. 176). The Southern DPS eulachon was listed as threatened under the ESA in 2010.

An updated review of the status of the Southern DPS was conducted in 2016. The review found that since listing, eulachon abundance had increased in all four Southern DPS subpopulations. However, the review recommended that the listing status of eulachon remain unchanged, since abundance increases were related to temporarily favorable ocean conditions which had since dissipated. In the words of the status review report:

> Although eulachon abundance in monitored populations has generally improved, especially in the 2013–2015 return years, recent poor ocean conditions and the likelihood that these conditions will persist into the near future suggest that population declines may be widespread in the upcoming return years (Gustafson et al. 2016).

This finding underscores the difficulty with assessing recovery for the Southern DPS eulachon. Observed population abundance depends on stochastic environmental conditions, and can vary substantially from year to year.

[10] All fisheries for eulachon occur on fish returning to spawn in freshwater. The only marine catch occurs as bycatch in shrimp and groundfish fisheries.

Historical and recent observations of eulachon abundance further illustrate challenges involved in determining the status of the Southern DPS. Historical accounts of Columbia River basin eulachon runs describe an unpredictable fishery with significant variation in run size, spatial distribution, and timing (Gustafson et al. 2010). However, yearly commercial fishery catch in the Columbia River was consistently over 500 metric tons (mt)—and often over 1,000 mt—for three-quarters of a century from about 1915 to 1992, regardless of the prevailing ocean conditions (Gustafson et al. 2010). Recent observations also reflect high variation in the abundance of Southern DPS eulachon in its freshwater and ocean habitats. In freshwater, for example, estimated total eulachon run biomass in the Columbia River between the years 2000 and 2010 ranged from a high of 3,105 mt in 2001 to a low of 35 mt in 2005 (Gustafson et al. 2016). Evidence suggests that eulachon abundance varies widely in the ocean as well. Particularly, the large variance in bycatch ratios (eulachon per mt of target fish) by commercial fishermen suggests significant interannual variation in the abundance of eulachon populations. Ward et al. (2015) examined eulachon bycatch in the west coast shrimp fishery and found “that increases in bycatch [are] not due to an increase in incidental targeting of eulachon by fishing vessels, but because of an increasing population size of eulachon.”

Assessment and Evaluation of Threats

In its 2010 status review, the BRT rated the threats facing each subpopulation of Southern DPS eulachon from one (very low) to five (very high). On aggregate, the BRT rated climate change impacts on ocean conditions as the most severe threat, rating it as high-level or moderate-level across all four subpopulations. Other identified threats included: bycatch, dams/water diversions, predation, dredging, and climate change impacts on freshwater systems. Table 3 contains a summary of the threats facing Southern DPS eulachon that received a moderate or higher rating of extinction risk for at least one subpopulation.

Table 3. Eulachon threats by severity for each subpopulation. Severity was assessed by the 2010 BRT.

		Threat severity			
Threat type	**Threat description**	**Klamath**	**Columbia**	**BC**	**Fraser**
Climate change	Climate change impacts on ocean conditions	high	high	high	high
	Climate change impacts on freshwater habitat	moderate	moderate	moderate	moderate
Interactions with human activities	Bycatch	moderate	high	high	moderate
	Dams/water diversions	moderate	moderate	very low	very low
	Dredging[a]	very low	moderate	very low	low
Interactions with other species	Predation	moderate	moderate	moderate	moderate
Habitat degradation	Water quality	moderate	moderate	low	moderate
	Shoreline construction	very low	moderate	low	moderate

[a] Can also degrade eulachon habitat.

Since its threatened listing under the ESA in 2010, the abundance of Southern DPS eulachon has increased in each subpopulation. High relative levels of eulachon abundance from 2011–16 are likely attributable to favorable ocean conditions during that period (Gustafson et al. 2016). However, recent conditions have been less favorable, including development of the "warm blob" during the winter of 2013–14 and the subsequent strong El Niño that occurred from 2015–16. Gustafson et al. (2016) concluded that these unfavorable ocean conditions, combined with an ongoing degradation of freshwater habitats, may reverse recent gains in eulachon abundance. Spawning stock biomass estimates in 2017–18 indicate that eulachon abundance in the Fraser, Columbia, and Klamath subpopulations has declined to levels similar to those seen at the time of listing.

Uncertainty is perhaps the greatest challenge in restoring eulachon populations. Particularly, fisheries scientists do not understand how environmental conditions directly or indirectly influence eulachon survival (Gustafson et al. 2016). Furthermore, observed abundance of eulachon can vary widely, making it difficult to discern trends in population viability and the effectiveness of policy instruments aimed at population recovery. Reducing this uncertainty will require substantial research and monitoring effort. In its five-year listing review (NMFS 2016a), NOAA Fisheries recommended implementing monitoring that can adequately detect changes in eulachon habitat and eulachon survival. In terms of habitat assessment, this effort would include monitoring of large-scale oceanographic conditions in the California Current and analysis of how these conditions influence planktonic assemblages and, in turn, larval survival in nearshore environments. Some of this information is already being collected or compiled by NOAA scientists through the California Current Integrated Ecosystem Assessment.[11] Furthermore, parallel monitoring and modeling efforts in freshwater environments will be required to understand how environmental conditions impact eulachon survival in riverine and estuarine habitats.

As noted above, climate change impacts were the most severe threat identified by the BRT in 2010. The drivers and impacts of climate change are highly diffuse, meaning that many parties are both responsible for and impacted by climate change at a global scale. Thus, it is nearly impossible to make responsible parties internalize the external costs of climate change on eulachon populations. Because of this, eulachon-specific policy instruments may not be sufficient to protect eulachon from climate-related impacts. Threats from interactions with human activities and habitat degradation are more easily addressed on a local scale. However, these threats are generally associated with economic activity (e.g., commercial fishing, river dredging, dams), and stakeholders may be reluctant to curb their production to improve conditions for eulachon. Understanding the incentives facing stakeholders and demonstrating the link between economic activities and eulachon population impacts is critical to crafting effective policies.

[11] https://www.integratedecosystemassessment.noaa.gov/regions/california-current-region/index.html

Instruments of Recovery

Since it was listed in 2010, the only regulation enacted for eulachon protection is the prohibition of take in California's inland waters (NMFS 2016a). However, a limited number of existing regulations restrict stakeholders' operations in eulachon habitats. After the listing, areas of critical habitat were designated for eulachon in the lower reaches of a number of large rivers within its range. Further, the ESA stipulates that actions with a federal nexus must consult with NOAA when said actions may affect eulachon populations or adversely modify designated critical habitat. In British Columbia, recreational fishing with nets is prohibited, and there is a dredging moratorium within eulachon habitat. While not mandated by regulation, one of the most promising recent advances has been the development of LED excluder lights that help prevent eulachon bycatch in the ocean shrimp fishery.[12]

Very few policy instruments directed at eulachon recovery have been deployed. The effectiveness of instruments that are in place is also largely unknown. This uncertainty stems from poor understanding of how biophysical conditions in eulachon habitat will evolve and, further, from poor understanding of the mechanism linking biophysical conditions to eulachon survival and recovery. One recovery effort that has demonstrated success in reducing eulachon mortality is the development and widespread adoption of LED gear lights by shrimp fishers to reduce eulachon bycatch, which is a high-priority threat in some subpopulations (Table 3). In 2014, after observing high eulachon bycatch rates in the ocean shrimp fishery, the Oregon Department of Fish and Wildlife developed LED gear lights as a technical bycatch reduction mechanism. Subsequent experiments suggested that these lights were highly effective. Particularly, 42 paired trials showed that gear equipped with the LED lights had 91% less eulachon bycatch than hauls using gear without LED lights (NMFS 2016a).

Determining which policy instruments will be effective in facilitating eulachon recovery requires a better understanding of the threats facing eulachon and the mechanisms by which threats impact eulachon populations.

Data Gaps and Opportunities for Applying Economic Analysis

One challenge in implementing this strategy is that the drivers and impacts of climate variability and climate change, which the BRT assessed as the largest threat facing eulachon, are not well understood. As climate science evolves, understanding of promising eulachon management instruments may also advance. This review concludes that opportunities exist for enacting recovery instruments that could improve the viability of other protected species (e.g., salmon and steelhead) as well as eulachon under the emerging ecosystem-based management (EBM) framework. Protecting multiple species within the same policy instrument may tip the scales when evaluating the costs and benefits of that instrument (e.g., estuary restoration, dam removal).

[12] Beginning 1 April 2018, all Oregon and Washington shrimpers are required by state law to use fishing lights on the footrope of each trawl net. This measure was estimated to cost about $1,300 annually per vessel, although most shrimpers were already using the lights since they reduce the sorting time of the catch (R. Gustafson, NWFSC, personal communication).

Puget Sound Rockfish

Current Status

Yelloweye rockfish (*Sebastes ruberrimus*) and bocaccio (*S. paucispinis*) live in Pacific Ocean waters from California to Alaska. Upon being petitioned, NOAA determined that the Puget Sound/Georgia Basin (hereafter referred to as Puget Sound) populations of yelloweye rockfish and bocaccio represent a DPS. Notably, the Puget Sound DPS extends north into the Canadian portion of the Salish Sea. In 2010, the Puget Sound DPSs of yelloweye rockfish and bocaccio were designated threatened and endangered, respectively. Canary rockfish (*S. pinniger*) were included in the original listing, but were delisted in 2017 after research (NMFS 2016d, Andrews et al. 2018) found that the Puget Sound population was not genetically distinct from the larger outer-coast population. Using this new information, along with known life history characteristics, NOAA determined that the population of canary rockfish in Puget Sound does not meet the criteria for being designated as a DPS. Total rockfish abundance has declined by ~70% over the past 40 years, while bocaccio and yelloweye rockfish have likely declined by an even greater amount.

Assessment and Evaluation of Threats

NOAA's final recovery plan (NMFS 2017) includes an assessment of the threats facing rockfish in Puget Sound. The ESA directs scientists to assess five threat "factors" and determine whether species being considered for listing are threatened or endangered by any of the five factors. The threat assessment considered the severity, certainty, and geographic range of the threat, as well as the potential for threat reduction. The five factors include: 1) the present or threatened destruction, modification, or curtailment of its habitat or range; 2) overutilization for commercial, recreational, scientific, or educational purposes; 3) disease or predation; 4) inadequacy of existing regulatory mechanisms; and 5) other natural or human-made factors affecting continued existence of the populations. Members of the Rockfish Recovery Plan Team determined that threats from four of the five categories pose moderate to high risks to Puget Sound rockfish. For the fifth category, disease and predation, there were not enough data to determine threat severity in this case. Table 4 presents a summary of NOAA's threat assessment findings by management unit. Notably, the relative severity of threats varies across space, a pattern that reflects the heterogeneity of economic activity and urbanization across management units. Habitat-based threats pose the highest risk in the southern management units (Hood Canal, Main Basin, South Sound) through hypoxia and nearshore habitat disruption. Within factor 5), derelict fishing gear poses a moderate or high risk in the northern (San Juan and Canada) and western (Hood Canal) management units, but very low risk in the other management areas (Main Basin, South Sound). Conversely, contamination poses a moderate or high threat in Hood Canal, Main Basin, and South Sound, but poses a low or very low risk in the other management units.

Commercial and recreational fishing for rockfish is prohibited in Puget Sound; however, there are a few fisheries (e.g., commercial and recreational halibut and recreational lingcod and salmon) in which bycatch of rockfish still poses mortality risk to ESA-listed rockfish, primarily in the San Juan management unit. To mitigate some of this risk, the Washington Department of Fish and Wildlife has limited recreational fishing for "bottom fish" (including lingcod) in Puget Sound to water depths <120 ft, as yelloweye rockfish and bocaccio are more likely to occur in deeper waters.

Table 4. Summary of threats assessment for ESA-listed rockfish in different management units of the Puget Sound/Georgia Basin (NMFS 2017).

	Listing factor	Canada	San Juan	Main Basin	South Sound	Hood Canal
Derelict fishing gear	E	1	1	2	4	4
Commercial catch/bycatch	B, D	3	1	3	3	3
Recreational catch/bycatch	B, D	3	1	2	3	4
Nearshore habitat disruption	A	4	3	1	1	2
Deepwater habitat disruption	A	3	3	3	3	3
Non-native species habitat disruption	E	P	P	P	P	P
Hypoxia/nutrient addition	E	4	4	3	2	1
Chemical contamination/bioaccumulants	A	3	3	1	1	2
		Entire Puget Sound/Georgia Basin				
Marine mammal predation	C	4				
Fish predation/hatchery practices	C, E	4				
Competition	C	P				
Diseases	C	P				
Oil spills	E	1				
Genetic changes	E	P				
Anthropogenic noise	E	P				
Ocean acidification	E	1				
Climate change	E	1				

Key:

A = Present or threatened destruction/modification/curtailment of habitat or range
B = Overutilization for commercial, recreational, scientific, or educational purpose
C = Disease or predation
D = Inadequacy of existing regulatory mechanism
E = Other natural or human-made factors affecting continued existence
P = Potential threat (not enough information to determine if it is a threat at the present time, but could plausibly become a threat in the future)

1 = High risk
2 = Moderate risk
3 = Low risk
4 = Very low risk

The interrelated threats of climate change and ocean acidification pose a significant risk to future persistence of listed rockfish in Puget Sound through a multitude of mechanisms. Likewise, all management units face high risk from oil-spill contamination.

Instruments of Recovery

NOAA evaluates recovery actions based on threat severity, uncertainty, and likelihood of recovery. The final recovery plan for ESA-listed rockfish in Puget Sound specifies recovery action priorities (NMFS 2017). Several recovery actions have been undertaken since the rockfish ESA listing. Government, academic, and conservation group researchers are collaborating with local stakeholders to improve understanding of rockfish in Puget Sound and the factors threatening their viability.

NOAA's 2017 final recovery plan (NMFS 2017) recommended and prioritized an additional 45 recovery actions for listed rockfish among five distinct categories. The categories included actions that improve understanding of rockfish abundance, biology, and habitat associations; actions that align fisheries management with recovery goals; actions that research and protect rockfish habitats; actions that educate fishermen on identifying rockfish and reducing rockfish bycatch mortality; and actions that secure additional resources for rockfish recovery. In 2010, the Washington State Fish and Wildlife Commission passed regulations prohibiting targeting of rockfish by recreational anglers. Managers also closed recreational angling for bottom fish in waters deeper than 120 feet to reduce rockfish bycatch (NMFS 2017).

The recovery plan specifies rockfish recovery actions, including cost estimates and priority scores. The actions target three objectives. The first objective is to further our understanding of the status of rockfish populations and their habitats. Associated actions include scientific surveys, genetic testing, and other biological research. The second objective is to reduce or eliminate existing threats to listed rockfish associated with fisheries and other human activities. The third objective is to restore, protect, and research rockfish habitat. The associated actions include derelict gear prevention and removal, and research on rockfish habitat.

Commercial fishing in Puget Sound for fish species has been largely nonexistent for a few decades, but the Washington Department of Fish and Wildlife (WDFW) did close several of the remaining nontribal commercial fisheries in the summer of 2010.[13] As a condition for receiving an incidental take permit, WDFW initiated a monitoring and management program to reduce interactions with protected rockfish in two remaining Puget Sound commercial fisheries. In particular, the state initiated an observer program in the shrimp trawl fishery.

The Department of Fisheries and Oceans (DFO) in Canada also took action to protect listed rockfish from fishing threats. The efforts included improving stock assessments and monitoring and designating rockfish conservation areas where fishing is limited. Using these tools, DFO's policy is to ensure that rockfish are subjected to fisheries mortality that is less than half that of natural mortality.

The Northwest Straits Commission received a federal grant to remove over 4,500 derelict fishing nets and 140 pots from waters less than 100 feet (NMFS 2015b). These efforts reduced the threat posed by derelict gear and improved habitat quality for rockfish. Removing derelict gear from deeper waters is much more technologically challenging, but NOAA has funded a pilot program in the San Juan management area to detect and map deep-water (i.e., >100 ft) derelict gear using sonar technology. NOAA Fisheries is also working with partners to quantify mortality associated with derelict gear through genetic sampling of dead fish found in gear.

In 2012, Washington State passed a bill requiring nontribal fishers to report lost nets within 24 hours of loss so that they can be retrieved. The legislature appropriated $3.5 million dollars to fund the program in 2013, leading to the removal of 5,660 nets and 3,800 shellfish pots and 813 acres of improved habitat (NMFS 2015b). Additionally, a variety of government, conservation, and academic partners are collaborating to map and characterize the Puget Sound benthic habitat in terms of its suitability for listed rockfish. This effort will inform, among other things, future abundance surveys, critical habitat designation reviews, and fisheries management decisions.

[13] These included the set net, set line, bottom trawl, inactive pelagic trawl, and inactive bottom fish pot fisheries.

Future habitat recommendations in the draft recovery plan include restoring nearshore and deep-water habitats, cleaning up contaminated sediments, and expanding habitat monitoring efforts. The recovery plan also recommends following Canada in designating priority habitats as protected areas for listed rockfish (e.g., marine protected areas or rockfish conservation areas).

Uncertainty about the biology and ecology of rockfish in Puget Sound and the threats they face is an obstacle to creating effective recovery plans. After the ESA listing, NOAA Fisheries initiated research to improve scientific understanding of rockfish in Puget Sound. A number of these efforts leveraged the knowledge and perspectives of fishermen to inform policy. Notably, in 2013–14, NOAA Fisheries partnered with WDFW and the recreational fishing community in Puget Sound to locate and genetically sample ESA-listed rockfish. The samples revealed that Puget Sound canary rockfish are not genetically distinct from canary rockfish on the outer coast, which led NOAA Fisheries to determine that canary rockfish did not meet the criteria of being designated a DPS, which led to the delisting of canary rockfish in 2017.

City, state, tribal, and federal agencies, along with regional foundations, aquariums, and nonprofit organizations, have ongoing research projects related to the recovery of rockfish in Puget Sound. These studies are targeted at understanding: a) historic and present-day rockfish abundance and diversity using remotely operated vehicle (ROV), trawl, and scuba-based surveys; b) spatial habitat usage and post-bycatch mortality using acoustic tracking technology; c) connectivity of populations using larval dispersal modeling; and d) fisheries interactions by working with local fishing boat captains and volunteer anglers (NMFS 2015b).

Data Gaps and Opportunities for Applying Economic Analysis

In many ways, the listing and recovery planning process for rockfish in Puget Sound reflects effective implementation of ESA principles and objectives. After the rockfish listing, NOAA biologists initiated research to reduce the uncertainty associated with the determination that these rockfish populations in Puget Sound are distinct population segments. This research engaged local fishing captains and anglers and leveraged their unique ecological knowledge to collect data on rockfish genetics. The results of this research led to the eventual delisting of canary rockfish and expanded the geographic boundaries of the Puget Sound DPS for yelloweye rockfish.

The process that led to the delisting of Puget Sound canary rockfish illustrates two effective components of MPR research. First, the researchers engaged local stakeholders to improve understanding of MPR science (Beaudreau et al. 2011, Andrews et al. 2018) and used historical population observations to inform delisting criteria (Williams et al. 2010). Second, the delisting illustrates adaptive management, where management priorities and strategies are modified as understanding of recovery science develops. Additionally, the cost estimates of recommended recovery actions in the Rockfish Recovery Plan are potentially useful for conducting future analyses. Economic analysis could further improve the Puget Sound Rockfish Recovery Plan through a quantitative prioritization of recovery actions. The draft plan acknowledges that all of the actions in the recovery plan are not possible under current funding levels (NMFS 2015b), yet rates 38 of the 45 recommended recovery actions as the highest possible priority. Application of CEA or other analysis frameworks to the plan may further differentiate prioritization across actions. Still, the analysis conducted for the listing and recovery of ESA-listed rockfish in Puget Sound stands out among the species considered in this report in terms of approach, implementation, and adaptability.

Green Sturgeon

Current Status

Green sturgeon (GS; *Acipenser medirostris*) is an ESA-listed species that inhabits mostly estuarine and coastal waters off the U.S. West Coast, from the Bering Sea in Alaska to Baja California, Mexico. GS are anadromous fish, meaning they migrate from ocean-rearing habitats into rivers to spawn in freshwater upstream. Despite their wide geographic range, known GS spawning habitat is restricted to the Sacramento, Klamath, and Rogue River basins (Moser et al. 2016).

A petition to list GS under the ESA was filed in 2001. The subsequent biological review (NMFS 2015c) found that the North American population of GS is split into distinct population segments. The review also found that the Southern DPS is composed of those fish spawning in the Sacramento River. The Southern DPS of GS was officially listed as threatened under the ESA in 2006. At that time, GS were assigned a Recovery Priority Number of 5, indicating a moderate risk of extinction. The five-year review in 2015 found the GS status unchanged, and further noted that while the recovery potential was high in some regions, conflicts exist between conservation and natural resource uses (NMFS 2015c).

Two primary factors likely contributed to historical declines in GS abundance: incidental harvest, and destruction and degradation of habitat (NMFS 2010). Improved understanding of GS life history traits, population trends, and habitat needs is necessary to facilitate effective management and recovery (Moser et al. 2016).

Assessment and Evaluation of Threats

The mandated 2010 biological review found that the principal threat to Southern DPS green sturgeon was restricted access to spawning habitat due to stream impoundments (NMFS 2010). In addition, stream impoundments alter the hydrograph and can cause harmful river flow and temperature conditions, particularly below large dams (e.g., Shasta, Keswick, Oroville, and Daguerre Point). Other important threats identified in the review, listed in no particular order, include:

- Bycatch in commercial and tribal fisheries.
- Illegal take in recreational fisheries.
- Displacement of prey by nonnative species.
- Entrainment in water diversions, ocean energy projects, and vessel strikes.
- Exposure to contaminants.

Annual GS bycatch in commercial, recreational, and tribal fisheries has historically numbered in the thousands of individuals. For example, bycatch of GS in Columbia River and Washington State commercial fisheries ranged from 3,000 to 7,500 individuals from 1985 to 1993 (Moser et al. 2016). Larval and juvenile GS are susceptible to entrainment in the over 3,000 water diversion pipes operating in the Sacramento–San Joaquin basin (Mussen et al. 2014).

Instruments of Recovery

Green sturgeon bycatch was historically sold to seafood processors, but commercial sale of GS was prohibited upon listing in 2006. Recreational harvest is also prohibited in Washington, Oregon, and California. All sturgeon fishing was prohibited year-round in the upper Sacramento River in 2010 to protect spawning adults (NMFS 2010). In 2009, critical habitat was designated for GS spanning marine, coastal, estuarine, and freshwater segments. The freshwater segments include the mainstem Sacramento River below Keswick Dam, the Feather River below Fish Barrier Dam, the Yuba River below Daguerre Point dam, and the San Joaquin River delta.

Additional protective regulations prohibiting the take of GS (the "4(d) rule") were passed in 2010. The 2010 recovery plan outline provides a list of potential conservation actions that may promote species recovery. These include actions aimed at improving scientific understanding of the GS life cycle and population biology, improving collaborations between recovery partners, and evaluating and preventing GS bycatch. The five-year review conducted in 2015 found little change to the GS status or threats. However, one potential threat was eliminated with the removal of the Red Bluff Diversion Dam (NMFS 2015c). The recovery plan for Southern DPS GS is still being developed. Once completed, the recovery plan will specify and rate the importance of recovery and research activities.

Data Gaps and Opportunities for Applying Economic Analysis

Scientific understanding of GS population biology and the relationships between threats and population dynamics is lacking. A first order of business in developing effective conservation and recovery plans is generating this understanding. Specifically of need are studies that contribute to population-scale understanding of GS biology. Population-scale studies require population-level sampling programs coordinated through interagency and interstate cooperation. Economic evaluation of recovery alternatives is limited by the dearth of scientific research on GS. Still, cost-effectiveness analysis may prove useful for evaluating alternative data collection or recovery plan strategies.

Black and White Abalone

Eight species of abalone are found off the California coast.[14] Five of these species—black, white, green, red, and pink—supported popular recreational and commercial fisheries until populations experienced severe declines during the second half of the 20th century. Two species, white and black abalone, are listed as endangered under the ESA. Three additional species—pinto, green, and pink abalone—are designated as species of concern. Red abalone populations, which support an economically important recreational fishery, are also in decline.[15] This section reviews the status of listed white and black abalone populations and the threats they face. It also reviews the policy instruments being deployed to recover populations and the evidence of their effectiveness, as well as opportunities for economic analysis to contribute to abalone conservation and recovery.

Current Status

Black Abalone

Historically, black abalone inhabited Pacific waters from Crescent City, California, to southern Baja California. Black abalone is the most shallow-living abalone (NMFS 2016b). They are generally found on intertidal rocky shores in coastal or offshore island areas. The most abundant black abalone populations are south of Monterey Bay. Black abalone were extremely abundant in the Channel Islands until 1986, before large-scale population declines began (NMFS 2009). The decline in black abalone is attributed to overfishing and the onset of withering syndrome, a usually fatal bacterial disease that gradually weakens the organism and results in a loss of ability to maintain adhesion to rocky substrata. Upon petition, black abalone were listed as endangered in 2009, and critical habitat was designated in 2011.

White Abalone

White abalone inhabit coastal habitats on the west coast from Point Conception, California, to Punta Abreojos, Mexico. The deepest west coast species of abalone, white abalone are usually reported at depths of 20–60 meters, with the highest densities between 25–30 m (Hobday and Tegner 2000). Historically, more than 360,000 legal-sized white abalone are estimated to have occupied California's coastal waters (Rogers-Bennett et al. 2002).

The height of white abalone exploitation in California spanned only a decade. The fishery began in earnest in 1968, and harvest was in rapid decline by 1978 (Hobday and Tegner 2000). In just ten years, overutilization by commercial and recreational fisheries led to drastic declines in white abalone populations. By the time the white abalone fishery was permanently closed in 1996, densities had fallen sufficiently so that white abalone populations were no longer self-sustaining. Specifically, the population declines decreased abalone density (i.e., increased the

[14] Abalone species: black (*Haliotis cracherodii*), white (*H. sorenseni*), green (*H. fulgens*), red (*H. rufescens*), pink (*H. corrugate*), pinto (*H. kamtschatkana*), flat (*H. walallensis*), and threaded (*H. assimilis*).

[15] The kelp forests where they live have been decimated by warming waters and purple sea urchins, so there is no food for the red abalone to eat. All recreational harvest of red abalone was closed in California in 2018.

distance between individuals) to the point where reproduction (successful fertilization of the broadcast gametes) was no longer possible. At their most abundant site in Southern California, the estimated white abalone population declined from 15,000 individuals in 2002 to 3,000 in 2010 (Stierhoff et al. 2012). The ESA status review in 2000 estimated that white abalone in the wild represented less than 0.1% of their estimated pre-fishery population size. The remaining wild individuals are aging, and the entire species will likely go extinct from natural mortality within 20 years without active conservation in the form of captive breeding and stocking (Catton et al. 2016). Following the status review, white abalone were listed as endangered in 2001, making them the first marine invertebrate listed as endangered under the ESA.

Assessment and Evaluation of Threats

Black Abalone

The most significant current and future threat to black abalone is withering syndrome, a bacterial disease that disrupts the abalone's ability to digest food, causing eventual starvation (Neuman et al. 2010). Other important anthropogenic threats include illegal take, ocean warming, and ocean acidification (NMFS 2016b). Withering syndrome was first detected in 1985 and has been decimating black abalone populations since. Infection of individuals in a population leads to rapid population declines of 90 percent or more. The disease is less prevalent in the northern part of the black abalone range, where water temperatures are lower. For example, eight of the nine northern populations (i.e., north of Cayucos, California) under monitoring showed no declines in abundance between 1975 and 2006. The 2009 status review team identified the risks associated with withering syndrome as the primary cause for concern about the persistence of black abalone as a species (NMFS 2009). Furthermore, the status review team unanimously agreed that black abalone are likely to go extinct in the coming decades unless effective measures are developed to combat the population-level effects of withering disease (NMFS 2009).

White Abalone

By the time white abalone were listed in 2010, overutilization had reduced population abundance and genetic diversity to levels where the populations were no longer self-sustaining, even if other threats were to be addressed. The white abalone recovery plan found that low abalone density (i.e., the number of abalone per square meter or square hectare) was the greatest threat facing white abalone on the west coast. Low density results in a failure to successfully reproduce, since broadcast eggs and sperm, released during spawning events, are too far apart to achieve successful fertilization. Other threats identified by the recovery plan (NMFS 2008a), listed in order of severity, include:

1. Inability to implement conservation and research efforts.
2. Inadequate enforcement.
3. Reduced genetic diversity, which leads to lower reproductive potential and fitness.
4. Spread of disease through supplementation.
5. Illegal harvesting.
6. Habitat modification from anthropogenic activities.

Of note, while habitat destruction/modification and disease were not significant drivers of the initial white abalone population decline (Hobday and Tegner 2000), these factors may influence the recovery outlook for white abalone.

Instruments of Recovery

Black Abalone

In response to observed population declines, all forms of legal black abalone harvest were suspended by state authorities in California in 1993 (NMFS 2009). Neuman et al. (2010) identified characteristics of recovery actions that are most likely to conserve and recover black abalone populations. These characteristics include: 1) actions that buffer black abalone from the anthropogenic sources of increased sea surface temperature (e.g., thermal effluent) and other conditions that promote the spread of withering syndrome, 2) actions that reduce illegal take of black abalone, and 3) human interventions (e.g., translocations, captive propagation) that promote population rebuilding with safe and cost-effective methods.

The black abalone recovery plan is still under development, but major recovery needs include:

- Mapping black abalone habitats and black abalone population densities.
- Establishing monitoring of populations in Baja California.
- Minimizing the potential for pathogen and invasive species introductions through regulations, inspections, and other instruments.
- Developing an oil-spill response plan for black abalone conservation.
- Collaborating with California state authorities to develop poaching deterrents.

White Abalone

The white abalone fishery was closed indefinitely in 1996, followed by the closure of all abalone fisheries in Central and Southern California in 1997. However, active conservation measures, including captive breeding, are required to prevent extinction of white abalone in the short run. Recovery plans drafted by NMFS and the state of California both identified captive breeding and stocking as necessary recovery actions to prevent extinction (CDFG 2005, NMFS 2008a). Wild population surveys and education and outreach are two additional recovery activities currently underway. As of 2015, the captive breeding and stocking program had produced enough small abalone to conduct stocking trials in the ocean. Experiments with red abalone are being used to inform and optimize strategies for white abalone stocking. The stocking trials and modeling conducted so far have yielded useful information. Catton et al. (2016) found that a juvenile stocking density of greater than 0.23 abalone/m^2 resulted in positive population growth over a 20-year period. Li and Rogers-Bennett (2017) found that successful stockings tended to supplement resident populations when the stocked abalone were larger. Ocean warming and poaching are associated with less successful stockings.

Data Gaps and Opportunities for Applying Economic Analysis

ESA-listed black and white abalone face very different threats. Black abalone are at risk primarily from a marine pathogen that has rapidly decimated populations in Southern California. Conversely, overfishing reduced white abalone populations to densities where the species can no longer effectively reproduce. Nonetheless, the conservation challenges and opportunities associated with white abalone have similarities to those associated with black abalone. Both species are relatively stationary broadcast spawners that face threats from illegal harvest and whose recovery prospects are inhibited by warming sea waters. Likewise, three additional abalone species in California are designated as species of concern: pinto, green and pink abalone.

The preceding review of black abalone and white abalone revealed several opportunities for applying economic analysis to inform abalone recovery and conservation. Economic analysis can contribute to evaluation of alternative abalone recovery actions using cost–benefit analysis and cost-effectiveness analysis. However, recovery action evaluation with economic analysis requires an understanding of the relationship between alternative actions and recovery outcomes. CEA may also contribute to prioritization of data collection, modeling, and monitoring alternatives based on their potential to reduce uncertainty and improve recovery decision-making. Recovery strategies for for black abalone involve managing the spread of a marine disease with monitoring and active population control measures, while recovery strategies for white abalone involve captive breeding and stocking. Bioeconomic modeling and dynamic simulation methods have promise for informing recovery planning in these contexts. For abalone populations under current or future exploitation, economics offers some theory and observation on alternative systems for assigning property rights. Notably, for sedentary species such as abalone, territorial use rights, where users are permitted to harvest from a specific area, may offer potential for sustainably managing abalone fisheries (see Wilen et al. 2012).

Finally, abalone protections and conservation objectives may conflict with or enhance other federal conservation policies or objectives. For example, protected sea otters (*Enhydra lutris*) prey upon black abalone populations in California, but the presence of sea otters is associated with improved black abalone population increases (Raimondi et al. 2015). Such situations present an opportunity for applying institutional analysis to explore a more holistic, ecosystem-based management strategy for managing co-occurring protected marine species.

Marine Sea Turtles

Current Status

Subpopulations or DPSs of four sea turtle species range into coastal and offshore waters of the U.S. West Coast: leatherback (*Dermochelys coriacea*), loggerhead (*Caretta caretta*), olive ridley (*Lepidochelys olivacea*), and green sea turtles (*Chelonia mydas*). These species experienced significant population declines in the late 20th century and have been listed under the ESA since 1978. Population declines have been noted for all species over the last several decades. However, with the implementation of conservation measures and recovery plans, there is evidence of rebuilding nesting populations in the U.S. and around the world.

Abundance estimates and trends for each species subpopulation are largely based on the number of nesting females that come ashore each year, and therefore vary considerably across locations and years. Western and eastern Pacific leatherback populations have shown declines of 80% and 97% since the 1980s, respectively (Santidrián Tomillo et al. 2007, Tapilatu et al. 2013). Nesting loggerhead females in the North Pacific DPS declined throughout the 1990s, increased steadily until 2005, declined again in 2006, and have been on the rise again in recent years. In 2004, one estimate stated that green sea turtles had likely declined 48–67% globally over the last three generations (Seminoff 2004). However, nesting for the East Pacific green sea turtle DPS has exhibited substantial increases since 1996 at the primary nesting site in Mexico.

All sea turtle species make long migrations between nesting beaches and foraging grounds, often across entire ocean basins. Though there are no nesting beaches on the U.S. West Coast, sea turtles migrate to the area to forage throughout the year. Leatherbacks and loggerheads are found foraging off the west coast from British Columbia to Southern California, though they are often concentrated in the productive waters of the California Current Ecosystem. Leatherbacks that occur in west coast waters originate from nesting beaches across the western Pacific (in Indonesia, Papua New Guinea, and the Solomon Islands). Loggerheads that forage in west coast waters nest almost exclusively in Japan. Olive ridley sea turtles are found from Mexico to northern Chile, with nesting occurring from Mexico to Peru. They are the only species in the eastern Pacific which undertake *arribada* (i.e., nesting en masse), which occurs on several beaches in both Costa Rica and Mexico. Green sea turtles found foraging in California waters are part of the East Pacific DPS, originating from nesting beaches in Mexico.

Assessment and Evaluation of Threats

Sea turtle populations are impacted by anthropogenic activities in both their terrestrial and marine habitats. Nesting beaches are a critical component of the sea turtle life cycle. Available nesting beach habitat has declined due to coastal development, pollution, beach erosion, and habitat changes from climate change. Rising egg incubation temperatures can also lead to the feminization of populations due to the Temperature Sex Determination phenomenon that results in more female hatchlings than male. In the marine environment, major threats to sea turtles include entanglement in fishing gear, ingestion of marine debris, and vessel ship strikes.

For turtles that inhabit U.S. West Coast waters at some point during their life cycle, the most significant threats they face can include bycatch in commercial and artisanal fisheries, climate change, ingestion of marine debris, vessel strikes, illegal harvest, and limited enforcement capacity for conservation measures in developing countries where these turtles nest.

Bycatch remains one of the most significant threats to sea turtles on the west coast (Lewison and Crowder 2007, Donlan et al. 2010, Finkbeiner et al. 2011). In general, longline fisheries are thought to be particularly impactful due to their tendency to catch older age classes with higher reproductive potential (Lewison and Crowder 2007), though sea turtle bycatch in trawls and gillnets has also been a long-standing concern. For North Pacific loggerheads, the most significant threat is bycatch in coastal pound nets off Japan and on bottom longlines in Baja California, with mortality estimates reaching 1,000 per year (Peckham et al. 2007, Conant et al. 2009). Similarly, bycatch remains a significant threat to olive ridley turtles in the Pacific, with estimates from the 1990s of more than 13,000 caught annually in shrimp trawls off Costa Rica being just one example (USFWS 2014). For the eastern Pacific DPS of green sea turtles, bycatch is a primary concern in small-scale gillnet fisheries in Baja California (estimated at 1,000 sea turtles per year in the mid-2000s) and artisanal longline fisheries operating off Ecuador and Chile (ranging from 0.36 to 0.60 sea turtles per 1,000 hooks; Seminoff et al. 2015).

Climate change remains a significant threat not only to nesting beaches but also by changing the marine habitat ecosystems on which sea turtles depend, with turtles' ability to adapt to changes in prey distribution, currents, productivity, etc., varying by species. Researchers have begun incorporating climate-related impacts into predictive modeling for fisheries outcomes (Pinsky and Mantua 2014), but have generally not yet extended their application to protected species (Komoroske and Lewison 2015). Hazen et al. (2013) examined the shift in habitat of top predators, including sea turtles, based on predicted changes in sea surface temperatures. Only recently have researchers been able to improve predictions about the spatiotemporal distribution of sea turtles in specific habitat areas based on environmental variables (Eguchi et al. 2017), which may assist in understanding how climate change might shift sea turtle distribution and overlap with fishing effort in the future. However, the rare nature of sea turtle bycatch events makes it difficult to mathematically model the socioeconomic and biological outcomes of various policy options aimed at minimizing entanglements and bycatch mortality. In terms of terrestrial habitat, model projections estimate the loss of 43% of loggerhead nesting habitat in Florida and 3–65% of green sea turtle nesting habitat in the Hawaiian Islands by 2100 (Baker et al. 2006, Reece et al. 2013).

Several sources of uncertainty make it challenging to assess and efficiently minimize or mitigate the impacts of anthropogenic activities on sea turtles. First, interannual variability in abundance estimates can arise naturally from inherent population dynamics or fluctuations in ecosystem productivity, but also from changes in beach monitoring or disturbance events such as oil spills. Using variable data to establish baseline conditions for impact analyses or recovery planning introduces inevitable uncertainty, where meaningful analyses would require conducting studies on a decadal scale.

Second, gaps in research knowledge about the distribution of sea turtles at specific life stages introduce uncertainty when modeling the overlap of sea turtles, their habitat, and anthropogenic activities. Predicting and minimizing the overlap and impact of anthropogenic activities is nearly impossible without a more precise understanding of how and when sea turtles use coastal and offshore habitat areas in each life stage.

Third, the nature and severity of certain threats remain unknown. An expert opinion survey ranking threats to sea turtles consistently identified bycatch as the most significant threat to populations around the world, though the relative severity of bycatch and compared to other threats varied greatly by region (Donlan et al. 2010). The survey results also identified direct take and pollution as problematic for loggerhead, olive ridley, and green sea turtles in the Pacific (Donlan et al. 2010). The Hawaii pelagic longline fishery has been cited as having higher mortality estimates compared to California gillnet and longline fisheries (Finkbeiner et al. 2011), though this is likely an artifact of local sea turtle abundance rather than the fisheries themselves.

There is a wide range of entities with a stake in sea turtle conservation and management. These include fishermen utilizing or passing through sea turtle habitat, wildlife tourism, beachgoers, community businesses near nesting beaches, and the general public. The goal of sea turtle management is to enact policies that yield net benefits to the public and coastal communities, and that promote the recovery of endangered species. Natural resource management decision-making is complex and requires comparing policy alternatives with imperfect or nonexistent data in perpetually changing conditions within the fishery or socioeconomic landscape. These tensions between resource use and conservation will likely persist into the future due to the overlapping nature of fisheries and sea turtles, and the inevitable economic trade-offs of reducing bycatch. Resource managers will continue to implement regulations based on the best available data, minimizing uncertainty and reducing the likelihood of bycatch.

Instruments of Recovery

In the marine environment, NOAA Fisheries has implemented a wide range of regulatory measures to reduce incidental take of sea turtles in fisheries through fishery management plans, biological opinions, and recovery plans under the ESA. These include top-down control measures such as spatial and temporal restrictions, gear requirements, gear modifications, and bycatch limits. All of these strategies for reducing sea turtle bycatch have a range of costs (either realized or opportunity) associated with them.

Restricting certain gear types or closing areas to fishing at certain times of the year are management techniques used to reduce sea turtle bycatch in a number of fisheries in the U.S. and around the world. In 2001, NOAA Fisheries implemented the Pacific Leatherback Conservation Area Closure as a time–area closure from mid-August to mid-November off California and Oregon to reduce bycatch in the U.S. swordfish fishery. NOAA Fisheries also implemented the Pacific Loggerhead Conservation Area, 25,000 square miles that close to gillnet fishing when temperatures rise above a certain threshold, often coinciding with El Niño conditions. Similarly, time and area closures have been implemented in the Atlantic sea scallop fishery and in the North Carolina inshore gillnet fishery when sea turtle interactions reach established limits. These spatial closures can be dynamic in nature, where areas are closed based on when sea turtles are known to be present, or if a higher number of interactions are reported. However, the effectiveness could likely be improved with real-time or near-real-time reporting, allowing fishermen to proactively (rather than in the next season) avoid bycatch hotspots (e.g., the TurtleWatch program in Hawaii, described below).

Gear modifications to reduce sea turtle bycatch have been developed for trawl, gillnet, and longline gears. In some instances, reducing effort (via reducing the number of hooks per set or gillnet soak times) is the primary goal. In other cases, the actual gear itself is changed to reduce the likelihood of entanglement. For trawls, Turtle Excluder Devices (TEDs) have been proven effective in excluding sea turtles from nets in the U.S. when installed in compliance with regulations, though bycatch reduction targets have not been met in all instances (Brewer et al. 2006, Cox et al. 2007, NMFS 2014b). For gillnets, researchers in Baja California have shown that both illuminated gillnets (Wang et al. 2010, 2013) and buoyless gillnets (Peckham et al. 2016) significantly reduce bycatch without reducing the volume of target species landings. For pelagic longline fisheries, particularly shallow-sets targeting swordfish, the transition to using wider circle hooks with fish bait (compared to the traditional approach of using J-hooks with squid bait) reduced sea turtle bycatch and mortality rates by more than 80% in some cases (Watson et al. 2005, Gilman et al. 2007).

Gear modifications in the Hawaii pelagic longline fishery were accompanied by a fleet-wide bycatch limit on the number of leatherback and loggerhead sea turtle mortalities each year. If the limit is reached, the fishery is shut down for the remainder of the year. These measures, implemented under the ESA biological opinion authorizing the continued operation of the fishery, have remained controversial and have been the subject of litigation and ongoing study. Researchers examined the broader impacts of these regulations, intended to reduce impacts on sea turtles, and concluded that, in fact, bycatch rate could be 11% higher due to regulatory spillover effects. By temporarily shutting down the U.S. pelagic longline swordfish fishery in the central Pacific, foreign fisheries operating elsewhere in the Pacific with little to no bycatch reduction measures ramped up production to meet demand within the U.S. Researchers estimated that bycatch would have been more than 80% lower if foreign fisheries had maintained bycatch rates as low as the U.S. fishery (Chan and Pan 2016). This situation illustrates one of the challenges of implementing effective policies to promote sea turtle conservation and management.

As illustrated by the longline swordfish closure described above, the Hawaii pelagic longline fishery is a prime example of the complexity in modeling and predicting economic trade-offs and biological outcomes. Researchers estimate that based on current bycatch limits and accompanying closure regulations, the shadow price[16] of reducing bycatch ranged from an average of over $30,000 up to more than $50,000 per sea turtle throughout the 1990s, depending on model assumptions and scenarios (Curtis and Hicks 2000, Huang and Leung 2007). Due to the low bycatch limits and the high costs of reducing bycatch in this fishery, researchers have examined the potential economic and conservation benefits of instituting a cap-and-trade framework, where gillnet fishermen in Baja California would relinquish some of their bycatch quota to Hawaii longliners in exchange for compensation dedicated to improving their gear and thereby further reducing sea turtle bycatch.

A cap-and-trade framework has also been discussed as a viable option for reducing bycatch in the Gulf of Mexico reef fish fishery, where researchers note that a socially optimal management plan can be implemented only when bycatch is observable and quota trades are relatively seamless and low in cost (Singh and Weninger 2015).

[16] In this context, the shadow price is the economic value to fishermen of relaxing bycatch regulations.

Data Gaps and Opportunities for Applying Economic Analysis

Potentially promising policy options to reduce sea turtle bycatch that are not in widespread use include quota risk pools, move-on rules with hotspot reporting, compensatory mitigation approaches such as bycatch and landings taxes, and adaptive measures based on prevailing oceanographic indices or thresholds. These mechanisms can be seen as incentive-based rather than traditional top-down regulatory approaches (Lent and Squires 2017), and could also be used together, particularly because the options available to different fisheries vary based on gear type, fishing location, target species, and the level of economic development where the fishery is prosecuted. Additionally, studies that scale up and aggregate impacts across gear types, fishing fleets, and regions (Wallace et al. 2008, 2010, 2013, Lewison et al. 2014) are critical to accurately estimate mortality and develop economically efficient mitigation strategies for migratory species such as sea turtles.

Because sea turtles migrate across ocean basins during different life stages and encounter a variety of threats on nesting beaches as well as in pelagic ocean ecosystems, particularly in countries with few to no protective measures, determining where conservation dollars and mitigation effort have the highest pay-off is important. For example, Pacific leatherback nesting beach conservation efforts in Indonesia have been shown to be less costly per adult female than instituting and maintaining Hawaii pelagic longline regulations and time–area closures in the California drift gillnet fishery by ten and 100 times, respectively (Gjertsen 2011, Gjertsen et al. 2014). In this case, addressing the primary threat may not be the most economically viable option relative to the costs when another strategy may provide equivalent benefits. Protecting the turtles during such a reproductively valuable time period is more effective, beneficial, and economically efficient than further restricting industry activities in the U.S., where protective measures such as circle hooks or TEDs are already in place. The benefits of investing in costly fisheries closures or gear changes can be nullified if sea turtle mortality is high elsewhere, highlighting the importance of compensatory mitigation (financed by developed nations) and holistic conservation strategies (Dutton and Squires 2011, Gjertsen 2011, Gjertsen et al. 2014). Notably, these policies also have distributional implications, as the costs and benefits are borne by different parties under the alternative regulations.

Bycatch taxes could be effectively implemented at the fleet level, where participants are either penalized or refunded if the limit is exceeded or not, thus incentivizing bycatch avoidance behaviors that may be costly in terms of gear changes or foregone target species catch (Segerson 2011). Bycatch taxes can be set at a fixed percentage of landings, or can be established on a per-individual basis. Assuming the price of bycatch is right, the high incentive to avoid and reduce bycatch with this framework makes it particularly promising. However, the approach can become problematic if the price is set too low (Abbott and Wilen 2009), as the penalty from exceeding the bycatch limit would be insufficient disincentive to reduce bycatch.

Bycatch caps have been found to be most effective when implemented in conjunction with other strategies such as gear modifications and/or fleet communication (O'Keefe et al. 2013). Allocating bycatch quotas to fishery participants and allowing for trade can result in greater economic efficiency, because individuals can optimize cost reduction and maintain optimal harvests (O'Keefe et al. 2013). In general, adaptive compensatory mitigation approaches such as these, that are informed by bioeconomic modeling, can help ascertain which conservation measures are both most cost-effective and have the highest conservation benefits (Wilcox and Donlan 2007).

The formation of risk pools allows fishermen to "pool" their quota for constraining or bycatch species and then access it if and when bycatch events occur, and is equivalent to purchasing insurance. Risk pools are most successful when bycatch is uncertain and the risk of exceeding quota is relatively high. This approach is used by some participants in the U.S. West Coast groundfish fishery (Holland 2010, Holland and Jannot 2012), where members have experienced increased utilization rates of constraining species (Kauer and Oberhoff 2015). If nontarget species are constraining and observer coverage is high, this framework could be applied to bycatch limits established for protected species. This approach is promising because it involves sharing risk among the members, which also encourages cooperation and enhanced accountability (unless the incentives are somehow skewed to encourage cheating, which is a weakness of this approach that requires attention during the developmental period).

Agent-based modeling, though not widely used, can incorporate fishing behavior into models to predict spatial and temporal patterns in fishing effort and the resulting economic benefits. Yu et al. (2009) developed this approach for the Hawaii pelagic longline fishery, enabling examination of fishing outcomes across policy alternatives. This could be adapted to include the costs of avoiding bycatch.

Lastly, another possible mechanism to reduce bycatch would be to implement management measures based on specific prevailing oceanographic conditions or indices that coincide with the presence or absence of forage species, the presence of sea turtles in foraging areas, specific migratory patterns, or the presence of particularly vulnerable age classes of bycatch species. Sea turtle movements and behaviors have been shown to follow patterns in convergence zones, where productivity and forage species are also common (Polovina et al. 2000, 2004). Formally incorporating this knowledge into management measures could lower the cost of bycatch avoidance because potentially restrictive measures would be limited to specific times and places with the highest likelihood of interactions (Hobday et al. 2013, Lewison et al. 2015, Maxwell et al. 2015).

Beginning in the late 2000s, this approach was employed to reduce loggerhead bycatch in the Hawaii pelagic longline fishery through the TurtleWatch program (Howell et al. 2015). Suggested areas to avoid are based on temperature isotherms and have most recently been expanded to include information about leatherback habitat areas (Howell et al. 2015). This type of adaptive EBM tool will be particularly useful if and when populations recover (making fisheries interactions more likely) or shift their ranges in response to changing ocean conditions (making static conservation areas potentially less effective). The quality and quantity of bycatch data available for assessing the impact of fisheries on sea turtles continue to improve. With enhanced spatial predictive mapping of bycatch hotspots for marine megafauna (Lewison et al. 2009, Žydelis et al. 2011, Lewison et al. 2014) and refined Bayesian modeling (Martin et al. 2015), implementing adaptive management to reduce sea turtle bycatch will become increasingly feasible.

As discussed above, managers will likely never have the perfect dataset that allows them to identify the precise marginal costs and benefits of various policy options, but they have to make decisions and implement regulations nonetheless. This dilemma highlights the importance of prioritizing policies that incentivize continued cooperative industry research and regulatory decisions that are adaptable to future changes. One potentially useful area of research would be determining the flexibility of models to adapt to various fisheries and environments, so that initial measures could be implemented based on knowledge and experience from other fisheries with similar gears or target species without having to garner funding to repeat studies in every fishery prior to implementation.

Pinnipeds

Current Status

Pinnipeds haul out on rookeries for mating and pupping and either make shorter foraging trips or migrate farther distances to foraging grounds. Foraging patterns depend on the species and can also vary by age and sex class. These life history characteristics and often remote rookery locations make population censuses difficult, with abundance estimates generally extrapolated and modeled from counts on rookeries, pup counts, mortality estimates, attendance patterns, and recruitment rates. Six species of pinnipeds live off the U.S. West Coast: harbor seals (*Phoca vitulina*), northern elephant seals (*Mirounga angustirostris*), California sea lions (*Zalophus californianus*), Guadalupe fur seals (*Arctocephalus townsendi*), northern fur seals (*Callorhinus ursinus*), and Steller sea lions (*Eumetopias jubatus*). Guadalupe fur seals and the western stock of Steller sea lions are listed under the ESA, and northern fur seals and the eastern stock of Steller sea lions are listed as depleted under the MMPA. These species are still recovering from commercial hunting during the 18th century that severely depleted populations (Carretta et al. 2016). These six pinniped species and their corresponding regional stocks have different abundances and population trends, and therefore different conservation statuses.

Guadalupe fur seals breed primarily in Baja California, and have recently been expanding into historical habitat in the Pacific Northwest, with populations thought to be increasing at approximately 10% per year (Carretta et al. 2016). Northern fur seals are found from Southern California throughout the North Pacific, with primary breeding sites in the Pribilof Islands. The eastern Pacific stock is listed as depleted under the MMPA, and pup production has continued to decline in recent years, though the trend varies across the primary breeding sites. Steller sea lions range from Northern California across the North Pacific into Japan, and have been an ongoing source of controversy among stakeholders and fishery managers. The eastern stock of Steller sea lions was delisted from the ESA in 2013 and is estimated to include 60–74,000 individuals, increasing approximately 4% per year (Carretta et al. 2016). The western stock, however, remains listed as endangered under the ESA, having experienced dramatic population declines from the 1970s to 2000. The population breeds throughout the Aleutians and into Russia, and pup counts exhibit varying trends depending on the area.

In contrast, harbor seals, northern elephant seals, and California sea lion populations are stable and increasing throughout their ranges and will not be discussed in depth in this review. Harbor seals are a cosmopolitan species found around the world, with separate west coast stocks delineated for California, Oregon/Washington, and inland Washington. Northern elephant seals breed along the California coast and in Baja California, and migrate far north into the Aleutians and central North Pacific to forage. The population is estimated at 179,000 individuals and has a 3.8% growth rate per year (Carretta et al. 2016). California sea lions also breed in Southern California and range up to the Canadian border, with a population estimate of almost 300,000 individuals and a growth rate of 5.4% per year (Carretta et al. 2016). These species are protected under the MMPA, but are not designated depleted or strategic, and therefore generally are not subject to conservation measures such as bycatch take reduction plans or ESA recovery plans and incidental take limits.

Assessment and Evaluation of Threats

Threats to ESA-listed pinnipeds on the west coast include being entangled in active or derelict fishing gear or marine debris, competition for prey resources with commercial fisheries, and shifting prey distributions related to ongoing climate variability and future long-term changes. Direct fisheries-related mortality estimates vary considerably depending on the abundance of the pinniped population and its overlap with fishing activities. For the eastern stock of Steller sea lions, approximately 14 individuals were estimated to have been killed per year in groundfish bottom and midwater trawl fisheries along the U.S. West Coast from 2010–13 (Carretta et al. 2016). An additional 15 individuals per year from 2012–14 were reported to the stranding network with gunshot wounds (Carretta et al. 2016). Fishing-related mortality estimates are extrapolated from fisheries observer data and or stranding data, but large degrees of uncertainty remain due to the challenges of scaling up and extrapolating from rare and patchily distributed occurrences. Mathematical uncertainty arises when considering the abundance of and anthropogenic impacts on populations that span international borders, where data collection might differ from that in the United States or be entirely lacking.

Derelict fishing gear is responsible for a large proportion of marine mammal bycatch. Derelict gear and packing bands (often used to secure bait boxes) have been documented as accounting for approximately half of all western Steller sea lion entanglement cases in Alaska and northern British Columbia (Raum-Suryan et al. 2009), and are therefore likely problematic for the eastern DPS as well. The impacts of marine debris and pollution greatly depend on the spatial overlap of high-density debris areas with important Steller sea lion habitat or life history stages. For example, Steller sea lions in British Columbia had the highest likelihood of overlapping with areas of debris near a large rookery (Williams et al. 2011).

Fisheries that target pinniped prey species may also have indirect impacts on populations. For Steller sea lions, particularly the western DPS, nutritional stress has been an ongoing concern and source of debate amongst scientists, managers, and the fishing industry. Since the early 2000s, significant research has been dedicated to investigating the potential correlation between declining Steller sea lion abundance, low reproductive success, and lower prey resource availability from both environmental variability and depletion by commercial fisheries (Rosen and Trites 2000, NRC 2003, Fritz and Hinckley 2005, Rosen and Trites 2005, Guénette et al. 2006, Trites et al. 2007, Atkinson et al. 2008, Österblom et al. 2008, Sigler et al. 2009, Goundie et al. 2015). Researchers have also made the distinction between anthropogenic threats likely contributing to the decline of the species since the 1960s versus those anthropogenic activities that may or may not actually be continuing to inhibit recovery (Atkinson et al. 2008). For example, Dillingham et al. (2006) noted that the estimated change in population growth with the elimination of all groundfish trawl fishing in the area would be small. This idea of depleted prey from competition with fisheries and the so-called "Junk Food Hypothesis" earned media attention and speculation fueled by scientific uncertainty, leading to strategic framing and blaming by various stakeholders that was ultimately unhelpful in guiding new research and management approaches (Mansfield and Haas 2006). This situation exemplifies the complexity of enacting effective EBM in an environment marked by short- and long-term oceanographic oscillations in addition to increasing climatic warming.

Pollution and disease affect all west coast pinnipeds, though the nature and severity of the impacts remain unknown. Naturally occurring bacteria, viruses, and algae can impact pinnipeds in a variety of ways during harmful algal blooms or through ingesting infected prey. Both natural toxins such as domoic acid and man-made chemical pollutants such as PCBs can become concentrated in the tissues and blubber of top predators due to bioaccumulation up the food web. Research suggests that these chemicals may contribute to issues ranging from reduced reproductive success or resilience against disease to increased prevalence of herpes or cancer, and may even have amplified or additive effects when agents act in conjunction with one another (Ylitalo et al. 2005, Buckles et al. 2006, Ross et al. 2013). Contaminant levels documented in the eastern Steller sea lion DPS are similar to those documented for Southern Resident killer whales but lower than those in the western Steller sea lion DPS and other pinnipeds in California and the Salish Sea (Cullon et al. 2005, Alava et al. 2012). Although levels of legacy contaminants such as PCBs and DDTs have been declining in harbor seals and Steller sea lions in the Pacific northwest, these species are also likely impacted by numerous other contaminants that are poorly understood and not monitored (Barron et al. 2003, Ross et al. 2013).

In addition to the above known and monitored threats, unknown agents can impact populations through what are termed "Unusual Mortality Events." Since 1991, there have been 16 of these occurrences on the U.S. West Coast for cetaceans and pinnipeds. Most recently, elevated numbers of California sea lion pups (2013–16) and Guadalupe fur seal pups (2015–17) stranded along the California coast. The start of this elevated stranding coincided with El Niño conditions along the coast, which caused poorer pup body condition and increased maternal foraging effort for both California sea lions and Guadalupe fur seals (Elorriaga-Verplancken et al. 2016). However, the cause of these events often remains unknown, highlighting the complex nature of studying and monitoring wildlife health in the marine environment. Research suggests that land-based pollution and the frequency of harmful algal blooms may be increasing on the west coast (Lewitus et al. 2012). The population-level and long-term impacts of these threats remain unknown, especially how they might be exacerbated by shifting oceanographic conditions or ocean warming. These uncertainties will likely persist into the future due to the difficulty of understanding the cumulative and potentially amplifying impacts of multiple pollutants and stressors on individuals, and how that can be scaled up to population-level impacts.

Instruments of Recovery

Cetaceans and pinnipeds are managed and protected under the Marine Mammal Protection Act (1972) and the Endangered Species Act (1973). These acts were designed to foster the collection of scientific data, minimize anthropogenic impacts to the extent practical, and provide for the recovery of depleted species. These laws require stock assessment reports, conservation and recovery plans, and the designation of critical habitat, though implementing regulations can be a slow and controversial process, making it challenging to react quickly enough to minimize impacts from a new or growing threat. While the laws have provisions that allow for establishing Take Reduction Teams or international trade sanctions against nations that do not implement basic safeguards for marine mammals, the implementation of effective regulations has been slow, and we do not often have the capacity, or even the knowledge about the magnitude of impacts, needed to enforce them. Additionally, the efficacy of provisions in the MMPA is highly context-dependent, proving successful in some circumstances but not others (Geijer and Read 2013).

Management and conservation mechanisms for reducing the impacts of anthropogenic activities on ESA-listed pinnipeds include: a) designating critical habitat, b) reducing bycatch through gear changes, area closures, acoustic deterrents, or incentive-based compensation mitigation, c) reducing the entangling nature of marine debris and packing bands, and d) improving the use and implementation of ecosystem-based fisheries management (EBFM).

Although no critical habitat has been designated for Guadalupe fur seals to date, upon delisting of the eastern Steller sea lion DPS, NOAA Fisheries developed a monitoring plan that would guide recovery and conservation measures. Steller sea lion critical habitat has been designated on rookery islands along the coasts of California and Oregon, though the most important aspect of their critical habitat is maintaining a sufficient prey base (NMFS 2013).

Regulatory mechanisms exist for reducing direct mortality from capture in active fishing gear, including gear modifications such as changing mesh size or requiring pingers or other acoustic deterrents. Similar to TEDs for sea turtles, acoustic bycatch deterrents can be effective when installed correctly, though ongoing research is needed in each unique circumstance to ensure that animals are not becoming habituated or attracted to the sounds (Cox et al. 2007). Requirements to minimize anthropogenic impacts such as bycatch are outlined in biological opinions under the ESA. As Guadalupe fur seal abundance continues to increase, bycatch could become more problematic. Bycatch estimates exist for Steller sea lions from observed fisheries, though less is known about bycatch in recreational fisheries or derelict gear, which are not covered by biological opinions. The only mechanism in this case would be documenting injuries and mortalities in stock assessment reports and ensuring that numbers do not exceed potential biological removal levels for stocks with a depleted status under the MMPA. Even if injuries and mortalities did exceed this level, management changes can be slow to address the issue.

While regulatory mechanisms exist to minimize the impacts of bycatch, only voluntary guidelines are in place for reducing the amount of marine debris in the ocean and reducing the entangling nature of debris. Lose the Loop is an outreach program that encourages best practices for gear disposal and minimizing ocean debris. Researchers are also investigating the feasibility of using alternative materials in the production of plastic packing bands that would degrade much faster when exposed to oxygen in the ocean (Hogan and Warlick 2017).

Data Gaps and Opportunities for Applying Economic Analysis

The primary gaps and opportunities for future research include understanding if and to what degree improving bycatch estimates is cost-effective, conducting cost-effectiveness analyses for existing bycatch mitigation technologies, and implementing dynamic ocean management (DOM) and EBFM that incorporate prey depletion by commercial fisheries and stable top predators such as seals and sea lions. As many pinniped populations continue to grow, conflicts and competing priorities will continue to plague managers working to recover ESA-listed salmonid species while considering competitive consumption by pinnipeds, cetaceans, and fishermen.

Improving estimates of serious injury and mortality due to bycatch would require raising fisheries observer coverage, developing regionally specific abundance estimates, and/or developing predictive models for the occurrence of bycatch events based on varying oceanographic conditions (upwelling, surface temperature, etc.), prey availability, and interannual changes in abundance or pup production. All of these may prove costlier or more time-intensive than the potential benefits of that improved precision, particularly in cases where affordable solutions already exist for minimizing impacts (such as gear modifications). It would likely be beneficial to invest in improved bycatch estimates for modeling the population-level impacts of bycatch on endangered and threatened populations or transboundary stocks. Without having a more complete picture of bycatch estimates across a species' range, it is difficult to evaluate U.S. bycatch and mitigation measures in concert with other fisheries in the species' range. Knowledge of bycatch reduction costs and their impacts on protected species would facilitate the development of more efficient and effective bycatch management policies.

Incentive-based mechanisms, such as tradeable bycatch quota or bycatch taxes, rather than traditional top-down regulatory approaches that reduce effort, could increase flexibility for fishermen on how and when to invest in bycatch avoidance (Lent and Squires 2017). This flexibility could provide fishermen the opportunity to make choices that result in lower costs, higher benefits, and therefore maximum economic efficiency while aiming to meet conservation and management goals. However, a tradeable bycatch quota system would require careful design to consider and include the existence values of marine mammals that are not always shared across various stakeholder groups (Smith et al. 2014), in addition to 100% observer coverage or electronic monitoring.

Research has shown that the potential value derived from target species caught in derelict gear is higher than the cost estimates for clean-up efforts to remove the gear (Good et al. 2009, Antonelis et al. 2011, Maselko et al. 2013, Jeffrey et al. 2016, Scheld et al. 2016). In Puget Sound, 4,000 nets, primarily salmon gillnets, were moved during the 2000s at a cost of several million dollars, prompting the development of best practices guidelines (Gibson 2013), though these voluntary measures are unlikely to adequately reduce the occurrence of lost gear. This topic is an area where cost–benefit analysis (CBA) could make significant contributions to show the total economic costs of derelict gear to fishermen, fish populations, and marine mammals. If these CBA studies were to include the nonmarket value of marine mammals, the benefits would even further exceed the costs of removal efforts. To date, CBA studies for Steller sea lions have used stated preference contingent valuation methods that focus on the implementation of a specific program or management measure, rather than the overall recovery of the population (Lew et al. 2010). Aside from Steller sea lions, very few CBA studies have been conducted on marine mammal conservation and management issues, nor have many studies been conducted on the public's willingness to pay for marine mammal conservation, though existing research does suggest that values vary across species (Wallmo and Lew 2012).

Compensation mitigation alternatives, where fishermen could pay instead for conservation investments that would benefit the species, could prove promising for marine mammals that encounter multiple threats across large geographic ranges. When informed by a cost-effectiveness analysis that identifies the most impactful and cost-effective conservation investments, employing this type of program would be mutually beneficial for pinniped populations and for fishermen looking to maximize net benefits (e.g., establishing a mandatory payment to a gear-removal fund based on salmon landings or revenue). However, one challenge with using bycatch taxes or other compensation mitigation schemes to incentivize bycatch reduction is that there are a) no strict bycatch limits for marine mammal species, and b) low observer coverage in many fisheries. Additionally, applying conservation offsets to marine mammals has been criticized because it does not effectively reduce actual mortality (Doak et al. 2007, Finkelstein et al. 2008, Žydelis et al. 2009). Pascoe et al. (2011) suggest that mitigation offsets could be seen as a short-term strategy while longer-term solutions to actually reduce mortality are being developed.

Similar to sea turtles, implementing dynamic ocean management would be beneficial to ESA-listed pinnipeds and could include a cost-effective combination of gear changes, area closures, or considerations for prey depletion depending on foraging conditions in a given season. DOM could be informed by fisheries-related economic profit modeling (Haynie and Layton 2010) to further hone in on spatially explicit management options that maximize conservation benefits and fishery revenue. EcoCast (2017) is an emerging tool that could be used in adaptive EBFM, where fishermen and other stakeholders can see the predicted spatiotemporal distribution of target and nontarget species. A holistic approach that includes these emerging technologies could be the most likely to lead to economically sound and viable outcomes.

Cetaceans

Current Status

Of more than 20 cetacean species that inhabit waters along the U.S. west coast, seven are listed under the ESA: fin whales (*Balaenoptera physalus*), blue whales (*B. musculus*), sei whales (*B. borealis*), sperm whales (*Physeter macrocephalus*), humpback whales (*Megaptera novaeangliae*), North Pacific right whales (*Eubalaena japonica*), and the Southern Resident killer whales (SRKW). All of these species are under active recovery plans under the ESA and are also listed as strategic stocks under the MMPA.

Fin whales are a cosmopolitan species divided into three stocks in the Pacific, one of which inhabits the waters off the U.S. West Coast. The west coast stock numbers just over 2,500 individuals whose population increased at a rate of 7% per year throughout the late 1990s and closer to 3.5% during the early 2000s (Carretta et al. 2016). Blue whales are another cosmopolitan species found throughout the world's oceans, with approximately half the number existing in the North Pacific compared to pre-whaling population estimates. Individuals from the eastern North Pacific stock migrate to forage off the west coast during summers after spending winters farther south. The most recent stock assessment estimates just over 1,600 individuals in this stock, with the population size possibly not having changed since the 1990s (Carretta et al. 2016). While nine areas off the coast of California have been identified as important feeding areas, the distribution of individuals has likely shifted northward in recent years.

Sei whales are found throughout the world's oceans, with the eastern North Pacific stock inhabiting waters off the U.S. West Coast and as far north as Vancouver Island, British Columbia, Canada, in the summer and fall. Sightings for this species are rare, and therefore much information remains unknown, including population trends. Best available data suggest that the eastern North Pacific stock includes approximately 126 individuals (Carretta et al. 2016). Sei whales are thought to inhabit more temperate latitudes farther offshore. Sperm whales are a cosmopolitan species divided into three Pacific Ocean stocks: Hawaii, North Pacific, and West Coast. Sperm whales are distributed globally; they are found off California throughout the year and in the Pacific Northwest in all seasons except winter (Carretta et al. 2016). The most recent stock assessment estimates just over 2,000 individuals in the west coast stock, likely with a stable population trend (Carretta et al. 2016). However, it remains uncertain whether increased detection in the most recent survey represented increased population abundance or a change in distribution or habitat use along the west coast.

Humpback whales occur in oceans around the world and are divided into subpopulations based on shared breeding areas. In the eastern North Pacific, humpback whales are sighted in two main feeding areas, Central California and northern Washington/British Columbia. Animals from both feeding groups are believed to winter in breeding areas off Central America and Mexico, but animals from the northern Washington feeding area also winter in Hawaii. Just under 2,000 individuals are thought to exist in this feeding area, with the northern Washington group being much smaller (Carretta et al. 2016). Population trend estimates are uncertain, but there is evidence that the populations of these west coast feeding groups have increased variably at an average rate of approximately 7% per year since the 1990s (Carretta et al. 2016).

North Pacific right whales are another population that is small in size, though much less is known about them. Individuals have been sighted throughout the North Pacific, and critical habitat has been identified off the Aleutian Islands in Alaska. It is thought that these animals forage in higher latitudes during the summer and migrate into temperate offshore waters in the winter, though foraging and calving grounds remain unknown. Most recent estimates number the population at just 31 individuals, with no information on population trends (Carretta et al. 2016).

Though killer whales are a cosmopolitan species found around the world, they have been divided into populations and subspecies, with three recognized ecotypes in the North Pacific: transient, offshore, and resident. Southern Resident killer whales (SRKW) were listed under the ESA in 2005 and are now an endangered DPS. SRKW are found along the U.S. West Coast and in inland waters in Washington and British Columbia during the summer. Population estimates have ranged from approximately 70 to 99 individuals since the 1970s, with just 76 individuals in 2017 (Carretta et al. 2016, NMFS 2016c, CWR 2017). The SRKW will be the focus of the remainder of this section on west coast cetaceans, though many of the issues discussed are also relevant to other listed large whales.

Assessment and Evaluation of Threats

Similar to pinnipeds, bycatch, entanglement in marine debris, prey depletion, pollution, and oceanographic shifts due to climate change are threats to cetaceans, with a similar uncertainty in scaling up from rare single occurrences (such as bycatch or ship strikes) to population-level impacts. In addition to these issues, there has been increasing attention and research effort being dedicated to the impacts of anthropogenic underwater noise, whale watching, prey depletion, pollutants, and cumulative impacts on both large and small whales, but particularly the SRKW.

Southern Resident killer whales preferentially consume Chinook salmon, and there is a growing body of evidence correlating whale demographics to salmon abundance (Ford et al. 2010, Ward et al. 2013, Strange 2016), though the precise mechanism of this connection remains unknown (Ward et al. 2013). SRKW health and body condition depend on foraging on a sufficient volume and quality of salmon, which varies across years, seasons, and regions (Williams et al. 2011, Ayres et al. 2012, Hilborn et al. 2012, Ward et al. 2013). For example, the majority of salmon in SRKW diet samples was found to have originated from the Fraser River, where salmon have a higher energy content than those from the Columbia River (Hanson et al. 2010, NMFS 2016c). Research has shown that SRKWs can also rely on coho salmon when Chinook abundance is low (Hilborn et al. 2012, Ford et al. 2016). Fecal hormones suggest that SRKW are nutritionally stressed when salmon availability is low and that physiological stress from vessels may be exacerbated during years with low prey availability (Ayres et al. 2012). Most recently, researchers have associated nutritional stress with the high number of failed pregnancies and suggest that this is likely hindering population recovery (Wasser et al. 2017). The severity of impacts on SRKW from low prey availability is considered high, and the likelihood of improvement would be considered high if ongoing salmon recovery efforts were successful (NMFS 2008b, 2016c, Wasser et al. 2017), though this may not take into account the uncertainty and costs associated with salmon recovery, particularly in a changing climate. Notably, the literature suggests that coordinating salmon and SRKW recoveries may improve recovery effectiveness and efficiency (e.g., Samhouri et al. 2017).

Ocean noise originates from a variety of sources including oil and gas exploration, whale watching vessels, and shipping and ferry traffic, and is recognized as a medium-high threat to SRKW in inland Washington waters, as it may disrupt echolocation, navigation, communication, and foraging behaviors (Erbe 2002, NMFS 2008b, Holt et al. 2009, 2012, NMFS 2016c, Veirs et al. 2016). Efforts are underway to improve our understanding of the magnitude and distribution of the impacts of underwater noise through NOAA's CetSound initiative. It has been determined that vessel speed correlates with the level of noise received by nearby individuals (Houghton et al. 2015). The mere presence of several vessels throughout the SRKW critical habitat causes many behavioral changes, including changes in swimming patterns, increased performance of surface active behaviors, and disruption of foraging behavior (Noren et al. 2009, Williams et al. 2009, New et al. 2015, Senigaglia et al. 2016). Lachmuth et al. (2011) suggest that SRKW exposure to vessel emissions is likely only below levels that would cause adverse health impacts when regulations and best practices are followed. The severity of impacts from the presence of vessels on SRKWs throughout Puget Sound and inland waters in the Pacific Northwest is considered high (NMFS 2008b).

The impacts on SRKW of various contaminants in inland Washington waters is considered high, with only a medium feasibility level for successful mitigation (NMFS 2008b). Likely impacts include endocrine disruption, reproductive failure, and cancer (Mongillo et al. 2016). In a study examining polybrominated diphenyl ether (PBDE) contaminant levels across marine species, SRKWs tested the highest of any other killer whale ecotype and higher than any other tested organism (Krahn et al. 2007, Alonso et al. 2014, NMFS 2016c). The exposure to and ingestion of these contaminants through the consumption of Chinook salmon prey is spatially dependent, as salmon sampled from more southern areas in Puget Sound were found to have higher contaminant levels and lower lipid content than those in British Columbia, possibly explaining the higher contaminant loads in SRKW compared to Northern Resident killer whales (Cullon et al. 2009). Prey depletion can further synergistically exacerbate exposure, as nutritional stress leads to metabolizing of contaminants stored in blubber (Lundin et al. 2016, Mongillo et al. 2016). Physical marine debris is also likely negatively impacting cetaceans, though knowledge to date is largely based on the small number of stranding necropsies, and therefore the mechanisms of ingestion and the extent of the threat remain unknown (Simmonds 2012).

Instruments of Recovery

As noted above, cetaceans and pinnipeds are managed and protected under the MMPA and the ESA, acts designed to foster the collection of scientific data, minimize anthropogenic impacts to the extent practical, and provide for the recovery of depleted species. These laws require stock assessment reports, conservation and recovery plans, and the designation of critical habitat, though implementing regulations can be a slow and controversial process, making it challenging to react quickly enough to minimize impacts from a new or growing threat. Additionally, the efficacy of provisions in the MMPA is highly context-dependent, proving successful in some circumstances but not others (Geijer and Read 2013). ESA-listed large whales are under active recovery plans, though the specific impacts of ongoing anthropogenic activities remain relatively unquantified. The SRKW is under an active recovery plan with a Priority Number of 1 based on the high potential for recovery and the likelihood of economic impacts from implementing recovery actions (NMFS 2008b, 2016c). Management and conservation options for reducing the impacts of anthropogenic activities on SRKW include: a) maintaining the integrity of critical habitat and reducing contaminant exposure, b) reducing vessel traffic noise and disturbance from whale watching, c) reducing direct and indirect fisheries impacts, and d) improving the use of cumulative impacts modeling and the implementation of EBFM.

Critical habitat designations remain an important aspect of cetacean conservation and recovery by minimizing the impacts of anthropogenic activities in areas that are vital to foraging, resting, reproducing, or migrating. However, a critical habitat designation does not preclude human activities from occurring in those places. Additionally, much remains unknown about the specific importance of these areas, the threshold for when degradation (from noise, pollution, derelict gear, oil, etc.) becomes detrimental, or how these spaces might geographically shift in the coming decades. On the U.S. West Coast, critical habitat has been designated for two cetacean populations: SRKW and North Pacific right whales. The conservation benefits of these designations remain unquantified. For SRKW, critical habitat spans inland Washington waters and includes maintaining sufficient prey resources, but this designation alone may be insufficient to ensure that habitat is conducive to population recovery.

In 2011, NOAA Fisheries implemented regulations instituting a 200-yard limit on how close vessels (commercial and private) could approach individual whales, though enforcing these regulations can be resource-limited, and studies are ongoing as to their effectiveness. The Be Whale Wise guidelines also encourage vessels to reduce their speeds, though these are voluntary best practices rather than legal requirements. For ESA-listed cetaceans all along the U.S. West Coast, ship strikes remain a significant threat to continued recovery.

Salmon recovery efforts are continuously underway, with collaborations across federal, state, local, and tribal entities. The success of these efforts also greatly depends on short- and long-term weather patterns, with recent trends toward warmer, drier conditions throughout California not being conducive to salmon habitat restoration or increasing run sizes. The legal framework provided by the ESA under which these recovery activities take place is one of the strongest in the country, though high economic costs to industry are a remaining impediment.

In terms of chemical pollutants, NOAA Fisheries continues to collaborate with the Puget Sound Partnership (PSP) and the EPA in restoring SRKW habitat and reducing contaminant loads in Puget Sound by removing flame retardants from wastewater, among other initiatives. Efforts are also underway to improve scientific understanding about the spatial extent of these chemicals and toxicological thresholds for SRKWs (NMFS 2016c). Though there is historical precedent for reducing certain deleterious chemicals from our environments, it remains a longer-term goal. Reducing and mediating non-point-source pollution is very difficult, as there is not a single, identifiable stakeholder to target for tighter regulations or compensatory mitigation funds.

Data Gaps and Opportunities for Applying Economic Analysis

Most of the threats to SRKW persist because there are economic benefits to be gained from continued operation of wildlife tourism, salmon fishing, and shipping industries. Many of the actions outlined in the recovery plan are feasible but come with significant economic costs, presenting a situation where the trade-offs between SRKW recovery and industry economic profit are evident and can create contention (Marshall et al. 2016). For example, prioritizing the recovery of certain salmon runs likely comes at a cost to specific fishermen, and limiting the number of whales a vessel can approach or the time they can be in close proximity could diminish the popularity of a given whale watching company. The SRKW situation is particularly challenging as there is not usually a direct link between anthropogenic activities and injury or mortality, as it is the accumulation of stressful indirect effects over time that is likely leading to poor health and insufficient reproductive output (NMFS 2016c). This situation could be seen as a tragedy of the commons, where benefits accrue to the users, but no single user is responsible for mitigation of the threats.

While the efficacy of vessel approach regulations continues to be studied, the whale watching industry can move toward long-term sustainability by adopting more comprehensive management that integrates all stakeholder groups and is adaptive to prevailing and changing conservation priorities (Higham et al. 2008, New et al. 2015). Researchers argue that whale watching can be viewed as a consumptive activity managed within a framework that accounts for the negative externalities created by the industry (Higham et al. 2016). Rather than being seen as a public resource, whales can be viewed as a common-pool resource, where the amount of time a vessel spends near one whale diminishes the amount of time available for other vessels to observe that whale (Higham et al. 2008, Pirotta and Lusseau 2015). In this way, limits could be established on the number of vessels and/or the amount of time vessels can spend in close proximity to individual whales. The regulations established by NOAA Fisheries in 2011 attempt to minimize disturbance by keeping vessels at a safe distance, but they do not necessarily limit the number of vessels or the amount of time vessels can follow a certain individual, nor are they likely to be consistently enforced throughout the SRKW range. It may prove necessary to adapt these regulations in the future to maximize effectiveness based on best available information. In terms of nontourism vessel impacts, some shipping companies have been investing in potential vessel quieting technologies, but there would have to be an economic incentive (e.g., reduced operating costs or improved efficiency for quieter propellers) for these changes to be embraced across the industry (Southall 2005).

An opportunity for economic research to inform the recovery and management of depleted prey and predator populations would be to conduct a cost–benefit analysis of a joint recovery plan for SRKW and salmon that includes the energy requirements of SRKWs (Noren 2011), dynamic oceanographic conditions, and predicted salmon runs. To date, no management strategies have been identified that would hasten SRKW recovery (Hilborn et al. 2012). Hilborn (2011) suggests that an "extended EBFM" that includes trophic interactions and spatial specificity is the ideal next step toward improved management, though potentially cost-prohibitive in its implementation. As technology improves and data gaps shrink, applications of sophisticated bioeconomic and ecosystem modeling (e.g., Atlantis ecosystem model, Kaplan and Levin 2009) may become feasible. CBA could lead to unbiased decisions and mediate contentions arising from competing priorities and trade-offs inherent in managing two ESA-listed species or multiple predators targeting a limited prey resource (Chasco et al. 2017a, 2017b).

Continuing to expand applications for DOM will facilitate multispecies decision-making, stakeholder involvement, and near-real-time monitoring. This type of management can be costly, and therefore could be an appropriate beneficiary of conservation mitigation funding. WhaleWatch is a whale density prediction tool that has been instituted to minimize human impacts on blue whales off the west coast, allowing near-real-time predictive occurrence based on environmental conditions (Hazen et al. 2016). Developing, implementing, and expanding these modeling techniques will require continued research into spatial ecology, trophic interactions, threshold contaminant exposure levels, and scaling up from individual- to population-level impacts. In addition, modeling the cumulative impacts of multiple threats throughout the ranges of both pinniped and cetacean populations will continue to be beneficial. In this way, managers can assess the impacts of multiple threats in a given area, or a single threat across space as animals migrate. We currently do not have sufficient information to assess how species are impacted by multiple anthropogenic activities throughout their range or over time. This becomes particularly challenging when weighing the benefits of different conservation and management alternatives, because it is impossible to quantify benefits to a population or species if total mortality and injury rates are unknown.

Though recovery action costs are high, they are perhaps best evaluated in the context of the potentially large benefits of recovery, such as increased property values, the continued viability of the whale watching industry and local tourism economies (which would decline precipitously without sufficient SRKWs), and the important ecological role SRKWs play in the ecosystems upon which salmon depend. A robust CBA of SRKW recovery—one that includes these indirect benefits in addition to inherent nonuse existence values—has not yet been conducted. Such an analysis, combined with a cost-effectiveness analysis of all recovery plan actions, could act as an important tool to find a holistic approach that minimizes industry economic trade-offs.

Conclusions

In this technical memorandum, we explored the current and potential use of economic analysis to inform MPR management in NOAA's West Coast Region. We identified and illustrated key challenges facing MPR managers, including increasingly scarce resources for recovery and difficulties with developing and prioritizing recovery policies. We also highlighted economic tools for addressing management challenges, including understanding the effects of policies on conservation and stakeholder outcomes.

Our survey of the literature, combined with our analysis of individual species, generated several broad insights regarding the application of economic analysis to MPR management. First, our review underscored the potential for economic methods and frameworks to inform the evaluation of candidate MPR policies and guide socially desirable MPR resource allocation. Second, our review highlighted the importance of institutions in moderating conservation outcomes, and the utility of tools from economics and other social sciences for understanding institutional effects. A final takeaway from this report is that all recovery aspects of MPR policy planning should plan for a world with shifting baseline conditions due to ongoing anthropogenic effects such as climate change and economic development.

References

Abbott, J. K., and J. E. Wilen. 2009. Regulation of fisheries bycatch with common-pool output quotas. Journal of Environmental Economics and Management 57(2):195–204.

Aburto-Oropeza, O., C. López-Sagástegui, M. Moreno-Báez, I. Mascareñas-Osorio, V. Jiménez-Esquivel, A. F. Johnson, and B. Erisman. 2018. Endangered species, ecosystem integrity, and human livelihoods. Conservation Letters 11(1):e12358.

Adamowicz, W. L. 2016. Economic analysis and species at risk: Lessons learned and future challenges. Canadian Journal of Agricultural Economics/Revue canadienne d'agroéconomie 64(1):21–32.

Alava, J. J., D. Lambourn, P. Olesiuk, M. Lance, S. J. Jeffries, F. A. Gobas, and P. S. Ross. 2012. PBDE flame retardants and PCBs in migrating Steller sea lions (*Eumetopias jubatus*) in the Strait of Georgia, British Columbia, Canada. Chemosphere 88(7):855–864.

Alonso, M. B., A. Azevedo, J. P. M. Torres, P. R. Dorneles, E. Eljarrat, D. Barceló, J. Lailson-Brito, and O. Malm. 2014. Anthropogenic (PBDE) and naturally-produced (MeO-PBDE) brominated compounds in cetaceans—a review. Science of the Total Environment 481:619–634.

Anderson, L. E., S. T. Lee, and P. S. Levin. 2013. Costs of delaying conservation: Regulations and the recreational values of exploited and co-occurring species. Land Economics 89(2):371–385.

Andrews, K. S., K. M. Nichols, A. Elz, N. Tolimieri, C. J. Harvey, R. Pacunski, D. Lowry, K. L. Yamanaka, and D. M. Tonnes. 2018. Cooperative research sheds light on population structure and listing status of threatened and endangered rockfish species. Conservation Genetics 19(4):1–14.

Antonelis, K., D. Huppert, D. Velasquez, and J. June. 2011. Dungeness crab mortality due to lost traps and a cost–benefit analysis of trap removal in Washington State waters of the Salish Sea. North American Journal of Fisheries Management 31(5):880–893.

Athey, S., and G. W. Imbens. 2017. The state of applied econometrics: Causality and policy evaluation. Journal of Economic Perspectives 31(2):3–32.

Atkinson, S., D. P. Demaster, and D. G. Calkins. 2008. Anthropogenic causes of the western Steller sea lion *Eumetopias jubatus* population decline and their threat to recovery. Mammal Review 38(1):1–18.

Ayres, K. L., R. K. Booth, J. A. Hempelmann, K. L. Koski, C. K. Emmons, R. W. Baird, K. Balcomb-Bartok, M. B. Hanson, M. J. Ford, and S. K. Wasser. 2012. Distinguishing the impacts of inadequate prey and vessel traffic on an endangered killer whale (*Orcinus orca*) population. PLOS One 7(6):e36842.

Baker, J. D., C. L. Littnan, and D. W. Johnston. 2006. Potential effects of sea level rise on the terrestrial habitats of endangered and endemic megafauna in the Northwestern Hawaiian Islands. Endangered Species Research 2:21–30.

Barbier, E. B. 2007. Valuing ecosystem services as productive inputs. Economic Policy 22(49):178–229.

Barnas, K. A., S. L. Katz, D. E. Hamm, M. C. Diaz, and C. E. Jordan. 2015. Is habitat restoration targeting relevant ecological needs for endangered species? Using Pacific salmon as a case study. Ecosphere 6(7):1–42.

Barnes-Mauthe, M., S. A. Gray, S. Arita, J. Lynham, and P. Leung. 2015. What determines social capital in a social–ecological system? Insights from a network perspective. Environmental Management 55(2):392–410.

Barron, M. G., R. Heintz, and M. M. Krahn. 2003. Contaminant exposure and effects in pinnipeds: Implications for Steller sea lion declines in Alaska. Science of the Total Environment 311(1):111–133.

Beaudreau, A. H., P. S. Levin, and K. C. Norman. 2011. Using folk taxonomies to understand stakeholder perceptions for species conservation. Conservation Letters 4(6):451–463.

Bell, K. P., D. Huppert, and R. L. Johnson. 2003. Willingness to pay for local coho salmon enhancement in coastal communities. Marine Resource Economics 18(1):15–31.

Bisack, K. D., and C. Das. 2015. Understanding non-compliance with protected species regulations in the northeast USA gillnet fishery. Frontiers in Marine Science 2:91.

Bisack, K. D., and G. Magnusson. 2014. Measuring the economic value of increased precision in scientific estimates of marine mammal abundance and bycatch: Harbor porpoise *Phocoena phocoena* in the northeast US gill-net fishery. North American Journal of Fisheries Management 34(2):311–321.

Bisack, K. D., and J. G. Sutinen. 2006. Harbor porpoise bycatch: ITQs or time/area closures in the New England gillnet fishery. Land Economics 82(1):85–102.

Boxall, P., W. Adamowicz, M. Olar, G. West, and G. Cantin. 2012. Analysis of the economic benefits associated with the recovery of threatened marine mammal species in the Canadian St. Lawrence Estuary. Marine Policy 36(1):189–197.

Brewer, D., D. Heales, D. Milton, Q. Dell, G. Fry, B. Venables, and P. Jones. 2006. The impact of turtle excluder devices and bycatch reduction devices on diverse tropical marine communities in Australia's northern prawn trawl fishery. Fisheries Research 81(2):176–188.

Buckles, E., L. Lowenstine, C. Funke, R. Vittore, H.-N. Wong, J. St. Leger, D. J. Greig, R. Duerr, F. Gulland, and J. Stott. 2006. Otarine herpesvirus-1, not papillomavirus, is associated with endemic tumours in California sea lions (*Zalophus californianus*). Journal of Comparative Pathology 135(4):183–189.

Buhle, E. R., K. K. Holsman, M. D. Scheuerell, and A. Albaugh. 2009. Using an unplanned experiment to evaluate the effects of hatcheries and environmental variation on threatened populations of wild salmon. Biological Conservation 142(11):2449–2455.

Carretta, J. V., E. M. Oleson, J. Baker, D. W. Weller, A. R. Lang, K. A. Forney, M. M. Muto, B. Hanson, A. J. Orr, H. Huber, M. S. Lowry, J. Barlow, J. E. Moore, D. Lynch, L. Carswell, and R. L. Brownell, Jr. 2016. U.S. Pacific Marine Mammal Stock Assessments: 2015. U.S Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-561.

Catton, C. A., K. L. Stierhoff, and L. Rogers-Bennett. 2016. Population status assessment and restoration modeling of white abalone *Haliotis sorenseni* in California. Journal of Shellfish Research 35(3):593–599.

CDFG (California Department of Fish and Game). 2005. Abalone Recovery and Management Plan. Sacramento, California.

Chan, H. L., and M. Pan. 2016. Spillover effects of environmental regulation for sea turtle protection in the Hawaii longline swordfish fishery. Marine Resource Economics 31(3):259–279.

Chasco, B., I. C. Kaplan, A. Thomas, A. Acevedo-Gutiérrez, D. Noren, M. J. Ford, M. B. Hanson, J. Scordino, S. Jeffries, S. Pearson, K. N. Marshall, and E. J. Ward. 2017a. Estimates of Chinook salmon consumption in Washington State inland waters by four marine mammal predators from 1970 to 2015. Canadian Journal of Fisheries and Aquatic Sciences. Available: www.nrcresearchpress.com/doi/abs/10.1139/cjfas-2016-0203 (October 2018).

Chasco, B. E., I. C. Kaplan, A. C. Thomas, A. Acevedo-Gutiérrez, D. P. Noren, M. J. Ford, M. B. Hanson, J. J. Scordino, S. J. Jeffries, K. N. Marshall, A. O. Shelton, C. Matkin, B. J. Burke, and E. J. Ward. 2017b. Competing tradeoffs between increasing marine mammal predation and fisheries harvest of Chinook salmon. Scientific Reports 7:15439.

Conant, T. A., P. H. Dutton, T. Eguchi, S. P. Epperly, C. C. Fahy, M. H. Godfrey, S. L. MacPherson, E. E. Possardt, B. A. Schroeder, J. A. Seminoff, M. L. Snover, C. M. Upite, and B. E. Witherington. 2009. Loggerhead Sea Turtle (*Caretta caretta*) 2009 Status Review Under the U.S. Endangered Species Act. Report of the loggerhead biological review team to the National Marine Fisheries Service.

Cox, T., R. Lewison, R. Žydelis, L. Crowder, C. Safina, and A. Read. 2007. Comparing effectiveness of experimental and implemented bycatch reduction measures: The ideal and the real. Conservation Biology 21(5):1155–1164.

Crozier, L. 2016. Impacts of Climate Change on Salmon of the Pacific Northwest. Available: www.nwfsc.noaa.gov/assets/4/9042_02102017_105951_Crozier.2016-BIOP-Lit-Rev-Salmon-Climate-Effects-2015.pdf (October 2018).

Cullon, D. L., S. J. Jeffries, and P. S. Ross. 2005. Persistent organic pollutants in the diet of harbor seals (*Phoca vitulina*) inhabiting Puget Sound, Washington (USA), and the Strait of Georgia, British Columbia (Canada): A food basket approach. Environmental Toxicology and Chemistry 24(10):2562–2572.

Cullon, D. L., M. B. Yunker, C. Alleyne, N. J. Dangerfield, S. O'Neill, M. J. Whiticar, and P. S. Ross. 2009. Persistent organic pollutants in Chinook salmon (*Oncorhynchus tshawytscha*): Implications for resident killer whales of British Columbia and adjacent waters. Environmental Toxicology and Chemistry 28(1):148–161.

Curtis, R., and R. L. Hicks. 2000. The cost of sea turtle preservation: The case of Hawaii's pelagic longliners. American Journal of Agricultural Economics 82(5):1191–1197.

CWR (Center for Whale Research). 2017. 2017 SRKW Population Update. Available: simplebooklet.com/userFiles/a/3/5/9/0/3/1/HiPDDCYGTuXh2pyNPxHwB6/HiPDDCYGTuXh2pyNPxHwB6.pdf (October 2018).

Dillingham, P. W., J. R. Skalski, and K. E. Ryding. 2006. Fine-scale geographic interactions between Steller sea lion (*Eumetopias jubatus*) trends and local fisheries. Canadian Journal of Fisheries and Aquatic Sciences 63(1):107–119.

Doak, D., V. Bakker, M. Finkelstein, B. Sullivan, R. Lewison, B. Keitt, J. Arnold, J. Croxall, F. Micheli, and M. Sanjayan. 2007. Compensatory mitigation for marine bycatch will do harm, not good. Frontiers in Ecology and the Environment 5(7):350–351.

DOI (U.S. Department of the Interior). 2012. Klamath Dam Removal Overview Report for the Secretary of the Interior: An Assessment of Science and Technical Information. Available: klamathrestoration.gov/sites/klamathrestoration.gov/files/2013%20Updates/Final%20SDOR%20/0.Final%20Accessible%20SDOR%2011.8.2012.pdf (October 2018).

Donlan, C., D. K. Wingfield, L. B. Crowder, and C. Wilcox. 2010. Using expert opinion surveys to rank threats to endangered species: A case study with sea turtles. Conservation Biology 24(6):1586–1595.

Dundas, S. J., R. H. von Haefen, and C. Mansfield. 2018. Recreation Costs of Endangered Species Protection: Evidence from Cape Hatteras National Seashore. Marine Resource Economics 33(1):1–25.

Dutton, P. H., and D. Squires. 2011. A holistic strategy for Pacific sea turtle conservation. Pages 37–59 *in* P. H. Dutton, D. Squires, and A. Mahfuzuddin, editors. Conservation and sustainable management of sea turtles in the Pacific Ocean. University of Hawaii Press, Honolulu.

Duvall, A. L., A. L. Metcalf, and P. S. Coates. 2017. Conserving the greater sage-grouse: A social-ecological systems case study from the California–Nevada region. Rangeland Ecology and Management 70(1):129–140.

EcoCast. 2017. EcoCast: Real-time data tools for dynamic fisheries management. Available: www.conservationecologylab.com/uploads/1/9/7/6/19763887/ecocast_for_web.pdf (October 2018).

Eguchi, T., S. R. Benson, D. G. Foley, and K. A. Forney. 2017. Predicting overlap between drift gillnet fishing and leatherback turtle habitat in the California Current Ecosystem. Fisheries Oceanography 26(1):17–33.

Elorriaga-Verplancken, F. R., G. E. Sierra-Rodríguez, H. Rosales-Nanduca, K. Acevedo-Whitehouse, and J. Sandoval-Sierra. 2016. Impact of the 2015 El Niño–Southern Oscillation on the abundance and foraging habits of Guadalupe fur seals and California sea lions from the San Benito Archipelago, Mexico. PLOS One 11(5):e0155034.

Erbe, C. 2002. Underwater noise of whale-watching boats and potential effects on killer whales (*Orcinus orca*), based on an acoustic impact model. Marine Mammal Science 18(2):394–418.

Essington, T. E., J. N. Sanchirico, and M. L. Baskett. 2018. Economic value of ecological information in ecosystem-based natural resource management depends on exploitation history. Proceedings of the National Academy of Sciences:201716858.

Field, C. R., A. A. Dayer, and C. S. Elphick. 2017. Landowner behavior can determine the success of conservation strategies for ecosystem migration under sea-level rise. Proceedings of the National Academy of Sciences:201620319.

Finkbeiner, E. M., B. P. Wallace, J. E. Moore, R. L. Lewison, L. B. Crowder, and A. J. Read. 2011. Cumulative estimates of sea turtle bycatch and mortality in USA fisheries between 1990 and 2007. Biological Conservation 144(11):2719–2727.

Finkelstein, M., V. Bakker, D. F. Doak, B. Sullivan, R. Lewison, W. H. Satterthwaite, P. B. McIntyre, S. Wolf, D. Priddel, and J. M. Arnold. 2008. Evaluating the potential effectiveness of compensatory mitigation strategies for marine bycatch. PLOS One 3(6):e2480.

Forbes, K., P. C. Boxall, W. L. Adamowicz, and A. De Maio Sukic. 2015. Recovering Pacific rockfish at risk: The economic valuation of management actions. Frontiers in Marine Science 2:71.

Ford, J. K. B., G. M. Ellis, P. F. Olesiuk, and K. C. Balcomb. 2010. Linking killer whale survival and prey abundance: Food limitation in the oceans' apex predator? Biology Letters 6(1):139–142.

Ford, M. J., J. Hempelmann, M. B. Hanson, K. L. Ayres, R. W. Baird, C. K. Emmons, J. I. Lundin, G. S. Schorr, S. K. Wasser, and L. K. Park. 2016. Estimation of a killer whale (*Orcinus orca*) population's diet using sequencing analysis of DNA from feces. PLOS One 11(1):e0144956.

Fritz, L. W., and S. Hinckley. 2005. A Critical Review of the Regime Shift-"Junk Food"-Nutritional Stress Hypothesis for the Decline of the Western Stock of Steller Sea Lion. Marine Mammal Science 21(3):476–518.

Fullerton, A., D. Jensen, E. A. Steel, D. Miller, and P. McElhany. 2010a. How certain are salmon recovery forecasts? A watershed-scale sensitivity analysis. Environmental Modeling & Assessment 15(1):13–26.

Fullerton, A. H., S. T. Lindley, G. R. Pess, B. E. Feist, E. A. Steel, and P. McElhany. 2011. Human influence on the spatial structure of threatened Pacific salmon metapopulations. Conservation Biology 25(5):932–944.

Fullerton, A. H., E. A. Steel, I. Lange, and Y. Caras. 2010b. Effects of spatial pattern and economic uncertainties on freshwater habitat restoration planning: a simulation exercise. Restoration Ecology 18(s2):354–369.

Geijer, C. K., and A. J. Read. 2013. Mitigation of marine mammal bycatch in US fisheries since 1994. Biological Conservation 159:54–60.

Gerber, L. R. 2016. Conservation triage or injurious neglect in endangered species recovery. Proceedings of the National Academy of Sciences 113(13):3563–3566.

Gibson, C. 2013. Preventing the Loss of Gillnets in Puget Sound Salmon Fisheries. Prepared by the Northwest Straits Marine Conservation Foundation with the Northwest Indian Fisheries Commission, Bellingham, Washington.

Gilman, E., D. Kobayashi, T. Swenarton, N. Brothers, P. Dalzell, and I. Kinan-Kelly. 2007. Reducing sea turtle interactions in the Hawaii-based longline swordfish fishery. Biological Conservation 139(1):19–28.

Gjertsen, H. 2011. Can we improve our conservation bang for the buck? Cost-effectiveness of alternative leatherback turtle conservation strategies. Pages 60–84 *in* P. H. Dutton, D. Squires, and M. Ahmed, editors. Conservation of Pacific Sea Turtles. University of Hawaii Press, Honolulu.

Gjertsen, H., D. Squires, P. H. Dutton, and T. Eguchi. 2014. Cost-Effectiveness of Alternative Conservation Strategies with Application to the Pacific Leatherback Turtle. Conservation Biology 28(1):140–149.

Good, T. P., J. A. June, M. A. Etnier, and G. Broadhurst. 2009. Ghosts of the Salish Sea: Threats to marine birds in Puget Sound and the Northwest Straits from derelict fishing gear. Marine Ornithology 37(1):67–76.

Goundie, E. T., D. A. Rosen, and A. W. Trites. 2015. Low prey abundance leads to less efficient foraging behavior in Steller sea lions. Journal of Experimental Marine Biology and Ecology 470:70–77.

Grand, L., K. D. Messer, and W. Allen. 2017. Understanding and overcoming the barriers for cost-effective conservation. Ecological Economics 138:139–144.

Guénette, S., S. J. Heymans, V. Christensen, and A. W. Trites. 2006. Ecosystem models show combined effects of fishing, predation, competition, and ocean productivity on Steller sea lions (*Eumetopias jubatus*) in Alaska. Canadian Journal of Fisheries and Aquatic Sciences 63(11):2495–2517.

Gustafson, R. G., editor. 2016. Status Review Update of Eulachon (*Thaleichthys pacificus*) Listed under the Endangered Species Act: Southern Distinct Population Segment. Available: www.westcoast.fisheries.noaa.gov/publications/status_reviews/other_species/eulachon/eulachon_2016_status_review_update.pdf (October 2018).

Gustafson, R. G., M. J. Ford, D. Teel, and J. S. Drake. 2010. Status Review of Eulachon (*Thaleichthys pacificus*) in Washington, Oregon, and California. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-105.

Gustafson, R. G., R. S. Waples, J. M. Myers, L. A. Weitkamp, G. J. Bryant, O. W. Johnson, and J. J. Hard. 2007. Pacific salmon extinctions: Quantifying lost and remaining diversity. Conservation Biology 21(4):1009–1020.

Hanna, S. 2008. Institutions for Managing Resilient Salmon (*Oncorhynchus* spp.) Ecosystems: The Role of Incentives and Transaction Costs. Ecology and Society 13(2):35.

Hanson, M. B., R. W. Baird, J. K. Ford, J. Hempelmann-Halos, D. M. Van Doornik, J. R. Candy, C. K. Emmons, G. S. Schorr, B. Gisborne, and K. L. Ayres. 2010. Species and stock identification of prey consumed by endangered southern resident killer whales in their summer range. Endangered Species Research 11(1):69–82.

Haynie, A. C., and D. F. Layton. 2010. An expected profit model for monetizing fishing location choices. Journal of Environmental Economics and Management 59(2):165–176.

Hazen, E. L., S. Jorgensen, R. R. Rykaczewski, S. J. Bograd, D. G. Foley, I. D. Jonsen, S. A. Shaffer, J. P. Dunne, D. P. Costa, and L. B. Crowder. 2013. Predicted habitat shifts of Pacific top predators in a changing climate. Nature Climate Change 3(3):234–238.

Hazen, E. L., D. M. Palacios, K. A. Forney, E. A. Howell, E. Becker, A. L. Hoover, L. Irvine, M. DeAngelis, S. J. Bograd, and B. R. Mate. 2016. WhaleWatch: A dynamic management tool for predicting blue whale density in the California Current. Journal of Applied Ecology 54(5):1415–1428.

Higham, J., L. Bejder, and D. Lusseau. 2008. An integrated and adaptive management model to address the long-term sustainability of tourist interactions with cetaceans. Environmental Conservation 35(4):294–302.

Higham, J. E., L. Bejder, S. J. Allen, P. J. Corkeron, and D. Lusseau. 2016. Managing whale-watching as a non-lethal consumptive activity. Journal of Sustainable Tourism 24(1):73–90.

Hilborn, R. 2011. Future directions in ecosystem based fisheries management: A personal perspective. Fisheries Research 108(2):235–239.

Hilborn, R., S. Cox, F. Gulland, D. Hankin, N. Hobbs, D. Schindler, and A. Trites. 2012. The Effects of Salmon Fisheries on Southern Resident Killer Whales: Final Report of the Independent Science Panel. Prepared with the assistance of D. R. Marmorek and A. W. Hall, ESSA Technologies Ltd., Vancouver. Available: www.westcoast.fisheries.noaa.gov/publications/protected_species/marine_mammals/killer_whales/recovery/kw-effects_of_salmon_fisheries_on_srkw-final-rpt.pdf (October 2018).

Hobday, A. J., and M. J. Tegner. 2000. Status Review of White Abalone (*Haliotis sorenseni*) Throughout its Range in California and Mexico. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWR-35.

Hobday, A. J., S. M. Maxwell, J. Forgie, and J. McDonald. 2013. Dynamic ocean management: Integrating scientific and technological capacity with law, policy, and management. Stanford Environmental Law Journal 33:125.

Hoekstra, J. M., K. K. Bartz, M. H. Ruckelshaus, J. M. Moslemi, and T. K. Harms. 2007. Quantitative threat analysis for management of an imperiled species: Chinook salmon (*Oncorhynchus tshawytscha*). Ecological Applications 17(7):2061–2073.

Hogan, E., and A. Warlick. 2017. Packing Bands Entangling Pinnipeds Around the World: Global Review and Policy. Journal of International Wildlife Law & Policy 20(1):96–104.

Holland, D. S. 2010. Markets, pooling and insurance for managing bycatch in fisheries. Ecological Economics 70(1):121–133.

Holland, D. S., and J. E. Jannot. 2012. Bycatch risk pools for the US West Coast groundfish fishery. Ecological Economics 78:132–147.

Holt, M. M., D. P. Noren, and C. K. Emmons. 2012. Does Vessel Noise Affect the Use of Sound by Foraging *Orcinus orca* (Killer Whales)? Pages 327–330 *in* A. N. Popper and A. Hawkins, editors. The Effects of Noise on Aquatic Life. Advances in Experimental Medicine and Biology, volume 730. Springer: New York.

Holt, M. M., D. P. Noren, V. Veirs, C. K. Emmons, and S. Veirs. 2009. Speaking up: Killer whales (*Orcinus orca*) increase their call amplitude in response to vessel noise. The Journal of the Acoustical Society of America 125(1):EL27–EL32.

Houghton, J., M. Holt, D. Giles, C. Emmons, B. Hanson, J. Hogan, T. Branch, and G. VanBlaricom. 2015. The relationship between vessel traffic and noise levels received by killer whales. The Journal of the Acoustical Society of America 134(5):4044–4044.

Howell, E. A., A. Hoover, S. R. Benson, H. Bailey, J. J. Polovina, J. A. Seminoff, and P. H. Dutton. 2015. Enhancing the TurtleWatch product for leatherback sea turtles, a dynamic habitat model for ecosystem-based management. Fisheries Oceanography 24(1):57–68.

Huang, H., and P. Leung. 2007. Modeling protected species as an undesirable output: The case of sea turtle interactions in Hawaii's longline fishery. Journal of Environmental Management 84(4):523–533.

Innes, J., S. Pascoe, C. Wilcox, S. Jennings, and S. Paredes. 2015. Mitigating undesirable impacts in the marine environment: A review of market-based management measures. Frontiers in Marine Science 2:76.

Jeffrey, C. F. G., K. J. Havens, H. W. Slacum, Jr., D. M. Bilkovic, D. Zaveta, A. M. Scheld, S. Willard, and J. D. Evans. 2016. Assessing Ecological and Economic Effects of Derelict Fishing Gear: A Guiding Framework. U.S. Department of Commerce, NOAA Marine Debris Program Office of Response and Restoration. Available: marinedebris.noaa.gov/reports/assessing-ecological-and-economic-effects-derelict-fishing-gear-guiding-framework (October 2018).

Kaplan I. C., and P. Levin. 2009. Ecosystem-Based Management of What? An Emerging Approach for Balancing Conflicting Objectives in Marine Resource Management. Pages 77–95 *in* R. J. Beamish and B. J. Rothschild, editors. The Future of Fisheries Science in North America. Fish & Fisheries Series, volume 31. Springer, Dordrecht, the Netherlands.

Kauer, K., and D. Oberhoff. 2015. California Groundfish Collective Annual Report 2014. Report to the Pacific Fisheries Management Council. Available: www.pcouncil.org/wp-content/uploads/2015/09/SUP_IR10_CA_GROUNDFISH_COLLECTIVE_RPT_SEPT2015BB.pdf (October 2018).

Komoroske, L. M., and R. L. Lewison. 2015. Addressing fisheries bycatch in a changing world. Frontiers in Marine Science 2:83.

Krahn, M. M., M. B. Hanson, R. W. Baird, R. H. Boyer, D. G. Burrows, C. K. Emmons, J. K. Ford, L. L. Jones, D. P. Noren, and P. S. Ross. 2007. Persistent organic pollutants and stable isotopes in biopsy samples (2004/2006) from Southern Resident killer whales. Marine Pollution Bulletin 54(12):1903–1911.

Lachmuth, C. L., L. G. Barrett-Lennard, D. Steyn, and W. K. Milsom. 2011. Estimation of southern resident killer whale exposure to exhaust emissions from whale-watching vessels and potential adverse health effects and toxicity thresholds. Marine Pollution Bulletin 62(4):792–805.

Langpap, C., and J. Kerkvliet. 2010. Allocating conservation resources under the Endangered Species Act. American Journal of Agricultural Economics 92(1):110–124.

Langpap, C., J. Kerkvliet, and J. Shogren. 2018. The Economics of the Endangered Species Act: A Review of Recent Developments. Review of Environmental Economics and Policy 12(1):69–91.

Langpap, C., and J. Wu. 2017. Thresholds, perverse incentives, and preemptive conservation of endangered species. Journal of the Association of Environmental and Resource Economics 4(S1):S227–S259.

LaRiviere, J., D. Kling, J. N. Sanchirico, C. Sims, and M. Springborn. 2017. The Treatment of Uncertainty and Learning in the Economics of Natural Resource and Environmental Management. Review of Environmental Economics and Policy 12(1):92–112.

Lawrence, D. J., B. Stewart-Koster, J. D. Olden, A. S. Ruesch, C. E. Torgersen, J. J. Lawler, D. P. Butcher, and J. K. Crown. 2014. The interactive effects of climate change, riparian management, and a nonnative predator on stream-rearing salmon. Ecological Applications 24(4):895–912.

Lennox, G. D., K. J. Gaston, S. Acs, M. Dallimer, N. Hanley, and P. R. Armsworth. 2013. Conservation when landowners have bargaining power: Continuous conservation investments and cost uncertainty. Ecological Economics 93:69–78.

Lent, R., and D. Squires. 2017. Reducing marine mammal bycatch in global fisheries: An economics approach. Deep Sea Research Part II: Topical Studies in Oceanography 140:268–277.

Lent, R. J. 2015. Conservation benefits of an interdisciplinary approach to marine mammal science. Frontiers in Marine Science 2:67.

Levin, P. S., and G. W. Stunz. 2005. Habitat triage for exploited fishes: Can we identify essential "Essential Fish Habitat?" Estuarine, Coastal and Shelf Science 64(1):70–78.

Lew, D. K. 2015. Willingness to pay for threatened and endangered marine species: A review of the literature and prospects for policy use. Frontiers in Marine Science 2:96.

Lew, D. K., D. F. Layton, and R. D. Rowe. 2010. Valuing enhancements to endangered species protection under alternative baseline futures: The case of the Steller sea lion. Marine Resource Economics 25(2):133–154.

Lewison, R. L., and L. B. Crowder. 2007. Putting longline bycatch of sea turtles into perspective. Conservation Biology 21(1):79–86.

Lewison, R. L., L. B. Crowder, B. P. Wallace, J. E. Moore, T. Cox, R. Zydelis, S. McDonald, A. DiMatteo, D. C. Dunn, and C. Y. Kot. 2014. Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. Proceedings of the National Academy of Sciences 111(14):5271–5276.

Lewison, R., A. J. Hobday, S. Maxwell, E. Hazen, J. R. Hartog, D. C. Dunn, D. Briscoe, S. Fossette, C. E. O'Keefe, M. Barnes, M. Abecassis, S. Bograd, N. D. Bethoney, H. Bailey, D. Wiley, S. Andrews, L Hazen, and L. B. Crowder. 2015. Dynamic ocean management: Identifying the critical ingredients of dynamic approaches to ocean resource management. BioScience 65(5):486–498.

Lewison, R. L., C. U. Soykan, and J. Franklin. 2009. Mapping the bycatch seascape: Multispecies and multi-scale spatial patterns of fisheries bycatch. Ecological Applications 19(4):920–930.

Lewitus, A. J., R. A. Horner, D. A. Caron, E. Garcia-Mendoza, B. M. Hickey, M. Hunter, D. D. Huppert, R. M. Kudela, G. W. Langlois, and J. L. Largier. 2012. Harmful algal blooms along the North American west coast region: History, trends, causes, and impacts. Harmful Algae 19:133–159.

Li, R., M. van den Brink, and J. Woltjer. 2016. Rules for the governance of coastal and marine ecosystem services: An evaluative framework based on the IAD framework. Land Use Policy 59:298–309.

Li, Y., and L. Rogers-Bennett. 2017. Evaluating factors affecting restoration of an endangered marine broadcast-spawning invertebrate using an individual-based model of white abalone. Endangered Species Research 32:293–308.

Lundin, J. I., G. M. Ylitalo, R. K. Booth, B. Anulacion, J. A. Hempelmann, K. M. Parsons, D. A. Giles, E. A. Seely, M. B. Hanson, and C. K. Emmons. 2016. Modulation in persistent organic pollutant concentration and profile by prey availability and reproductive status in Southern Resident killer whale scat samples. Environmental Science & Technology 50(12):6506–6516.

Maas-Hebner, K. G., C. Schreck, R. M. Hughes, J. A. Yeakley, and N. Molina. 2016. Scientifically Defensible Fish Conservation and Recovery Plans: Addressing Diffuse Threats and Developing Rigorous Adaptive Management Plans. Fisheries 41(6):276–285.

Mansfield, B., and J. Haas. 2006. Scale framing of scientific uncertainty in controversy over the endangered Steller sea lion. Environmental Politics 15(1):78–94.

Mansfield, C., G. Van Houtven, A. Hendershott, P. Chen, J. Porter, V. Nourani, and V. Kilambi. 2012. Klamath River Basin Restoration Nonuse Value Survey. U.S. Bureau of Reclamation, RTI Project Number 0212485.001.010. Available: klamathrestoration.gov/sites/klamathrestoration.gov/files/DDDDD.Printable.Klamath%20Nonuse%20Survey%20Final%20Report%202012%5B1%5D.pdf (October 2018).

Marshall, K. N., A. C. Stier, J. F. Samhouri, R. P. Kelly, and E. J. Ward. 2016. Conservation challenges of predator recovery. Conservation Letters 9(1):70–78.

Martin, S. L., S. M. Stohs, and J. E. Moore. 2015. Bayesian inference and assessment for rare-event bycatch in marine fisheries: A drift gillnet fishery case study. Ecological Applications 25(2):416–429.

Maselko, J., G. Bishop, and P. Murphy. 2013. Ghost fishing in the Southeast Alaska commercial Dungeness crab fishery. North American Journal of Fisheries Management 33(2):422–431.

Maxwell, S. M., E. L. Hazen, R. L. Lewison, D. C. Dunn, H. Bailey, S. J. Bograd, D. K. Briscoe, S. Fossette, A. J. Hobday, and M. Bennett. 2015. Dynamic ocean management: Defining and conceptualizing real-time management of the ocean. Marine Policy 58:42–50.

McClure, M. M., E. E. Holmes, B. L. Sanderson, and C. E. Jordan. 2003. A Large-Scale, Multispecies Status Assessment: Anadromous Salmonids in the Columbia River Basin. Ecological Applications 13(4):964–989.

McElhany, P., M. Ruckelshaus, M. Ford, T. Wainwright, and E. Bjorkstedt. 2000. Viable Salmonid Populations and the Recovery of Evolutionarily Significant Units. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-42.

Mongillo, T. M., G. M. Ylitalo, L. D. Rhodes, S. M. O'Neill, D. P. Noren, and M. B. Hanson. 2016. Exposure to a Mixture of Toxic Chemicals: Implications for the Health of Endangered Southern Resident Killer Whales. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-135.

Moody, M., and T. Pitcher. 2010. Eulachon (*Thaleichthys pacificus*): Past and Present. Fisheries Centre, University of British Columbia, Vancouver.

Moser, M., J. Israel, M. Neuman, S. Lindley, D. Erickson, B. McCovey, and A. Klimley. 2016. Biology and life history of Green Sturgeon (*Acipenser medirostris Ayres*, 1854): State of the science. Journal of Applied Ichthyology 32(S1):67–86.

Mussen, T. D., D. Cocherell, J. B. Poletto, J. S. Reardon, Z. Hockett, A. Ercan, H. Bandeh, M. L. Kavvas, J. J. Cech, Jr., and N. A. Fangue. 2014. Unscreened water-diversion pipes pose an entrainment risk to the threatened green sturgeon, *Acipenser medirostris*. PLOS One 9(1):e86321.

Naish, K. A., J. E. Taylor, P. S. Levin, T. P. Quinn, J. R. Winton, D. Huppert, and R. Hilborn. 2007. An evaluation of the effects of conservation and fishery enhancement hatcheries on wild populations of salmon. Advances in Marine Biology 53:61–194.

Netusil, N. R., and M. T. Summers. 2009. Valuing instream flows using the hedonic price method. Water Resources Research 45:W11429.

Neuman, M., B. Tissot, and G. Vanblaricom. 2010. Overall status and threats assessment of black abalone (*Haliotis cracherodii Leach*, 1814) populations in California. Journal of Shellfish Research 29(3):577–586.

New, L. F., A. J. Hall, R. Harcourt, G. Kaufman, E. Parsons, H. C. Pearson, A. M. Cosentino, and R. S. Schick. 2015. The modelling and assessment of whale-watching impacts. Ocean & Coastal Management 115:10–16.

Newbold, S. C., and J. Siikamäki. 2009. Prioritizing conservation activities using reserve site selection methods and population viability analysis. Ecological Applications 19(7):1774–1790.

NMFS (National Marine Fisheries Service). 2008a. Final White Abalone Recovery Plan (*Haliotis sorenseni*). National Marine Fisheries Service, Long Beach, California.

NMFS (National Marine Fisheries Service). 2008b. Recovery Plan for Southern Resident Killer Whales (*Orcinus orca*). National Marine Fisheries Service, Seattle.

NMFS (National Marine Fisheries Service). 2009. Status Review Report for Black Abalone (*Haliotis cracherodii*). National Marine Fisheries Service, Long Beach, California.

NMFS (National Marine Fisheries Service). 2010. Federal Recovery Outline, North American Green Sturgeon Southern Distinct Population Segment. National Marine Fisheries Service, Santa Rosa, California.

NMFS (National Marine Fisheries Service). 2013. Status Review of The Eastern Distinct Population Segment of Steller Sea Lion (*Eumetopias jubatus*). National Marine Fisheries Service, Juneau, Alaska.

NMFS (National Marine Fisheries Service). 2014a. Proceedings of the 2014 NOAA Economics of Protected Resources Workshop. National Marine Fisheries Service, La Jolla, California.

NMFS (National Marine Fisheries Service). 2014b. Section 7 Consultation on the Continued Implementation of the Sea Turtle Conservation Regulations under ESA and the Continued Authorization of the Southeast U.S. Shrimp Fisheries in Federal Waters under the Magnuson–Stevens Fishery Management and Conservation Act. National Marine Fisheries Service, St. Petersburg, Florida.

NMFS (National Marine Fisheries Service). 2015a. Chair's Summary of West Coast Protected Fish Species Science Program Review. National Matine Fisheries Service, La Jolla, California.

NMFS (National Marine Fisheries Service). 2015b. Draft Rockfish Recovery Plan: Puget Sound/Georgia Basin Yelloweye Rockfish (*Sebastes ruberrimus*) and Bocaccio (*Sebastes paucispinis*). National Marine Fisheries Service, Seattle.

NMFS (National Marine Fisheries Service). 2015c. Southern Distinct Population Segment of the North American Green Sturgeon (*Acipenser medirostris*) 5-Year Review: Summary and Evaluation. National Marine Fisheries Service, Long Beach, California.

NMFS (National Marine Fisheries Service). 2016a. 2016 5-Year Review: Summary & Evaluation of Eulachon. National Marine Fisheries Service, Portland, Oregon.

NMFS (National Marine Fisheries Service). 2016b. Recovery Plan Outline for Black Abalone (*Haliotis cracherodii*). National Marine Fisheries Service, Long Beach, California.

NMFS (National Marine Fisheries Service). 2016c. Southern Resident Killer Whale 5-Year Review: Summary and Evaluation. National Marine Fisheries Service, Seattle.

NMFS (National Marine Fisheries Service). 2016d. Yelloweye rockfish (*Sebastes ruberrimus*), canary rockfish (*Sebastes pinniger*), and boaccio (*Sebastes paucispinis*) of the Puget Sound/Georgia Basin 5-Year Review: Summary and Evaluation. National Marine Fisheries Service, Seattle.

NMFS (National Marine Fisheries Service). 2017. Rockfish Recovery Plan: Puget Sound/Georgia Basin Yelloweye Rockfish (*Sebastes ruberrimus*) and Bocaccio (*Sebastes paucispinis*). National Marine Fisheries Service, Seattle.

Noren, D., A. Johnson, D. Rehder, and A. Larson. 2009. Close approaches by vessels elicit surface active behaviors by Southern Resident killer whales. Endangered Species Research 8(3):179–192.

Noren, D. P. 2011. Estimated field metabolic rates and prey requirements of resident killer whales. Marine Mammal Science 27(1):60–77.

NRC (National Research Council). 2003. The decline of the Steller sea lion in Alaskan waters: Untangling food webs and fishing nets. National Academies Press, Washington, D.C.

Null, S. E., and J. Lund. 2012. Fish habitat optimization to prioritize river restoration decisions. River Research and Applications 28(9):1378–1393.

O'Keefe, C. E., S. X. Cadrin, and K. D. Stokesbury. 2013. Evaluating effectiveness of time/area closures, quotas/caps, and fleet communications to reduce fisheries bycatch. ICES Journal of Marine Science 71(5):1286–1297.

ODFW (Oregon Department of Fish and Wildlife). 1998. The Oregon Plan for Salmon and Watersheds. Oregon Department of Fish and Wildlife, Portland, Oregon.

Ogston, L., S. Gidora, M. Foy, and J. Rosenfeld. 2014. Watershed-scale effectiveness of floodplain habitat restoration for juvenile coho salmon in the Chilliwack River, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 72(4):479–490.

Österblom, H., O. Olsson, T. Blenckner, and R. W. Furness. 2008. Junk-food in marine ecosystems. Oikos 117(7):967–977.

Ostrom, E. 2002. Common-pool resources and institutions: Toward a revised theory. Handbook of Agricultural Economics 2:1315–1339.

Ostrom, E., and X. Basurto. 2011. Crafting analytical tools to study institutional change. Journal of Institutional Economics 7(3):317–343.

Pascoe, S., C. Wilcox, and C. J. Donlan. 2011. Biodiversity offsets: A cost-effective interim solution to seabird bycatch in fisheries? PLOS One 6(10):e25762.

Paulsen, C. M., and K. Wernstedt. 1995. Cost-effectiveness analysis for complex managed hydrosystems: an application to the Columbia River basin. Journal of Environmental Economics and Management 28(3):388–400.

Peckham, S. H., D. M. Diaz, A. Walli, G. Ruiz, L. B. Crowder, and W. J. Nichols. 2007. Small-scale fisheries bycatch jeopardizes endangered Pacific loggerhead turtles. PLOS One 2(10):e1041.

Peckham, S. H., J. Lucero-Romero, D. Maldonado-Díaz, A. Rodríguez-Sánchez, J. Senko, M. Wojakowski, and A. Gaos. 2016. Buoyless Nets Reduce Sea Turtle Bycatch in Coastal Net Fisheries. Conservation Letters 9(2):114–121.

Pfaff, A., and J. Robalino. 2017. Spillovers from Conservation Programs. Annual Review of Resource Economics 9:299–315. DOI: 10.1146/annurev-resource-100516-053543

Pinsky, M. L., and N. J. Mantua. 2014. Emerging adaptation approaches for climate-ready fisheries management. Oceanography 27(4):146–159.

Pirotta, E., and D. Lusseau. 2015. Managing the wildlife tourism commons. Ecological Applications 25(3):729–741.

Plantinga, A. J., T. L. Helvoigt, and K. Walker. 2014. Critical habitat for threatened and endangered species: How should the economic costs be evaluated? Journal of Environmental Management 134:127–135.

Polovina, J. J., G. H. Balazs, E. A. Howell, D. M. Parker, M. P. Seki, and P. H. Dutton. 2004. Forage and migration habitat of loggerhead (*Caretta caretta*) and olive ridley (*Lepidochelys olivacea*) sea turtles in the central North Pacific Ocean. Fisheries Oceanography 13(1):36–51.

Polovina, J. J., D. R. Kobayashi, D. M. Parker, M. P. Seki, and G. H. Balazs. 2000. Turtles on the edge: Movement of loggerhead turtles (*Caretta caretta*) along oceanic fronts, spanning longline fishing grounds in the central North Pacific, 1997–1998. Fisheries Oceanography 9(1):71–82.

Raimondi, P., L. J. Jurgens, and M. T. Tinker. 2015. Evaluating potential conservation conflicts between two listed species: Sea otters and black abalone. Ecology 96(11):3102–3108.

Raum-Suryan, K. L., L. A. Jemison, and K. W. Pitcher. 2009. Entanglement of Steller sea lions (*Eumetopias jubatus*) in marine debris: Identifying causes and finding solutions. Marine Pollution Bulletin 58(10):1487–1495.

Reece, J. S., D. Passeri, L. Ehrhart, S. C. Hagen, A. Hays, C. Long, R. F. Noss, M. Bilskie, C. Sanchez, and M. V. Schwoerer. 2013. Sea level rise, land use, and climate change influence the distribution of loggerhead turtle nests at the largest USA rookery (Melbourne Beach, Florida). Marine Ecology Progress Series 493:259–274.

Rogers-Bennett, L., P. L. Haaker, T. O. Huff, and P. K. Dayton. 2002. Estimating baseline abundances of abalone in California for restoration. Reports of California Cooperative Oceanic Fisheries Investigations 43:97–111.

Rosen, D. A., and A. W. Trites. 2000. Pollock and the decline of Steller sea lions: Testing the junk-food hypothesis. Canadian Journal of Zoology 78(7):1243–1250.

Rosen, D. A., and A. W. Trites. 2005. Examining the potential for nutritional stress in young Steller sea lions: Physiological effects of prey composition. Journal of Comparative Physiology B 175(4):265–273.

Ross, P. S., M. Noël, D. Lambourn, N. Dangerfield, J. Calambokidis, and S. Jeffries. 2013. Declining concentrations of persistent PCBs, PBDEs, PCDEs, and PCNs in harbor seals (*Phoca vitulina*) from the Salish Sea. Progress in Oceanography 115:160–170.

Ruckelshaus, M. H., P. Levin, J. B. Johnson, and P. M. Kareiva. 2002. The Pacific salmon wars: What science brings to the challenge of recovering species. Annual Review of Ecology and Systematics 33(1):665–706.

Samhouri, J. F., A. C. Stier, S. M. Hennessey, M. Novak, B. S. Halpern, and P. S. Levin. 2017. Rapid and direct recoveries of predators and prey through synchronized ecosystem management. Nature Ecology & Evolution 1(4):0068.

Sanchirico, J. N., D. K. Lew, A. C. Haynie, D. M. Kling, and D. F. Layton. 2013. Conservation values in marine ecosystem-based management. Marine Policy 38:523–530.

Santidrián Tomillo, P., E. Vélez, R. D. Reina, R. Piedra, F. V. Paladino, and J. R. Spotila. 2007. Reassessment of the Leatherback Turtle (*Dermochelys coriacea*) Nesting Population at Parque Nacional Marino Las Baulas, Costa Rica: Effects of Conservation Efforts. Chelonian Conservation and Biology 6(1):54–62.

Sawchuk, J. H., A. H. Beaudreau, D. Tonnes, and D. Fluharty. 2015. Using stakeholder engagement to inform endangered species management and improve conservation. Marine Policy 54:98–107.

Scheld, A., D. Bilkovic, and K. Havens. 2016. The dilemma of derelict gear. Scientific Reports 6:19671.

Scheuerell, M. D., R. Hilborn, M. H. Ruckelshaus, K. K. Bartz, K. M. Lagueux, A. D. Haas, and K. Rawson. 2006. The Shiraz model: A tool for incorporating anthropogenic effects and fish-habitat relationships in conservation planning. Canadian Journal of Fisheries and Aquatic Sciences 63(7):1596–1607.

Segerson, K. 2011. Policies to reduce stochastic sea turtle bycatch. Pages 370–395 *in* P. Dutton, D. Squires, and M. Ahmed, editors. Conservation of Pacific Sea Turtles. University of Hawaii Press, Honolulu, Hawaii.

Seminoff, J. 2004. Marine Turtle Specialist Group Review: 2004 Global Status Assessment, Green Turtle (*Chelonia mydas*). Marine Turtle Specialist Group. The World Conservation Union (IUCN) Species Survival Commission Red List Programme, La Jolla, California. Available: mtsg.files.wordpress.com/2010/07/mtsg_chelonia_mydas_assessment_expanded-format.pdf (October 2018).

Seminoff, J., C. Allen, G. Balazs, P. Dutton, T. Eguchi, H. Hass, S. Hargrove, M. Jensen, D. Klemm, A. Lauritsen, S. MacPherson, P. Opay, E. Possardt, S. Pultz, E. Seney, K. Van Houtan, and R. Waples. 2015. Status Review of the Green Turtle (*Chelonia mydas*) Under the Endangered Species Act. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SWFSC-539.

Senigaglia, V., F. Christiansen, L. Bejder, D. Gendron, D. Lundquist, D. Noren, A. Schaffar, J. Smith, R. Williams, and E. Martinez. 2016. Meta-analyses of whale-watching impact studies: Comparisons of cetacean responses to disturbance. Marine Ecology Progress Series 542:251–263.

Shogren, J. F., J. Tschirhart, T. Anderson, A. W. Ando, S. R. Beissinger, D. Brookshire, G. M. Brown, D. Coursey, R. Innes, and S. M. Meyer. 1999. Why economics matters for endangered species protection. Conservation Biology 13(6):1257–1261.

Sigler, M. F., D. J. Tollit, J. J. Vollenweider, J. F. Thedinga, D. J. Csepp, J. N. Womble, M. A. Wong, M. J. Rehberg, and A. W. Trites. 2009. Steller sea lion foraging response to seasonal changes in prey availability. Marine Ecology Progress Series 388:243–261.

Simmonds, M. P. 2012. Cetaceans and marine debris: The great unknown. Journal of Marine Biology 2012:684279. DOI: 10.1155/2012/684279

Singh, R., and Q. Weninger. 2015. Cap-and-trade bycatch management with costly avoidance and stock uncertainty. Marine Resource Economics 30(1):97–119.

Smith, M. D., F. Asche, L. S. Bennear, E. Havice, A. J. Read, and D. Squires. 2014. Will a catch share for whales improve social welfare? Ecological Applications 24(1):15–23.

Southall, B. L. 2005. Shipping Noise and Marine Mammals: A Forum for Science, Management, and Technology. Final report of the National Oceanic and Atmospheric Administration (NOAA) International Symposium 2005. National Marine Fisheries Service, Arlington, Virginia.

Speir, C., S. Pittman, and D. Tomberlin. 2015. Uncertainty, Irreversibility and the Optimal Timing of Large-Scale Investments in Protected Species Habitat Restoration. Frontiers in Marine Science 2:101.

Stephenson, R. L., A. J. Benson, K. Brooks, A. Charles, P. Degnbol, C. M. Dichmont, M. Kraan, S. Pascoe, S. D. Paul, and A. Rindorf. 2017. Practical steps toward integrating economic, social and institutional elements in fisheries policy and management. ICES Journal of Marine Science 74(7):1981–1989.

Stierhoff, K. L., M. Neuman, and J. L. Butler. 2012. On the road to extinction? Population declines of the endangered white abalone, *Haliotis sorenseni*. Biological Conservation 152:46–52.

Strange, E. L. 2016. Assessing Southern Resident killer whale (*Orcinus orca*) prey abundance: The effects of Chinook salmon (*Oncorhynchus tshawystcha*) ocean fishery reductions and related hatchery production. California State University Press, Sacramento, California.

Tapilatu, R. F., P. H. Dutton, M. Tiwari, T. Wibbels, H. V. Ferdinandus, W. G. Iwanggin, and B. H. Nugroho. 2013. Long-term decline of the western Pacific leatherback, *Dermochelys coriacea*: A globally important sea turtle population. Ecosphere 4(2):1–15.

Tomberlin, D. 2010. Endangered seabird habitat management as a partially observable Markov decision process. Marine Resource Economics 25(1):93–104.

Trites, A. W., A. J. Miller, H. D. Maschner, M. A. Alexander, S. J. Bograd, J. A. Calder, A. Capotondi, K. O. Coyle, E. D. Lorenzo, and B. P. Finney. 2007. Bottom-up forcing and the decline of Steller sea lions (*Eumetopias jubatus*) in Alaska: Assessing the ocean climate hypothesis. Fisheries Oceanography 16(1):46–67.

USFWS (U.S. Fish and Wildlife Service). 2014. Olive Ridley sea turtle 5-year review: Summary and evaluation. U.S. Fish and Wildlife Service, Jacksonville, Florida.

USOFR (U.S. Office of the Federal Register). 1996. Policy Regarding the Recognition of Distinct Vertebrate Population Segments Under the Endangered Species Act. Federal Register 61:26(7 February 1996):4722–4725.

USOFR (U.S. Office of the Federal Register). 2016. Fisheries of the Exclusive Economic Zone Off Alaska; Bycatch Management in the Bering Sea Pollock Fishery, final rule (50 CFR Part 679). Federal Register 81:112(10 June 2016):37534–37556.

Veirs, S., V. Veirs, and J. D. Wood. 2016. Ship noise extends to frequencies used for echolocation by endangered killer whales. PeerJ 4:e1657.

Waldron, A., D. C. Miller, D. Redding, A. Mooers, T. S. Kuhn, N. Nibbelink, J. T. Roberts, J. A. Tobias, and J. L. Gittleman. 2017. Reductions in global biodiversity loss predicted from conservation spending. Nature 551(7680):364.

Wallace, B. P., S. S. Heppell, R. L. Lewison, S. Kelez, and L. B. Crowder. 2008. Impacts of fisheries bycatch on loggerhead turtles worldwide inferred from reproductive value analyses. Journal of Applied Ecology 45(4):1076–1085.

Wallace, B. P., C. Y. Kot, A. D. DiMatteo, T. Lee, L. B. Crowder, and R. L. Lewison. 2013. Impacts of fisheries bycatch on marine turtle populations worldwide: toward conservation and research priorities. Ecosphere 4(3):1–49.

Wallace, B. P., R. L. Lewison, S. L. McDonald, R. K. McDonald, C. Y. Kot, S. Kelez, R. K. Bjorkland, E. M. Finkbeiner, and L. B. Crowder. 2010. Global patterns of marine turtle bycatch. Conservation Letters 3(3):131–142.

Wallmo, K., and D. K. Lew. 2012. Public willingness to pay for recovering and downlisting threatened and endangered marine species. Conservation Biology 26(5):830–839.

Wang, J., J. Barkan, S. Fisler, C. Godinez-Reyes, and Y. Swimmer. 2013. Developing ultraviolet illumination of gillnets as a method to reduce sea turtle bycatch. Biology letters 9(5):20130383.

Wang, J. H., S. Fisler, and Y. Swimmer. 2010. Developing visual deterrents to reduce sea turtle bycatch in gill net fisheries. Marine Ecology Progress Series 408:241–250.

Waples, R. S. 1991. Pacific salmon, *Oncorhynchus* spp., and the definition of "species" under the Endangered Species Act. Marine Fisheries Review 53(3):11–22.

Waples, R. S., M. J. Ford, and D. Schmitt. 2007. Empirical results of salmon supplementation in the Northeast Pacific: A preliminary assessment. Pages 383–403 *in* T. M. Bert, editor. Ecological and genetic implications of aquaculture activities. Springer Netherlands. DOI: 10.1007/978-1-4020-6148-6

Ward, E. J., M. J. Ford, R. G. Kope, J. K. Ford, L. A. Vélez-Espino, C. K. Parken, L. LaVoy, M. B. Hanson, and K. C. Balcomb. 2013. Estimating the Impacts of Chinook Salmon Abundance and Prey Removal by Ocean Fishing on Southern Resident Killer Whale Population Dynamics. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-123.

Ward, E. J., J. E. Jannot, Y.-W. Lee, K. Ono, A. O. Shelton, and J. T. Thorson. 2015. Using spatiotemporal species distribution models to identify temporally evolving hotspots of species co-occurrence. Ecological Applications 25(8):2198–2209.

Wasser, S. K., J. I. Lundin, K. Ayres, E. Seely, D. Giles, K. Balcomb, J. Hempelmann, K. Parsons, and R. Booth. 2017. Population growth is limited by nutritional impacts on pregnancy success in endangered Southern Resident killer whales (*Orcinus orca*). PLOS One 12(6):e0179824.

Watanabe, M., R. M. Adams, and J. Wu. 2006. Economics of environmental management in a spatially heterogeneous river basin. American Journal of Agricultural Economics 88(3):617–631.

Watanabe, M., R. M. Adams, J. Wu, J. P. Bolte, M. M. Cox, S. L. Johnson, W. J. Liss, W. G. Boggess, and J. L. Ebersole. 2005. Toward efficient riparian restoration: Integrating economic, physical, and biological models. Journal of Environmental Management 75(2):93–104.

Watson, J. W., S. P. Epperly, A. K. Shah, and D. G. Foster. 2005. Fishing methods to reduce sea turtle mortality associated with pelagic longlines. Canadian Journal of Fisheries and Aquatic Sciences 62(5):965–981.

Weber, M. A. 2015. Navigating benefit transfer for salmon improvements in the western US. Frontiers in Marine Science 2:74.

White, C., C. Costello, B. E. Kendall, and C. J. Brown. 2012. The value of coordinated management of interacting ecosystem services. Ecology Letters 15(6):509–519.

Wilcox, C., and C. J. Donlan. 2007. Compensatory mitigation as a solution to fisheries bycatch–biodiversity conservation conflicts. Frontiers in Ecology and the Environment 5(6):325–331.

Wilen, J. E., J. Cancino, and H. Uchida. 2012. The economics of territorial use rights fisheries, or TURFs. Review of Environmental Economics and Policy 6(2):237–257.

Williams, G. D., P. S. Levin, and W. A. Palsson. 2010. Rockfish in Puget Sound: An ecological history of exploitation. Marine Policy 34(5):1010–1020.

Williams, R., E. Ashe, and P. D. O'Hara. 2011. Marine mammals and debris in coastal waters of British Columbia, Canada. Marine Pollution Bulletin 62(6):1303–1316.

Williams, R., D. E. Bain, J. C. Smith, and D. Lusseau. 2009. Effects of vessels on behaviour patterns of individual southern resident killer whales *Orcinus orca*. Endangered Species Research 6(3):199–209.

Williamson, O. E. 2000. The new institutional economics: Taking stock, looking ahead. Journal of Economic Literature 38(3):595–613.

Willson, M., R. Armstrong, M. Hermans, and K. Koski. 2006. Eulachon: A review of biology and an annotated bibliography. U.S. Department of Commerce, AFSC Processed Report 2006-12. Available: www.afsc.noaa.gov/publications/ProcRpt/PR%202006-12.pdf (October 2018).

Ylitalo, G. M., J. E. Stein, T. Hom, L. L. Johnson, K. L. Tilbury, A. J. Hall, T. Rowles, D. Greig, L. J. Lowenstine, and F. M. Gulland. 2005. The role of organochlorines in cancer-associated mortality in California sea lions (*Zalophus californianus*). Marine Pollution Bulletin 50(1):30–39.

Yu, R., M. Pan, S. F. Railsback, and P. Leung. 2009. Exploratory application of agent based modeling in Hawaii's longline fishery. Page 20 *in* Proceedings of the 2009 Spring Simulation Multiconference. Society for Computer Simulation International, San Diego, California.

Žydelis, R., R. L. Lewison, S. A. Shaffer, J. E. Moore, A. M. Boustany, J. J. Roberts, M. Sims, D. C. Dunn, B. D. Best, and Y. Tremblay. 2011. Dynamic habitat models: Using telemetry data to project fisheries bycatch. Proceedings of the Royal Society of London B: Biological Sciences 278(1722):3191–3200.

Žydelis, R., B. P. Wallace, E. L. Gilman, and T. B. Werner. 2009. Conservation of marine megafauna through minimization of fisheries bycatch. Conservation Biology 23(3):608–616.

Recently published by the Northwest Fisheries Science Center

NOAA Technical Memorandum NMFS-NWFSC-

143 **Harsch, M., L. Pfeiffer, E. Steiner, and M. Guldin. 2018.** Economic Performance Metrics: An Overview of Metrics and the Use of Web Applications to Disseminate Outcomes in the U.S. West Coast Groundfish Trawl Catch Share Program. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-143. NTIS number PB2019-100087. https://doi.org/10.25923/a4g5-cq83

142 **Jannot, J. E., T. Good, V. Tuttle, A. M. Eich, and S. Fitzgerald, editors. 2018.** U.S. West Coast and Alaska Trawl Fisheries Seabird Cable Strike Mitigation Workshop, November 2017: Summary Report. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-142. NTIS number PB2018-101082. https://doi.org/10.7289/V5/TM-NWFSC-142

141 **McClure, M., J. Anderson, G. Pess, T. Cooney, R. Carmichael, C. Baldwin, J. Hesse, L. Weitkamp, D. Holzer, M. Sheer, and S. Lindley. 2018.** Anadromous Salmonid Reintroductions: General Planning Principles for Long-Term Viability and Recovery. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-141. NTIS number PB2018-101081. https://doi.org/10.7289/V5/TM-NWFSC-141

140 **Buhle, E. R., M. D. Scheuerell, T. D. Cooney, M. J. Ford, R. W. Zabel, and J. T. Thorson. 2018.** Using Integrated Population Models to Evaluate Fishery and Environmental Impacts on Pacific Salmon Viability. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-140. NTIS number PB2018-101080. https://doi.org/10.7289/V5/TM-NWFSC-140

139 **Harvey, C., N. Garfield, G. Williams, K. Andrews, C. Barceló, K. Barnas, S. Bograd, R. Brodeur, B. Burke, J. Cope, L. deWitt, J. Field, J. Fisher, C. Greene, T. Good, E. Hazen, D. Holland, M. Jacox, S. Kasperski, S. Kim, A. Leising, S. Melin, C. Morgan, S. Munsch, K. Norman, W. T. Peterson, M. Poe, J. Samhouri, I. Schroeder, W. Sydeman, J. Thayer, A. Thompson, N. Tolimieri, A. Varney, B. Wells, T. Williams, and J. Zamon. 2017.** Ecosystem Status Report of the California Current for 2017: A Summary of Ecosystem Indicators Compiled by the California Current Integrated Ecosystem Assessment Team (CCIEA). U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-139. NTIS number PB2018-100477. https://doi.org/10.7289/V5/TM-NWFSC-139

138 **Kamikawa, D. J. 2017.** Survey Fishes: An Illustrated List of the Fishes Captured during the Northwest Fisheries Science Center's Fishery Resource Analysis and Monitoring Division's West Coast Surveys. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-138. NTIS number PB2018-100308. https://doi.org/10.7289/V5/TM-NWFSC-138

137 **Beechie, T. J., O. Stefankiv, B. Timpane-Padgham, J. E. Hall, G. R. Pess, M. Rowse, M. Liermann, K. Fresh, and M. J. Ford. 2017.** Monitoring Salmon Habitat Status and Trends in Puget Sound: Development of Sample Designs, Monitoring Metrics, and Sampling Protocols for Large River, Floodplain, Delta, and Nearshore Environments. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-137. NTIS number PB2017-102556. https://doi.org/10.7289/V5/TM-NWFSC-137

136 **Keller, A. A., J. R. Wallace, and R. D. Methot. 2017.** The Northwest Fisheries Science Center's West Coast Groundfish Bottom Trawl Survey: History, Design, and Description. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-136. NTIS number PB2017-101432. https://doi.org/10.7289/V5/TM-NWFSC-136

NOAA Technical Memorandums NMFS-NWFSC are available at the Northwest Fisheries Science Center website, https://www.nwfsc.noaa.gov/index.cfm.