

# Marine Historical Ecology in Conservation

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*Applying the Past to Manage for the Future*

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Foreword by Daniel Pauly



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## FOREWORD

*Marine Historical Ecology in Conservation*, the title of this book, may be hard on potential readers, in that each of its two nouns and two adjectives can be seen as potential challenges:

- “Ecology,” because some find it difficult to distinguish the scientific discipline of ecology from the passion of environmentalism;
- “Historical,” because until recently, many academic ecologists suffering from physics envy were attempting to ban history and contingency from ecology;
- “Marine,” because we are air-breathing, terrestrial animals with a strong bias against the watery world that covers most of the surface of our ill-named planet; and finally,
- “Conservation,” because the word implies, for still too many, a departure from what scientists are supposed to do (describe our world, as opposed to changing it, or in this case, developing the tools to prevent it from being dismantled).

Why do we need marine historical ecology and conservation? The fact is that since Darwin’s *On the Origin of Species*, we have become quite good at inferring what existed—in terms of animals and plants—if only because we have (a) fossils and (b) a powerful theory which allows, nay demands, that we interpolate between the forms we know existed, because we have fossils, and the forms for which we have no direct evidence but which we can link to present forms, including us humans.

Thus, in a sense, we know most of what *was there* since the Cambrian, and this knowledge becomes more precise and accurate the closer we come to the present. However, we don’t know *how much* of what was there actually was there, and this may be seen as the defining feature of historical ecology and its potential use in marine conservation.

One way to view this is that while evolution’s “central casting” provides us with a reliable stable of actors (e.g., a wide range of dinosaurs in the Triassic or a flurry of mammals in the

Pleistocene), it is for historical ecology to give them roles to play. (Note that these examples imply that historical ecology should mean the ecology of past systems and not only past ecology as recoverable through written documents, as one could assume when relying on a narrow interpretation of the word “history”.)

Thus, an ecosystem with, say, sea turtles in it will function in a radically different way if these turtles are very abundant (as they appear to have been, e.g., in the pre-Columbian Caribbean) than it will where sea turtles are marginal, as is now the case in the Caribbean.

The Earth’s ecosystems have all been modified by human activities, and this applies also to essentially all marine ecosystems, which whaling and hunting of other marine mammals, and later fishing, have reduced to shadows of their former selves in terms of the larger organisms they now support and the benefits they can provide us.

Some of these ecosystem modifications were unavoidable, as humans living on coastlines are largely incompatible with large populations of, say, sturgeons, sea turtles, or pinnipeds, and our appetite for fish implies that some fish populations will have to be reduced by fishing. But to a large extent, the depredations that we have imposed on the oceans have been entirely gratuitous: we need not have eradicated the great auk (*Pinguinus impennis*) or the Caribbean monk seal (*Monachus tropicalis*) to satisfy our seafood requirements, and thus it is perfectly reasonable to ask ourselves how we could prevent such catastrophes in the future (each species loss is a catastrophe) and whether we can rebuild now depleted populations of marine organisms so as to reduce the risk of this occurring again, and to have more to enjoy.

This is what marine historical ecology in conservation is for: to inform us about what these populations have been in the past, and under which conditions these populations could flourish so that we can start helping them do so. This is what the neat book you have in your hands is about.

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August 2013

## Improving Fisheries Assessments Using Historical Data

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### Stock Status and Catch Limits

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The health of marine fish stocks is inherently difficult to assess because catches are only partially recorded and abundance cannot be directly observed. Understanding the current status of stocks requires an estimate of what the stock is capable of producing in the absence of fishing, yet fisheries data almost never extend back to pre-exploitation states. Without catch and abundance estimates across a range of fish densities, it can be exceedingly difficult to estimate the productive capacity of a fishery, or to develop reference points to approximate this capacity. Historical data (e.g., historical records, archaeological information, geological records, ecological reconstructions, local ecological knowledge, and traditional ecological knowledge) provide unconventional opportunities to develop more realistic reference points and examine stock status prior to large-scale intensive fishing. These data can help managers avoid the pitfalls of the “shifting baseline syndrome,” in which conventional stock-rebuilding programs are heavily influenced by the most recent peak in productivity. Such approaches can also be productively employed in small-scale fisheries, where standard stock-assessment techniques and assumptions are not applicable and where historically based analyses can provide valid scientific advice to guide management decisions. This chapter focuses on how historical data can inform non-traditional fishery-assessment methods, using case studies from small-scale tropical fisheries, which present particularly complex assessment challenges due to the large number of species exploited, the wide variety of gear employed, and the diffuse nature of fishing locations and landing sites. A growing number of communities around the world are combining historical data and locally situated knowledge systems such as local ecological knowledge and traditional ecological knowledge to assess and manage fish stock status. By incorporating these data into population assessment models and management practices, we gain insight into the yield of these ecosystems in the past and provide guidance for future management actions.

## INTRODUCTION TO STOCK ASSESSMENT MODELS AND METHODS

The main tool for gauging the health of a fish stock is the statistical stock assessment model. It fits biological parameters, including life-history traits such as size-at-age, age, sex, recruitment rates, and natural mortality, to a predefined model structure using measures of the relative abundance of fish and the quantity of fish caught over time. Models identify the biological growth and mortality required of a population to explain observed changes in abundance, given the quantity of fish removed by fishing. These growth patterns are then compared across different stock abundances to estimate the effects of fish density on productivity. Together, the growth and mortality patterns, as modulated by density, allow predictions of how the stock will respond to fishing pressure and, thus, form the basis for policy guidance, such as total allowable catch limits (Clark 1990, Quinn and Deriso 1999).

Time plays a crucial role in stock assessment models. Catch data are often used to calculate abundance parameters, and a long history of data provides insight over a greater range of conditions and potentially reduces variance in models of a stock's productive capacity and response to fishing. However, data spanning the history of a fishery are rare. Usually, data-collection programs are not enacted until a fishery has already matured and been fished well below its pristine (pre-exploitation) abundance level. This nearly universal data gap causes problems, which historical information can help address.

At first approximation, growth and mortality patterns can be described using three key reference points, including (1) the pristine abundance ( $B_0$ ), typically measured as biomass that the stock may have achieved in the absence of fishing; (2) the abundance of the stock that would sustain maximum sustainable yields ( $B_{MSY}$ ); and (3) the associated fishing mortality rate ( $F_{MSY}$ ). Maximum sustainable yields are associated with intermediate abundance levels, often less than half of  $B_0$ . Low abundance can affect a stock's productivity through a lack of individuals to reproduce. High stock abundance can also affect overall productivity, because of density-dependent factors such as competition, disease, and predation. To understand the challenges in establishing these reference points, it is helpful to look in greater detail at how they are estimated from catch measures and abundance indices.

### Data Sources and Limitations

Fisheries data are fraught with gaps and uncertainties (Haltuch et al. 2009, Gårdmark et al. 2011). Although catches are arguably the easiest component of fisheries to measure, such data carry many challenges. In fact, it is exceptionally rare that catches are fully recorded. Some fleets carry observers who record all fish caught under their watch. However, observer programs cover only a few large-boat industrial fisheries, and even in these fisheries, observers typically cover only a small percentage of all trips conducted. Large-boat industrial fisheries are often characterized by a few easily monitored landing sites that allow for dockside measurements of fish brought to port. If there are no observers, we also have to estimate at-sea discards, which can be substantial and are often underestimated (Hall and Mainprize 2005, Harrington et al. 2005). When it comes to small-boat commercial, artisanal, and

recreational fisheries, even landings are difficult to monitor because of the diffuse nature of fishing and landing sites (Galluci et al. 1996, Pauly 2006). For these fisheries, we often rely on self reporting, and it is common for these reports to lump multiple species into market categories because of the difficulty of identification or the burden of reporting when many species are caught simultaneously. Even some large-boat fisheries contribute to missing catch data via illegal, unreported, and unregulated (IUU) fishing (Bray 2000).

If catch seems difficult to measure accurately, fish abundance is even harder to estimate. In some fisheries, scientists perform random scientific sampling, which provides a fishery-independent index of abundance. These data can be highly variable as a result of patchy habitats and factors such as seasonal variability in fish populations. Surveys provide better guidance when performed regularly over wide areas and with consistent methods. While many large-scale industrial fisheries have such indices that stretch back many years, they are rare in small-scale fisheries, particularly those in diverse tropical regions. Every index we are aware of started after the fisheries were already well developed or, in some cases, overexploited. When fishery-independent indices are absent or weak, we typically rely on fishery-dependent measures of abundance. To do so, we use catch per unit effort (CPUE), under the assumption that a more abundant stock will yield a higher CPUE. After standardizing effort, we typically assume that CPUE is directly proportional to abundance. Thus, measuring and standardizing effort are essential for fisheries that lack well-developed fishery-independent indices.

If ships carry observers, effort can be directly measured and categorized. Few do, but there are some techniques to estimate effort in the absence of observers. Dockside monitoring can provide data on the number of days fishing, the crew size, and the type of gear used. However, these measures are crude approximations of the actual effort expended and are particularly lacking in information to standardize effort. Visual surveys (e.g., aerial, satellite, land, and boat-based observers) can monitor fishing activity close to shore, but these are rare and expensive. An emerging technique, available only to some fisheries (typically large-scale industrial fleets), is the use of vessel-monitoring systems, which provide real-time location and movement of registered fishing vessels and sometimes include video monitoring of the catch.

However, for most fisheries, particularly small-scale and data-poor ones, we must depend on self reporting. Often, these reports are limited to days at sea, crew size, and gear employed. These limits make it challenging to standardize effort. CPUE will vary, sometimes dramatically, depending on the location and timing of fishing, and on the detailed configuration and deployment of fishing gear. Standardizing CPUE indices requires taking into account spatiotemporal patterns, which range from broad-scale (e.g., latitude and season) to fine-scale (e.g., depth, habitat, and time of day), and accounting for differences in gear type, configuration, and deployment. Without information to guide standardization, the quantification of effective effort may contain large errors and lead to inappropriate conclusions.

Recall that the ultimate goal of a stock assessment is the statistical estimation of the growth potential of the fishery. Poor data can make such estimations impossible. For

example, undocumented increases in the efficiency of a fleet can lead to seemingly higher CPUE at the same time that catches are increasing. In this case, we may not be able to fit sensible parameters to a model built on the assumption that increasing catches should lead to lower stock abundance and CPUE. With all these data limitations and challenges, we often cannot directly estimate fisheries productivity and must rely on proxies for the above-mentioned reference points: unfished biomass ( $B_0$ ), maximally productive biomass ( $B_{MSY}$ ), and maximally productive fishing mortality rate ( $F_{MSY}$ ).

#### Current State of Knowledge and Practice: Ongoing Uncertainty

Given the complexities of fisheries and data limitations, it is not surprising that the global status of fisheries is poorly understood. What is most striking about the state of world fisheries is how little we know. In the United States, the overfished condition, overfishing status, or both remain unknown for over half of all stocks under federal management (National Marine Fisheries Service 2012). For nonpelagic (primarily reef) vertebrate fisheries under federal management in the tropical United States, 75% are unknown (National Marine Fisheries Service 2012). Globally, the problem is almost certainly far greater because of resource and data limitations associated with fisheries (e.g., Beddington et al. 2007, Costello et al. 2012).

There are a number of reasons why stocks go unassessed. In the United States, it is common for formal assessments to be conducted only every 3–6 years because of limited resources, although most stocks are reviewed annually (National Marine Fisheries Service 2012). Elsewhere around the world, the capacity for formal assessments is more limiting. A frequent cause of unassessed stocks is a paucity of adequate conventional fisheries data. In some cases, data exist but are inadequate to inform a stock assessment model, because of either data gaps or contradictory trends. In other cases, the data simply do not exist. For example, when catches are reported by market categories rather than by species, species-level assessments are not possible. Scientists are making laudable efforts to assess mixed stock complexes (e.g., Hutchinson 2008, McClanahan et al. 2011), but these efforts are fraught with problems. In particular, multispecies assessments allow the relatively weaker stocks within a complex to suffer the brunt of the effects of fishing pressure while their decline goes undetected (Hilborn et al. 2004).

In summary, stock assessment efforts are plagued by a scarcity of conventional data and constrained by methods that were developed primarily for large-scale and relatively data-rich industrial fisheries. Most fisheries worldwide do not fit the assumptions embedded in these methods (Ruddle and Hickey 2008, Fenner 2012). To adequately assess more stocks, we need to explore the promise of unconventional data sources and develop and implement techniques to make better use of them in fisheries management practice.

#### ALTERNATIVES TO CURRENT PRACTICES

Many promising alternatives, based in part on greater incorporation and use of historical data, have emerged as viable alternatives to conventional stock assessment. Below, we first

describe a set of complex modeling efforts, which aim to use ecological relationships as a way of inferring details of important fish populations. Some of this work has specifically focused on recreating past ecosystems by following energy flows and species interactions. Next, we review key studies that used historical information, rather than models (and assumptions), to gain a perspective on the productive potential of fish stocks. Third, we explore the information that can be gleaned from traditional ecological knowledge and local ecological knowledge, because fishing communities often know a great deal about the ecosystems that sustain them. We then look at geological and archaeological evidence for estimating historical abundance, showing how these records allow us to assess the development of fisheries and characterize natural cycles in fishery populations. Next, we discuss the option of using the biomass of pristine unfished areas as a reference point, using a space-for-time substitution approach. Finally, we examine data-limited management and review historical management practices, illustrating how fisheries were sustained for centuries using some simple techniques.

### Simulation Modeling

Rebuilding fish stocks implies reconstructing elements of past ecosystems. A trophic mass-balance model, ECOPATH (Polovina 1984), and two derived dynamic simulations, ECOSIM (Walters et al. 1997) and ECOSPACE (Walters et al. 1998, 1999), are some of the most widely used tools to model past ecosystems. Using data on fisheries catch by sector, production-to-biomass ratios, consumption rates, and a diet matrix for up to 50 defined components of an ecosystem, ECOPATH tallies the flows of matter within the components of a system, defines trophic levels, and can be used to estimate biomass per trophic level, given diet, mortality, and consumption rates (Christensen and Pauly 1992, 1993). ECOSIM evaluates the impact of changes in fishing rates selectively across gear types, and investigators can tune the model to defined time series for biomass estimates (Christensen and Walters 2004). ECOSPACE allows investigators to engage in spatial ecosystem modeling by replicating the ECOSIM simulations across a grid of habitat cells (Walters et al. 1999).

An interesting reconstruction process that utilizes this EcoPath simulation package is “Back to the Future” (BTF), which employs traditional ecological knowledge and local ecological knowledge, historical documentation, and archaeology to facilitate ecological modeling of past systems and uses these states to help provide policy goals for the future (Pitcher 2001, 2005). For example, a reconstruction of the Strait of Georgia, British Columbia, marine ecosystem was conducted for (1) the present day, (2) 100 yr BP, and (3) 500 yr BP (Pitcher 1998). Results from this work highlight the enormous changes in abundance, size, and composition of fisheries populations over a 500-year period and show how a long-term approach is essential for determining the natural productivity of an ecosystem (Dalsgaard et al. 1998). This methodology has been applied to a variety of ecosystems in locations ranging from Hong Kong (Buchary et al. 2003) to Newfoundland (Pitcher et al. 2002) and northern British Columbia (Ainsworth et al. 2002), all with similar outcomes, revealing evidence of dramatic declines in biomass and shifts to lower trophic levels across these diverse systems.

Although complex models are tempting as a means to generate reference points, particularly historical ones, we nevertheless must use caution. ECOPATH and other complex models rely on data to inform them, just like conventional fisheries models. Unlike conventional models, ECOPATH incorporates ecological interactions and thus adds complexity, which requires additional data needs. Although we sometimes have some information about the type, strength, and direction of ecological interactions within a food web, this information is frequently missing. Researchers can nevertheless fill in data gaps via assumptions. Typically, a modeler will examine the behavior of their modeled ecosystem and then tweak these assumptions until the model's behavior seems plausible, a process referred to as "tuning." As a reality check, we must remember that tuning is subjective and driven by assumptions rather than observation. Thus, we recommend using complex models as a tool for generating testable hypotheses and focusing on empirical evidence to learn about the historical capacity of ecosystems. Fortunately, we have a growing toolbox for doing so.

#### Historical Catch Records

Historical records give us a rare opportunity to look into the past without complicated methods or dangerous assumptions. For some fisheries, it is possible to use historical sources to examine trends over long periods (e.g., decades to centuries) and estimate biomass in the early stages of fisheries exploitation. By taking a long-term view, we can provide more realistic insights into the past productivity of these ecosystems, not just recently observed catch levels (Rosenberg et al. 2005). A shortcoming of this approach is that it can be difficult to find records that date back to the early stages of a fishery, let alone enough reliable data to reconstruct catches. Despite this limitation, there are numerous examples of the utility of historical catch records in estimating past stock abundances.

#### *Case 1: Cod Fishery in the Western North Atlantic*

Commercial fishing for cod in the western north Atlantic dates back to the 1500s, when Basque fishermen discovered the Grand Banks off Newfoundland (Kurlansky 1997). Over the next 300 years, the fishing industry thrived, with many periods of low and high production. Using detailed catch logs from cod fishing schooners based in Beverly, Massachusetts, from 1852 to 1859, Rosenberg et al. (2005) were able to reconstruct biomass of cod for the Scotian Shelf, Canada, and compare these results to present-day estimates of cod standing stock for the same area.

Estimates of adult biomass of cod from 1852 (~1.3 million mt) are 96% higher than biomass estimates by Canada's Department of Fisheries and Oceans in the 1980s, and three orders of magnitude greater than biomass estimates from 2002 (Rosenberg et al. 2005). The Scotian Shelf was heavily fished in the 1850s by well over a thousand fishing schooners, so the estimates of biomass are far from pristine, or pre-exploitation. Current fisheries management policies for the region use the 1980s biomass estimate as a target for stock rebuilding despite the fact that these values represent 4% of the biomass estimates from the 1850s,

which in itself is likely very distant from unfished biomass (Rosenberg et al. 2005). Using these historical logs provides a more accurate assessment of cod stocks in the past and offers a more realistic benchmark for future management.

### *Case 2: Hawaiian Archipelago*

Ancient Hawai'i had a long history of sustainable subsistence fishing (Kirch 1982, Kittinger et al. 2011), followed by a shift to commercial fishery after Western contact (after AD 1778; Schug 2001). By 1900, commercial fisheries had become a dominant feature in local island economies, with fish markets established on each of the main Hawaiian Islands (McClenahan and Kittinger 2012).

As a result of their importance to the Hawaiian economy, major quantitative surveys of the commercial fisheries were conducted in 1900 and 1903 by the U.S. Fish Commission (Cobb 1902, 1905a, 1905b), followed by data collection by the Territory of Hawai'i in the 1920s and 1930s, with continuous data collection by the territory, and then by the state of Hawaii, since 1948. Records of landings by trip began in 1966, enabling calculations of CPUE starting at this point.

Commercial landings for a number of important species have shown dramatic declines since the early 1900s, with most recent landings at <1% of maximum landings (Figure 5.1). The character of Hawaii's commercial fisheries has changed dramatically over the past 100 years, so trends in total landings may be driven by a number of factors other than fish abundance (e.g., transition to a tourism-based economy, other economic opportunities). Therefore, it is illustrative to look at trends in catch rates, which are more closely correlated with stock abundance than are total landings. Improvements in boats, engines, and fishing technology over time have resulted in large increases in fishing-gear efficiency. We used a conservative estimate of a 2% increase in fishing efficiency per year and examined  $CPUE_{now}/CPUE_{max}$  to determine current catch rates compared with maximum catch rates. Because data by trip began in 1966, decades after the onset of commercial fishing, our ratio is a conservative estimate, given that  $CPUE_{max}$  does not likely represent an unfished, or even lightly fished, condition. For nearly all species and all gear types examined, current CPUE values are generally well below 1% of historical highs (Table 5.1). A few exceptions include the handline fishery for soldierfishes (Holocentridae), where CPUE is currently 15% of  $CPUE_{max}$ ; and the lay gill net fishery for rudderfishes (Kyphosidae), with current CPUE at 9.5% of  $CPUE_{max}$ . These ratios are still extremely low compared to historical values and suggest substantial declines in stock size.

### *Case 3: Florida and Cuba Grouper Fisheries*

The histories of the fisheries of Cuba and Florida are intertwined and date back to pre-Columbian times. Prior to 1955, insular fisheries on the Cuban shelf consisted mostly of small-scale artisanal fisheries with catch never exceeding 10,000 t annually (Claro et al. 2001). By the mid-1980s, however, increases in fishing effort and efficiency resulted in landings of 79,000 t, but signs of overfishing were evident for a number of valuable target

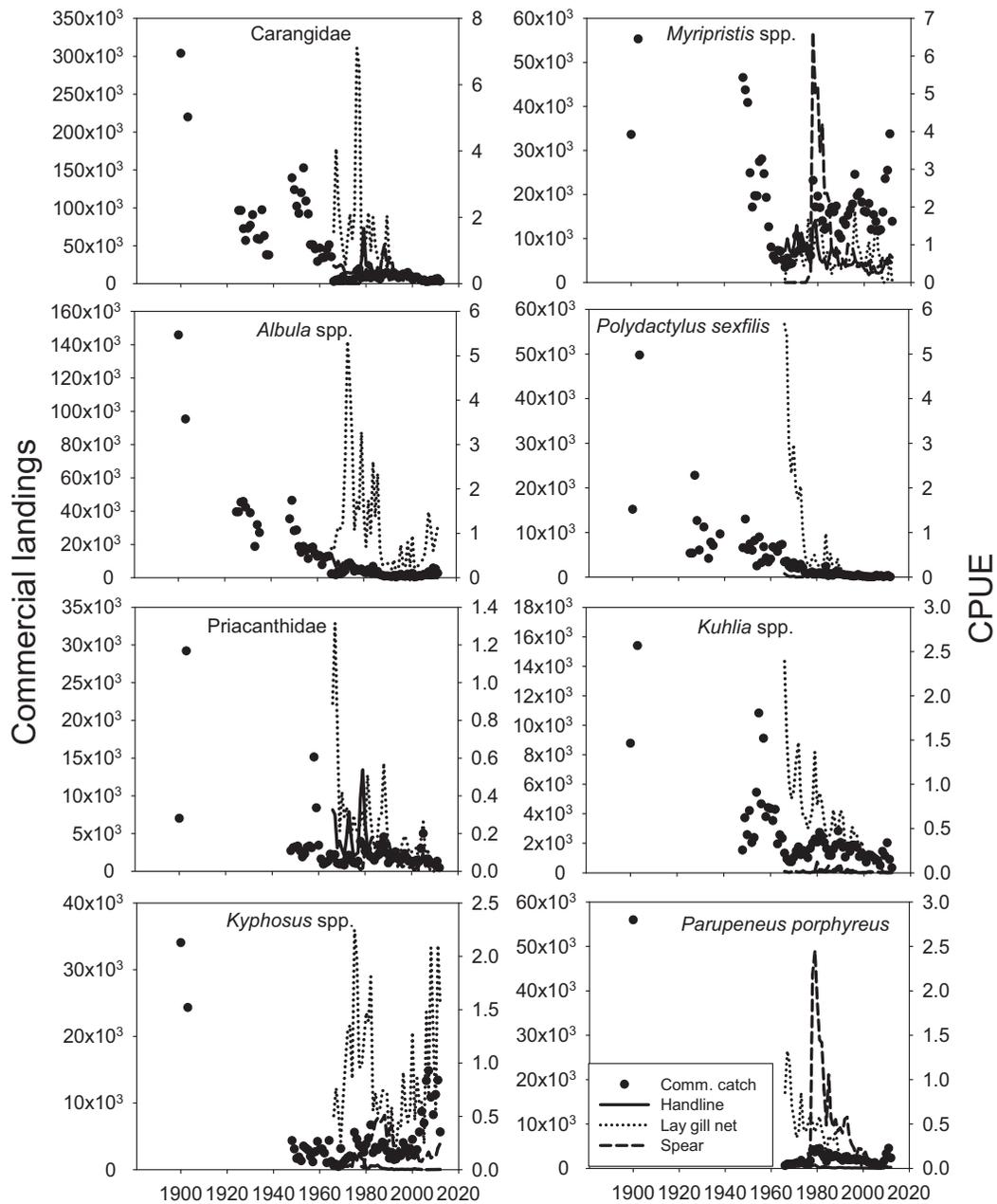


FIGURE 5.1 Total landings (kg) and catch per unit effort (CPUE; kg trip<sup>-1</sup>) by gear type from Hawaii commercial fisheries catch records. Major quantitative surveys of the commercial fisheries were conducted in 1900 and 1903 by the U.S. Fish Commission (Cobb 1902, 1905a, 1905b), followed by data collection by the Territory of Hawai'i in the 1920s and 1930s, with continuous data collection by the territory, and then the state of Hawaii, since 1948. Records of landings by trip began in 1966 (Hawaii Division of Aquatic Resources data), thus enabling calculations of CPUE starting at this point.

TABLE 5.1 Hawaii Commercial Fish Landing and Catch Rates

CPUE = catch per unit effort (kg trip<sup>-1</sup>)

Taxa	Statistics	Handline CPUE	Lay Gill Net CPUE	Spear CPUE	Commercial Catch (kg)
Jacks (Carangidae)	Maximum	1.7	7.1	1.2	303,848
	Current % of maximum	<1.0	<1.0	<1.0	<1.0
Bonefishes ( <i>Albula</i> spp.)	Maximum	0.50	5.33	0.04	145,886
	Current % of maximum	<1.0	2.1	1.5	<1.0
Soldierfishes (Holocentridae)	Maximum	1.65	1.89	6.64	55,316
	Current % of maximum	15.0	1.6	1.9	6.3
Threadfin ( <i>Polydactylus sexfilis</i> )	Maximum	0.08	5.67	0.02	49,768
	Current % of maximum	<1.0	<1.0	2.5	<1.0
Whitesaddle goatfish ( <i>Parupeneus porphyreus</i> )	Maximum	0.11	1.32	2.45	43,820
	Current % of maximum	<1.0	<1.0	1.6	1.4
Rudderfishes (Kyphosidae)	Maximum	0.15	2.25	0.51	34,047
	Current % of maximum	<1.0	9.5	3.3	1.3
Bigeyes (Priacanthidae)	Maximum	0.54	1.32	0.18	29,211
	Current % of maximum	3.5	<1.0	3.6	1.6
Flagtail ( <i>Kuhlia</i> spp.)	Maximum	0.04	2.39	0.12	15,403
	Current % of maximum	<1.0	<1.0	1.6	2.2

Notes: Landings data begin in 1900. Catch rates begin in 1966, when data began being reported by individual trip.

species despite the introduction of significant management measures (Claro et al. 2001, 2009). The government-owned and -managed fishing industry in Cuba provides a uniquely detailed multidecadal database of fisheries landings that is a valuable tool for assessing trends in a wide variety of species and gear types (Claro et al. 2009).

Following European contact, commercial fisheries developed in the Florida Keys because of their proximity to the mainland North American settlements and northern Caribbean islands. Recreational fisheries commenced in the mid-1800s, and over the past 4 decades there has been a dramatic increase in the amount of recreational fishing pressure, with the number of registered vessels quadrupling during this period (Ault et al. 1998).

Nassau grouper (*Epinephelus striatus*) was once an important component of Cuba's near-shore fisheries, reaching 1,728 t in 1963, but has declined to <2% of that peak in recent years (Claro et al. 2009; Figure 5.2). Examination of the headboat and commercial catch of Nassau grouper in south Florida since the 1980s shows a similar proportional decline to that of the Cuba data, irrespective of the two-orders-of-magnitude overall difference in catch (Bohnsack 2003, NOAA unpublished data). These similarities in more recent trends suggest that Nassau grouper populations may have been much larger in south Florida in the past. This

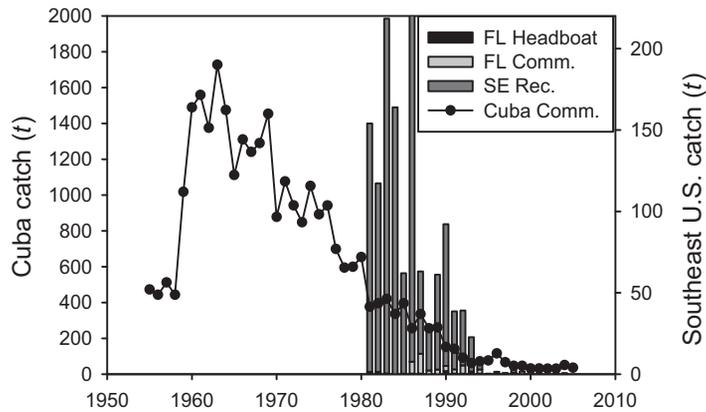


FIGURE 5.2 Historical Nassau grouper (*Epinephelus striatus*) landings for Cuba and south Florida. Commercial landings from Cuba date back to 1958 (Claro et al. 2001, 2009). Data from Florida come from commercial landing data (Fish and Wildlife Research Institute; <http://myfwc.com/research/saltwater/fishstats/commercial-fisheries/landings-in-florida/>) and recreational headboats (Bohnsack 2003, National Marine Fisheries Service 2012) starting in the mid-1980s.

case, as with the cod fisheries in the northeast Atlantic and Hawaiian coral reef fisheries, highlights the “shifting baseline syndrome” (Pauly 1995), which results when there is inadequate long-term data prior to full exploitation, or overexploitation, of a fishery (Bohnsack 2003).

### Catch Reconstructions

Nontraditional sources (e.g., fishers’ interviews, maritime records, unpublished reports, etc.) can be used to reconstruct past landings in the absence of catch data (Pauly et al. 1998, Zeller et al. 2007; also see chapter 6, this volume). These diverse data sets can be used to estimate historical catch rates, which, combined with demographic information, can be used to estimate total catch (Zeller et al. 2006, 2008). For example, reconstruction of the fishery in American Samoa showed a nearly 80% decline in overall coral reef catches since the 1950s, revealing long-term overfishing of nearshore resources, which can go undocumented in the absence of formal fisheries catch data (Zeller et al. 2006). These reconstructions have been performed for numerous locations around the world (see chapter 6, this volume) and highlight the limitations of fisheries catch statistics and the need to take a long-term perspective when it comes to estimating potential stock productivity. However, catch reconstructions can also pose challenges for fisheries managers. As noted in Box 5.1, catch reconstructions can be less important to managers than more pressing challenges, such as protecting and restoring critical fisheries habitats.

Catch reconstructions can help establish reference points to estimate the full productivity of unfished marine ecosystems. For example, a comparison of reconstructed yields from Hawai’i and Florida provides insight into precontact productivity for these two regions.

### BOX 5.1 Viewpoint from a Practitioner: Real-world Constraints to Applying Historical Baselines in U.S. Fisheries

John Henderschedt

The “shifting baseline syndrome” may influence the frame of reference in rebuilding targets for depleted fisheries, but how is this significant? While historical evidence of an abundant “unfished” biomass of a presently smaller stock may speak volumes about failures of management and long-term changes in fishery ecosystems, it is not necessarily an appropriate reference point for present-day managers working to balance stock rebuilding with fisheries yield. Fishing mortality, loss or damage to habitat, shifts in trophic dynamics, and impacts of a changing climate might individually or collectively preclude rebuilding a stock to its unfished level. Even when there are adequate data to support a historical reconstruction of stock size, the indelible impacts of human activity and natural shifts within the ecosystem over decades, and in some cases centuries, often place the past and present into differing contexts.

In recommending harvest strategies that are compliant with the 10 national standards of U.S. federal fisheries law (the Magnuson-Stevens Fishery Conservation and Management Act) and the implementation guidelines for those standards, I am much more interested in current estimates of maximum sustainable yield, and in the trajectory of the stock in relation to that metric. That said, there are many data-poor scenarios in which traditional and local ecological knowledge (TEK and LEK) may prove to be

valuable management inputs. This is particularly the case when such data can inform the identification of sustainable fishery mortality rates, which may, in many cases, prove more effective than biomass reference points. Understanding the role and vulnerability of the species within a dynamic ecosystem enables managers to be responsive and proactive within current natural states, and TEK and LEK may provide valuable insight into these dynamics.

TEK and LEK can, and should, play a significant role in achieving these goals. Where conventional assessment data are lacking, TEK and LEK can be invaluable tools for estimating present abundance of stocks as well as current trends within the ecosystem. Even in data-rich scenarios, TEK and LEK may assist scientists and managers in validating conclusions and in responding to observed changes within the ecosystem. Generally speaking, I do not consider the best use of TEK and LEK to be the reconstruction of historical baselines, but rather to assist in understanding the current ecosystem and stock dynamics to support the achievement of sustainable yield going forward. Looking beyond the potential value of TEK and LEK in establishing biological reference points, these tools can also inform managers about sustainable fishing practices and allocation methods in contemporary small-scale fisheries.

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Despite markedly different exploitation histories, the modern levels of extraction on Florida (12–13 t km<sup>-2</sup>) and Hawaiian (10–12 t km<sup>-2</sup>) coral reefs are similar (McClenachan and Kittinger 2012). Based on population data and potential per capita catch rates, the total reconstructed yield for wild-caught coral reef fisheries in Hawai'i achieved a maximum in the mid-1400s (>17 mt km<sup>-2</sup>; McClenachan and Kittinger 2012). Precontact fisheries' reconstructed yields ranged from 12 to 17 mt km<sup>-2</sup> for nearly 400 years, suggesting

sustainable rates of fishing mortality over that period. In Florida, by contrast, total catch remained  $<5 \text{ t km}^{-2}$  until 1930 (McClenachan and Kittinger 2012). Before European settlement, catch was well below  $1 \text{ t km}^{-2}$  and slowly increased until the second half of the 20th century, when rates increased rapidly, exceeding  $20 \text{ t km}^{-2}$  for much of the 1980s and 1990s. Reductions in overall landings occurred since the mid-1990s, with a decline of 50% between 1996 and 2008. Historical reconstructions help establish more accurate reference points and need to be considered when developing contemporary fisheries policies.

### Traditional and Local Ecological Knowledge

Traditional ecological knowledge (TEK) is defined as “a cumulative body of knowledge, practice and belief, evolving by adaptive processes and handed down through generations by cultural transmission, about the relationships of living beings (including humans) with one another and with their environments” (Berkes 2008). In the Pacific, island cultures have depended on the sea as their primary source of food for millennia, and along the way they invented nearly all of the basic fisheries conservation measures that we have in place today (e.g., closed areas, closed seasons, size limits, and restricted entry; Johannes 1998). One key management measure was the recognition of property rights by local communities, with local chiefs able to enact fishery management policies knowing that the future benefits of present sacrifices would benefit their own community. Area and seasonal closures were common, particularly when the chief felt that a stock was overfished (Johannes 1978, 1981). Areas were also left in reserve in anticipation of future needs, such as closing a particular area to accumulate fish to be caught for upcoming ritual feasts, as well as unanticipated ones, such as closing calm inshore areas to be used only during extended periods of rough weather (Titcomb 1972). Additionally, some fish were purposely allowed to escape, with the intention that they would serve to repopulate the stock.

In Hawai'i, Polynesian cultures developed a lunar calendar, which encompassed a detailed understanding of the marine environment and was used to help regulate fishing effort and timing (Poepoe et al. 2007). The moon calendar emphasized certain repetitive biological and ecological processes (e.g., fish spawning, aggregation, and feeding habits), which function at different time scales (e.g., seasonal, monthly, and daily; Friedlander et al. 2013). These practices were based on detailed TEK of these marine systems and allowed these cultures to maintain resilient and responsive management practices that reduced pressures at signs of overfishing and allowed heavier fishing for healthy stocks.

Similarly, the native inhabitants of the Lower Klamath River basin, in coastal northern California, relied on salmon for the bulk of their dietary protein and developed a complex system of legal rights and religious observations to maintain these stocks (McEvoy 1986). Fish were primarily caught in large communal weirs (funneling fish traps) that had strict requirements for construction and dismantling based on ritual and religious beliefs (Swezey and Heizer 1977). Construction and blessing of the weir took 10 days,

during which time salmon escaped upstream. The weir was dismantled 10 days later, allowing additional escapement. In this way, escaped salmon could swim to upstream spawning grounds before and after the use of the weir. In good years, catches would be moderated by escapements, which indirectly limited human population growth. In bad years, catches would be low but fish would still be allowed to escape and maintain the viability of the fishery.

In addition to TEK, contemporary local ecological knowledge (LEK) can be used to help inform the status of fish stocks, particularly when other forms of data are limited or absent (Neis et al. 1999). For example, Australia's southeast region is one of the most important fishing areas in the country (Bax and Williams 2001). With fishers' input, scientists were able to identify fisheries-independent survey sites that are important for these fisheries without the intense effort usually associated with mapping projects (Williams and Bax 2007). Another example where critical habitat was identified through fishers' knowledge was the cod spawning grounds in New England. Prior to a fisher-based spawning-ground study, very few spawning locations were known and researchers had difficulty determining the basic life history of these local stocks without the knowledge of fishers (E. P. Ames 2004, T. Ames 2007).

One recently developed approach is Participatory Fisheries Stock Assessment (ParFish), which aims to obtain information on stock condition in situations where data are inadequate for a conventional assessment (Walmsley et al. 2005). This multicriterion decision-making methodology uses interviews of fishers to identify stakeholder preference among various management outcomes and to create a preliminary estimate of stock status. It utilizes Bayesian decision analysis, with uncertainty in the results shown as probability density functions that can be broken down into simpler components, thus making multispecies assessments more viable. ParFish methodology has been developed and tested through a number of pilot studies conducted on various fisheries throughout the world, including in the Caribbean, East Africa, and India.

Research on TEK and LEK systems has increased dramatically over the past several decades. Customary fishery management practices based on TEK and LEK are being implemented in policy in many places worldwide and are increasingly integrated with conventional management approaches (see chapter 7, this volume). Managers are also increasingly engaging with TEK and LEK systems, particularly when conventional fishery data are lacking (Box 5.1). Such approaches allow researchers and practitioners to engage productively with fishers, who interact with the resources on a daily basis and are intimate with the status of many fish stocks. Inclusion of TEK and LEK provides further insight into effective management systems, which in many cases were successful at maintaining sustainable fisheries. Additionally, it is important to match the scales of management to those of the community that engages with fishery resources. (See Box 5.2, which discusses the importance of matching scales and how incorporation of LEK and TEK can help aid decision-making processes.)

## BOX 5.2 Viewpoint from a Practitioner: The Importance of Matching Scales in Fisheries Comanagement

Dean Wendt

It is my opinion that any sustainable management system has to be at a scale that matches people's inherent connection with the ecosystem and the services it provides them. People will only conserve and protect that with which they identify. Fisheries management in the United States occurs on a geographic scale that is so large that it misses the inherent social and ecological heterogeneity of fishing communities and fish stocks, thereby weakening the critical connection between people and the ecological resources on which they rely. Recognition of the mismatch in social and ecological scales and our fisheries management system is creating momentum in the United States for a more bottom-up, comanagement approach whereby communities engage with authority in the management of their local resource.

In addition, the policy change in 1998 of the Magnuson-Stevens Fishery Conservation and Management Act states that federal fisheries management councils should adopt a precautionary approach to specifying optimum yield of a stock. The federal government has essentially reversed the burden of proof, and we now

implement significant conservation measures even in the absence of scientific evidence that a stock is being overexploited. Put another way, without good data on the status of a fish stock, the "Restrepo rule" dictates that historical catches be reduced significantly (Restrepo et al. 1998).

As Friedlander and colleagues show in this chapter, such shifts in management and policy necessarily require the development and use of new tools and methods in the assessment, allocation, and utilization of a broad diversity of data sources. They also incentivize fishermen and fishing communities to engage in formal data collection for management and to contribute their knowledge to the management process. Management at smaller, more appropriate social and ecological scales using a precautionary approach will require incorporation of local and traditional ecological knowledge into decision-making processes. If communities are given a viable way to incorporate their knowledge and collaborate with government, I'm confident that decisions will be made that yield the best solution for sustained social and ecological benefits.

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## Paleoecological and Archaeological Evidence

Paleoecological evidence has also been used to examine historical changes in fishery species. Since most fishery species have some form of calcified body parts, changes in catch can be inferred through shifts in abundance and sizes in archaeological deposits and sediment samples, which may date back thousands of years. This paleoecological evidence offers important insight into both human harvesting behaviors and natural cycles of abundance and can sometimes be used to elucidate whether recent changes in resource abundance might be natural, rather than human-induced, cycles. For example, a fish population may show random variation among years but may demonstrate large oscillations that occur at scales of decades to centuries (Finney et al. 2010, Valdés et al. 2008). If we step back even

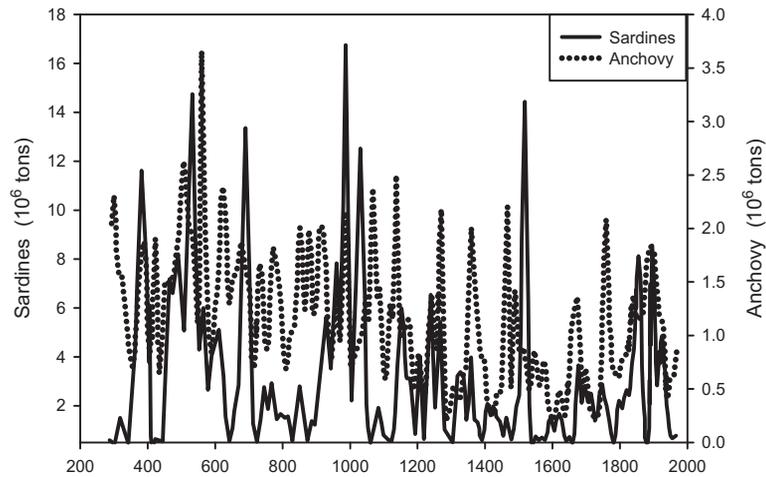


FIGURE 5.3 Two-thousand-year reconstruction of sardine and anchovy populations based on scale deposits in the Santa Barbara Channel, California (modified from Baumgartner et al. 1992).

farther and look at evolutionary time scales of millions of years, we may see population changes due to shifts in local, regional, and even global climatic conditions (Jackson 2010).

Furthermore, by unearthing paleoecological evidence, we can place recent population changes into a longer-term historical context. For example, Baumgartner et al. (1992) counted the sardine- and anchovy-scale deposits in sediment from the fishing ground off the coast of California and calculated the biomass of these two fished species for the past 2,000 years (Figure 5.3). The reconstructed sardine and anchovy stocks showed repeated fluctuation over the interval of several decades or longer, associated with cycles in oceanic conditions (Jacobson and MacCall 1995). Sardines were the subject of intense fishing pressure throughout the early 1900s, but the fishery collapsed in the 1950s. The collapse led to a moratorium on sardine fishing in 1967, by which time stocks along the west coast of the United States had already collapsed (Radovich 1982). Both the intensive pressure and the moratorium were policies that did not reflect the cyclical nature of this fishery. Starting in 1980, sardine populations started to show recovery, and the fishery is currently under much stricter regulations, which limit catches to 25% of estimated maximum sustainable yields. This policy was crafted with a desire to moderate annual fluctuations in allowable catch levels while considering production cycles and the potential for a future stock collapse. However, these estimates continue to generate controversy (e.g., Sugihara et al. 2012).

Zooarchaeological remains can also tell us a great deal about past fish stocks over extremely long time series, although the temporal resolution is rather coarse, usually on the order of decades to centuries (Erlandson and Rick 2010). Using species remains from archaeological deposits, a number of researchers have demonstrated reductions in the size of fish caught through time (Amorosi et al. 1994, Jackson et al. 2001), changes in fish growth rates (Van Neer et al. 2002), changes in the genetic diversity of populations (Larson

et al. 2002), and changes in species composition (Butler 2001), all of which can yield information about fishing intensity. Prior to European contact, faunal assemblages recovered from midden deposits in a number of Pacific Islands show overall declines in both catch and effort through time, suggesting overfishing, changes in agriculture or other subsistence practices, or a combination of the two (Erlandson and Rick 2010, Nagaoka 2001).

Wing and Wing (2001) studied the faunal remains from archaeological sites on five Caribbean islands, each with an early (1,850–1,280 yr BP) and late (1,415–560 yr BP) occupation. On each island, the mean size of reef-obligate species (e.g., parrotfishes, surgeonfishes, snappers, and groupers) showed large declines while facultative species such as jacks and herring showed little change in size. The authors also found a sharp decline in total reef fish biomass and mean trophic level from early to late occupation, suggesting heavy exploitation even in prehistoric times. These results are consistent with modern patterns of overexploitation and suggest that growth overfishing and fishing down the food web occurred long before European contact.

In several well-studied midden sites in Hawai'i, fish remains shift over time from a predominance of carnivorous to herbivorous reef fishes (Kittinger et al. 2011), suggesting early examples of fishing down the food web (Pauly et al. 1998). However, a decreased reliance on marine protein as a result of increased animal husbandry and sophisticated resource-management systems resulted in several hundred years of stable harvest levels. For example, modest increases in the size of parrotfish bones and limpet shells suggest release of these populations from exploitation pressure during the development of an agrarian society (about AD 1400–1778+; Kittinger et al. 2011). Zooarchaeological remains can therefore tell us a great deal about long-term historical changes in catch composition and mortality rates and can serve as proxies for estimates of abundance.

Another way to study archaeological materials is through ancient DNA. Recent improvements in molecular techniques to recover genetic material have allowed scientists to compare genetic diversity and population structures of current fish stocks to their historical state. Studies of herring in British Columbia (Speller et al. 2012), North Sea cod (Hutchinson et al. 2003), and snappers in New Zealand (Hauser et al. 2002) have shown large reductions in genetic diversity since the onset of each of these fisheries, while examination of Plaice DNA from the 1920s for the North Sea and Iceland show an effective population size five orders of magnitude smaller than the estimated population size today, with significant heterozygote deficiencies that coincide with increased fishing mortality after World War II (Hoarau et al. 2005).

#### Using Unfished Reference Areas

“Space-for-time” substitution has been used in many instances as an alternative to long-term studies to assess the impact of human-induced changes where pre-impact records are sparse or nonexistent (Pickett 1989). Surveys of remote coral reefs in the Pacific (Friedlander and DeMartini 2002, Sandin et al. 2008, Williams et al. 2008) support historical reports of high fish abundance and predator domination that characterized coral reefs before extensive fishing occurred. These areas therefore give us a window into the past as to what reefs looked

like prior to human extraction and provide baselines for comparisons with more exploited locations (Knowlton and Jackson 2008).

Biomass estimates from unfished areas have recently been used as a substitute for pristine, unfished estimates in fishery stock-assessment approaches (Babcock and MacCall 2011, McClanahan et al. 2011). The ratio of fish density outside versus inside unfished areas can be used as a proxy for biomass depletion ( $B/B_{\text{target}}$ ) in fisheries, thus eliminating the need for a stock assessment to estimate depletion. This approach is also advantageous in that it requires no historical data, and because in some places older, large marine reserves exist, which can be used as rough approximations of unfished biomass (Babcock and MacCall 2011). For example, in the western Indian Ocean, McClanahan et al. (2011) used unfished reference areas and the oldest no-take marine parks in the region, estimating the unfished reef fish biomass ( $B_0$ ) at  $\approx 1,200 \text{ kg ha}^{-1}$ .

There are two things to note when conducting or interpreting these types of analyses. The first is that no-fishing areas (e.g., marine reserves) used for analyses must be large when approximating unfished conditions. Most no-fishing reserves are small and likely inadequate, at least for mobile predators. The second point of caution is that a large unfished area may show strong biogeographic gradients among species, therefore obscuring potential responses due to fishing. Large unfished areas with the potential to serve as robust reference sites include the Chagos Marine Protected Area–Indian Ocean (640,000 km<sup>2</sup>), the Phoenix Islands Protected Area–Kiribati (408,250 km<sup>2</sup>), the Papahānaumokuākea Marine National Monument (362,073 km<sup>2</sup>), and no-take areas within the Great Barrier Reef Marine Park (113,982 km<sup>2</sup>).

A total of 57 fish species in Hawai'i were assessed by comparing biomass within the populated main Hawaiian Islands to the remote and virtually unfished northwestern Hawaiian Islands (Papahānaumokuākea Marine National Monument). Based on this assessment technique, one-quarter of the species examined in the main Hawaiian Islands were depleted below 10% of unfished abundance, while close to half were below 25% of unfished abundance. Large predators were especially affected, but many other target and nontarget species also appeared to be depleted. This study highlighted the value of large unfished areas as reference points for fisheries management and contrasted with previous works, which identified no-fishing areas as impediments to assessing stocks because their effects can complicate the interpretation of conventional fisheries data (Punt and Methot 2004, Field et al. 2006). However, this study also showed that small and sparse no-fishing areas in the MHI were inadequate to reestablish the full biological potential of many species and did not represent adequate reference areas.

A number of stock assessment parameters can be estimated with the help of unfished reference areas. For example, because the northwestern Hawaiian Island (NWHI) fish populations experience little or no fishing pressure, all mortality is considered to be natural mortality ( $M$ ), whereas the MHI populations experience both natural and fishing mortality ( $F$ ). Size frequency analysis of the blue trevally (*Caranx melampygus*), a highly prized recreational species, in the unfished NWHI produced an estimate of  $M = 0.27$  and an estimate of total mortality ( $Z = M + F$ ) in the MHI of 0.69, which can be used to calculate  $F = 0.42$  (Figure 5.4A, B; Fried-

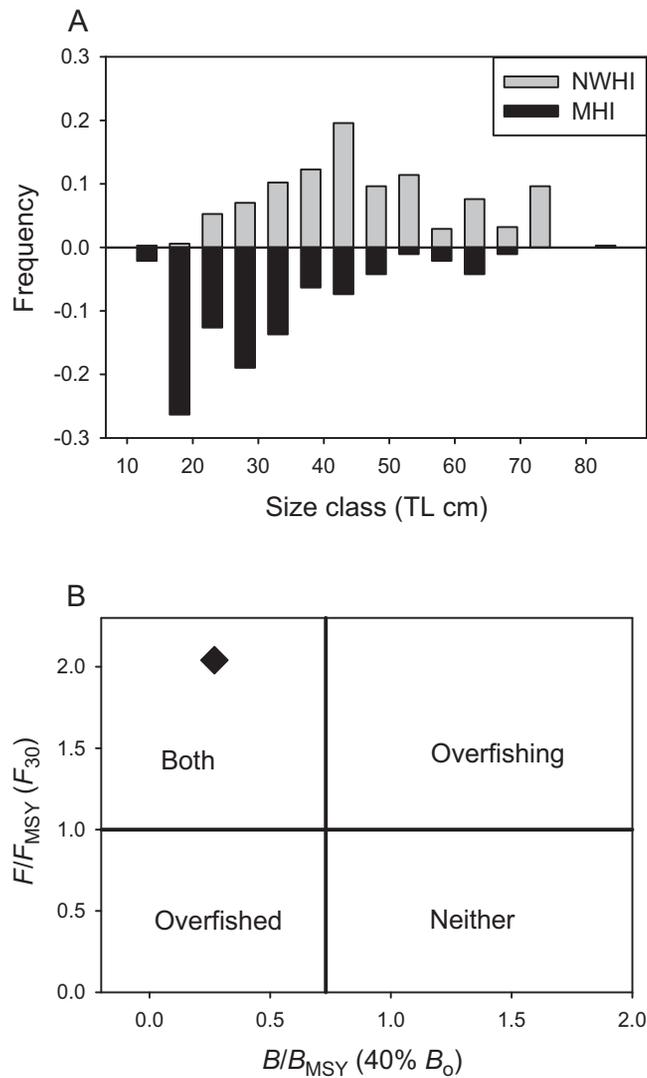


FIGURE 5.4 Length-based assessments of blue trevally (*Caranx melampygus*) in the main Hawaiian Islands (MHI) using the northwestern Hawaiian Islands (NWHI) as an unfished reference area. (A) Comparison of length frequencies of blue trevally in the MHI and NWHI (TL = total length). (B) Estimates of actual biomass ( $B$ ),  $B_{MSY} = 40\% B_o$ , fishing mortality rate ( $F$ ), and  $F_{MSY} = F_{30}$  in the MHI. See text for details.

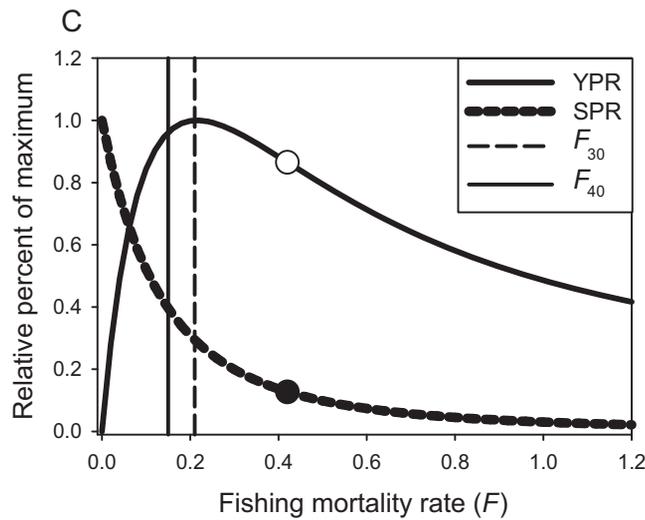


FIGURE 5.4 (continued) (C) Spawning potential ratio (SPR) and yield per recruit (YPR) models for *C. melampygu* in the MHI. See text for details.

lander et al. 2008). It is common to set fishing mortality-rate limits that allow individuals within the population to produce 30–40% of its reproductive potential in the absence of fishing ( $F_{30}$  to  $F_{40}$ ; National Marine Fisheries Service 2012). For this species,  $F_{30}$  was estimated at 0.21 and  $F_{40}$  at 0.15, which suggests that recent fishing rates were 2 to 3 times higher than a reasonable fishing limit. The spawning-potential ratio calculation in the study indicated that blue trevally in MHI are currently producing only 13% of their reproductive potential (Figure 5.4C). These results are consistent with analyses of the relative biomass densities of this species in the MHI and NWHI that indicated that the MHI population may have dropped to 2% of its unfished abundance (Friedlander and DeMartini 2002).

#### APPLICATION OF REFERENCE POINTS USING UNCONVENTIONAL MANAGEMENT TECHNIQUES

Effective fisheries management systems can be developed even if historical analyses provide only a few reference points (see Box 5.2). Fisheries management systems perform well, even with limited data, if they identify a target abundance level, allow reasonable levels of fishing when stocks are healthy, and reduce fishing effort decisively when stocks drop below target levels (e.g., Restrepo et al. 1998, Sladek Nowlis and Bollermann 2002, MacCall 2009). Simulations have demonstrated the potential of these techniques to yield fairly accurate catch limits (MacCall 2009), sustain high average catches (Sladek Nowlis and Bollermann 2002), and achieve a desired balance among several competing objectives (Sladek Nowlis 2004), even when the techniques rely solely on roughly estimated reference points.

To illustrate these capabilities, let's consider the dynamics of a fish population in terms of additions and removals as functions of population size. For additions, let's focus on natural productive capacity, which includes growth and reproduction as mediated by natural mortality. We expect this natural productive capacity to be zero when a population is extinct, for lack of seedstock; and when a population is highly abundant, for lack of resources (Figure 5.5, parabolic curves). For removals, we have fish catches, which are influenced by the fishery management system. A given system may result in a point where additions balance subtractions (Figure 5.5A), associated with a target catch and target abundance. These are the sorts of reference points that can be informed by historical studies but also need to reflect societal objectives. We can choose targets (Figure 5.5B), recognizing that higher catches lead to lower abundance. We can also choose how responsive our systems will be to changes in abundance both below (Figure 5.5C) and above target levels (Figure 5.5D). Responsiveness increases sustainability (Sladek Nowlis and Bollermann 2002), which may be especially important for data-poor fisheries, but decreases the predictability of catches and, consequently, economic returns (Sladek Nowlis 2004). Annual catch-limit rules are commonly used as part of such a system, but we can also use size limits and closed areas (Sladek Nowlis and Bollermann 2002). Ultimately, the best policy should reflect a balance among competing objectives based on societal needs (Sladek Nowlis 2004). Viewing fishery management systems in this manner highlights ways in which approaches can be developed for data-poor stocks, which are often overlooked. When only catch data exist, Restrepo et al. (1998) suggested estimating sustainable catch levels using average recent catches for fisheries with a suitably long history of catch records, as long as there is no evidence of stock decline (also see Box 5.2). MacCall (2009) recommended accounting for the stock depletion expected with the development of a new fishery with a fairly simple formula that relies on catch history, a proxy for  $B_{MSY}/B_o = 0.4$  and a proxy for  $F_{MSY} = cM$ , where  $c$  is a tuning parameter whose value would be chosen by expert judgment, often somewhere in the range of 0.6 to 0.8;  $M$  would also be estimated by expert judgment, aided by records of oldest observed fish where available. The final variable necessary for this technique is an estimate of the degree to which a stock has been depleted over the history of the fishery. Absent a fishery-independent or CPUE index of abundance, such information might be collected anecdotally from experienced fishermen.

Interestingly, the principles behind these data-limited management systems are consistent with evidence we have from TEK. Not all indigenous groups managed their fishery resources sustainably (e.g., Diamond 2005). Examples where they did (e.g., the previously discussed Hawaiian Island and Lower Klamath River societies), though, show strong evidence for adaptive management approaches where adjustments were made to catches at early signs of depletion. These systems are being described with increasing intensity by researchers (see chapter 7, this volume). Some related concepts have recently been proposed or even implemented, partly in response to tightening standards that address the need for annual catch limits in federally managed U.S. fisheries.

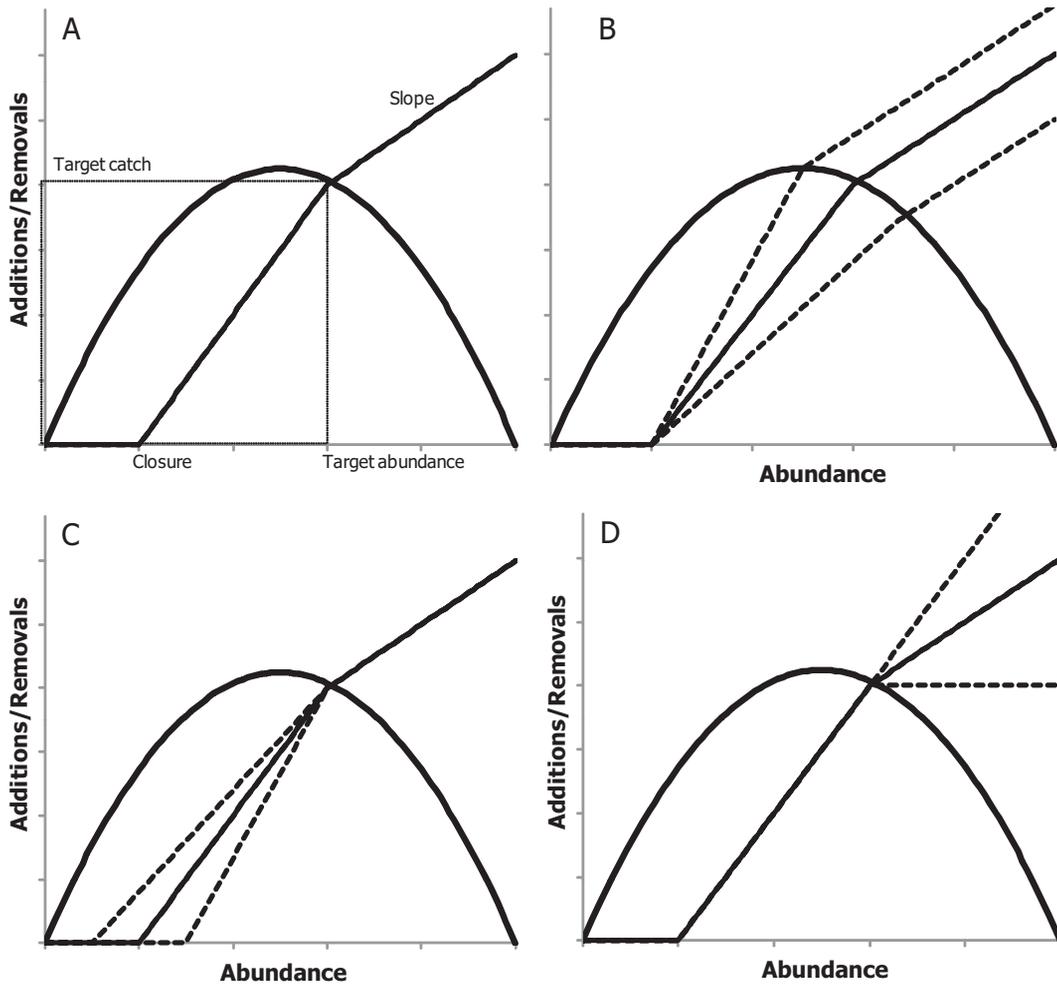


FIGURE 5.5 Data-poor management in theory. Fisheries can be characterized by abundance-based additions (in the form of natural productive capacity; the parabolas) and removals (in the form of catch limits; kinked lines, both dotted and solid), the intersection of which represents target catch and abundance (A). Fisheries productivity and catch limits can be paired to achieve desired results, subject to fundamental trade-offs. We can vary the targets (B) and the responsiveness of the management system to abundance changes below (C) or above (D) target levels. (Details adapted from Sladek Nowlis and Bollermann 2002, Sladek Nowlis 2004.)

### CONCLUSIONS

The global fisheries crisis, combined with ever increasing exploitation pressures on these resources, requires us to dramatically change the way we interact with and manage our marine ecosystems. This includes taking into account the important ecological roles of fishery species in maintaining ecosystem resilience (for an example from the Caribbean, see

### BOX 5.3 Viewpoint from Practitioners: Historical Baselines and Fishery Management—Parrotfish and Caribbean Corals

Steve Roady and Andrea Treece

Elkhorn coral (*Acropora palmata*) and staghorn coral (*A. cervicornis*) were once the major reef builders in the U.S. Caribbean. Both have declined precipitously, with most populations losing 80–98% of their abundance as measured against a 1970s baseline. The U.S. Endangered Species Act protects these two coral species in recognition of their precarious condition.

Along with declines in such key species, many Caribbean reefs have also undergone a “phase shift” during the past three decades; as a result, macroalgae now dominate these systems. The regional shift from high coral cover to high macroalgal cover is attributed to a variety of factors, including the overfishing that has depleted herbivorous grazing species such as parrotfish (Scaridae), which control macroalgal growth.

Following the catastrophic near extinction of the sea urchin *Diadema antillarum* due to disease, parrotfish are now the only group of species remaining in the U.S. Caribbean that are capable of removing significant amounts of fleshy macroalgae from coral reefs. Parrotfish and other herbivorous fish help preserve coral reef habitat by grazing on algae that would otherwise crowd out these corals. Abundant and diverse herbivorous fish populations are necessary to remove sufficient algae to prevent algal overgrowth and create open space for new corals. Algae-dominated reefs provide far less productive habitat than coral-dominated ones, leading to a negative feedback loop in which fish populations decline as coral cover declines, and vice versa.

The struggle to protect Caribbean parrotfish and to foster their vital role in preserving critically depleted corals graphically demonstrates the importance of including historical baselines in fisheries management. A failure by the U.S. government to consider such baselines in Puerto Rico and the U.S. Virgin Islands threatens to degrade these corals further by allowing continued fishing of parrotfish. Despite the acknowledged importance of parrotfish in protecting the critically depleted elkhorn and staghorn corals (NOAA Biological Opinion 2011:116, Mumby et al. 2007), the U.S. government failed to consider history when establishing catch limits for those fish in 2011. The government neither endeavored to reconstruct historical baselines of parrotfish populations nor analyzed the numbers necessary to restore their previous role in reef ecosystems. Indeed, rather than relying on historical data or generating estimates of the parrotfish abundance required to restore healthy elkhorn and staghorn populations, the government downplayed the relative importance of grazing by the current (depleted) parrotfish assemblage (NOAA Biological Opinion 2011:50).

This failure to establish a historical baseline incorporating a robust and functional Caribbean reef ecosystem deprives managers of the context required to consider the true effects of fishing. Resulting management decisions allow continued fishing pressure on parrotfish—vital herbivores essential to coral protection—and risk irreversible damage to the coral reef ecosystem that sustains the fishery and, thus, to the fishery itself.

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Box 5.3). Fish stocks that have been exploited close to their maximum capacity have lower resilience to stochastic events due to truncated size structure and decreased recruitment potential (Berkeley et al. 2004, Musick 1999). As our climate changes, the number of these extreme climatic events is expected to increase. Historical fisheries data allow us to examine past variability, and it is critical to have this historical knowledge if we are to efficiently manage fisheries stocks into the future with a rapidly changing climate.

A long-term perspective is critical to avoid the pitfalls of the shifting baseline and to help us better understand the past productivity of many marine ecosystems around the world. Assessing the status of most stocks has been hindered by the lack of adequate information. In order to assess a greater number of stocks, we must utilize a wider range of data sets that are often overlooked by conventional stock assessment. Historical data are typically underutilized in fisheries stock assessments because of a lack of standardization and difficulties in incorporating these data into standard stock assessment models. A better understanding of past ecosystem productivity is critical if we are to effectively manage these ecosystems into the future. Unfished biomass is almost never known, and this nearly universal data gap causes problems, ones that historical information offers promise for addressing. By incorporating historical data, TEK, and LEK into population assessment models and management practices, we bring insight into the yield of these ecosystems in the past and provide guidance for future management actions.

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