Robbing Peter to pay Paul: replacing unintended cross-taxa conflicts with intentional tradeoffs by moving from piecemeal to integrated fisheries bycatch management

Eric Gilman, Milani Chaloupka, Laurent Dagorn, Martin Hall, Alistair Hobday, Michael Musyl, Tony Pitcher, François Poisson, Victor Restrepo, Petri Suuronen

Rev Fish Biol Fisheries (2019) 29:93–123
https://doi.org/10.1007/s11160-019-09547-1

REVIEWS

Robbing Peter to pay Paul: replacing unintended cross-taxa conflicts with intentional tradeoffs by moving from piecemeal to integrated fisheries bycatch management

Eric Gilman · Milani Chaloupka · Laurent Dagorn · Martin Hall · Alistair Hobday · Michael Musyl · Tony Pitcher · Francois Poisson · Victor Restrepo · Petri Suuronen

Received: 1 October 2018 / Accepted: 3 January 2019 / Published online: 8 January 2019

Abstract Bycatch in fisheries can have profound effects on the abundance of species with relatively low resilience to increased mortality, can alter the evolutionary characteristics and concomitant fitness of affected populations through heritable trait-based selective removals, and can alter ecosystem functions, structure and services through food web trophic links. We challenge current piecemeal bycatch management paradigms, which reduce the mortality of one taxon of conservation concern at the unintended expense of others. Bycatch mitigation measures may also reduce intraspecific genetic diversity. We drew examples of broadly prescribed 'best practice' methods to mitigate bycatch that result in unintended cross-taxa conflicts from pelagic longline, tuna purse seine, gillnet and trawl fisheries. We identified priority improvements in data quality and in understanding ecological effects of bycatch fishing mortality to support holistic ecological risk assessments of the effects of bycatch removals conducted through semi-quantitative and model-based

E. Gilman (✉)
Hawaii Pacific University, Honolulu, USA
e-mail: EricLGilman@gmail.com

E. Gilman
Indo-Pacific Tuna Program, The Nature Conservancy, San Francisco, USA

M. Chaloupka
Ecological Modeling Services, St. Lucia, Australia

M. Chaloupka
University of Queensland, St. Lucia, Australia

L. Dagorn · F. Poisson
MARBEC, Univ Montpellier, CNRS, Ifremer, IRD, Montpellier, France

M. Hall
Inter-American Tropical Tuna Commission, La Jolla, USA

A. Hobday
CSIRO Oceans and Atmosphere, Hobart, Australia

M. Musyl
Pelagic Research Group, Honolulu, USA

T. Pitcher
University of British Columbia, Vancouver, Canada

V. Restrepo
International Seafood Sustainability Foundation, McLean, USA

P. Suuronen
Natural Resources Institute Finland, Helsinki, Finland

approaches. A transition to integrated bycatch assessment and management that comprehensively consider biodiversity across its hierarchical manifestations is needed, where relative risks and conflicts from alternative bycatch management measures are evaluated and accounted for in fisheries decision-making processes. This would enable managers to select measures with intentional and acceptable tradeoffs to best meet objectives, when conflicts are unavoidable.

Keywords Bycatch · Conflicts · Decision support tool · Fisheries-induced evolution · Holistic management · Integrated management

Introduction

Fisheries that target relatively fecund species can have large impacts on incidentally caught species that, due to their lower reproduction rates and other 'slow' life history traits, are relatively vulnerable to increased mortality, including seabirds, sea turtles, sea snakes, marine mammals, elasmobranchs and some teleosts (Goni 1998; Hall et al. 2000; Branch et al. 2013; Gray and Kennelly 2018). Affected populations can decline over short temporal scales (decades and shorter) and are slow to recover once depleted (Musick 1999; Hall et al. 2000; Stevens et al. 2000; Kitchell et al. 2002; Dulvy et al. 2008). Intraspecific selective mortality can cause fisheries-induced evolution (FIE). This occurs when fisheries selectively remove individuals based on highly heritable traits that vary within affected populations (Heino et al. 2015; Tuck et al. 2015; Audzijonyte et al. 2016; Lennox et al. 2017; Hollins et al. 2018). Selective mortality on heritable traits reduces the range of phenotypes for these traits within the exploited populations. This, in turn, can reduce population fitness, resistance and resilience to natural pressures, and reduce the ability of populations to evolve in response to changes in environmental conditions (Saccheri et al. 1998; Westemeier et al. 1998; Reed and Frankham 2003; Jorgensen et al. 2007; Evans and Sheldon 2008).

Objectives of managing fisheries bycatch include avoiding loss of unique alleles and genetic diversity, preventing protracted or irreparable harm, or permanent loss, of affected populations per se, and sustaining a desired state of ecosystem balance that produces targeted services, including fishery yields (FAO 1995, 2003, 2011). These objectives are interlinked as fisheries bycatch removals and alterations to the evolutionary characteristics of affected populations can alter ecosystem functions and structure through food web trophic links (Pace et al. 1999; Stevens et al. 2000; Ward and Myers 2005; Ferretti et al. 2010). In some cases, use of a bycatch mitigation method that achieves large reductions in catch rates and at-vessel mortality of one species or group of conservation concern is at the expense of another (e.g., Beverly et al. 2009; Salerno et al. 2010; Hall 1998, 2015; Gilman et al. 2016b). When bycatch is managed in a piecemeal, species-by-species fashion, this can result in cross-taxa conflicts that are unplanned, with unintended consequences. Instead, holistically evaluating and identifying conflicts as well as mutual benefits amongst species groups from implementing bycatch management options supports carefully considered, intentional tradeoffs when cross-taxa conflicts cannot be avoided. The integrated bycatch management measures adopted by decision-makers would cause known and acceptable cross-taxa tradeoffs that best meet objectives.

Fisheries bycatch is currently largely managed in a piecemeal manner, both at broad regional levels and at the individual fishery level (Hall 1998, 2015; Gilman et al. 2014a, b, 2016b). Global guidance on bycatch management, including from the Food and Agriculture Organization of the United Nations, is single-taxa specific (seabirds, FAO 1999a; sharks, FAO 1999b; sea turtles, FAO 2010). Some 'best practices' prescribed to mitigate bycatch recommended by environmental non-governmental organizations (NGOs) and their seafood corporate partners (e.g., pelagic longline circle hooks to benefit sea turtles, Wegmans 2016) likewise are single-taxa specific. Regional fisheries management organizations employ taxa-specific bycatch conservation measures, e.g., a stand-alone measure to manage fisheries bycatch of sea turtles, a separate measure for seabirds, etc. (Gilman et al. 2014b; ISSF 2017a). Domestic fisheries management systems employ similar taxa-specific bycatch mitigation measures (e.g., standalone U.S. longline regulations on sea turtles, NMFS 2001, 2015; seabirds, NMFS 2005; odontocetes, NMFS 2012; sharks, NMFS 2014, 2016). These taxa-specific approaches to managing bycatch may not be integrated within an overarching holistic framework for managing bycatch.

As a result, because they do not account for conflicts between groups of conservation concern that result from their implementation, bycatch mitigation measures designed to address problems within one species group can exacerbate bycatch of other vulnerable taxa in some fisheries (Gilman et al. 2014a, b). Frameworks for bycatch risk assessment and management likewise largely do not account for effects at other levels of biodiversity, including effects of bycatch removals on genetic diversity within affected populations and broad changes in ecosystem functions and structure.

There are, however, numerous examples of bycatch mitigation methods that do not create conflicts and in some cases result in shared benefits to multiple bycatch groups. For example, non-entangling and biodegradable designs of fish aggregating devices (FADs) used in tuna purse seine fisheries reduce the entanglement of sharks, sea turtles and other organisms (ISSF 2015; Moreno et al. 2016). The backdown maneuver to release dolphins in tuna purse seine fisheries does not create conflicts (Hall 2015). Various seabird bycatch mitigation methods that prevent seabird access to pelagic longline baited hooks during setting, such as side-setting, bird-scaring *tori* lines, and underwater setting devices, are unlikely to cause cross-taxa conflicts (Gilman et al. 2005; Clarke et al. 2014). Various gillnet bycatch reduction approaches likely pose low probability of causing cross-taxa conflicts. This includes increasing net tension (stiffness) to reduce the likelihood of entangling sea turtles and other large organisms, such as by: reducing the net profile, increasing filament diameter, using stiffer weaves, using larger floats on the top rope and heavier weights on the bottom rope, and infusing certain compounds (Werner et al. 2006; Larsen et al. 2007; Thorpe and Frierson 2009). Eliminating or increasing the length of tiedowns used in anchored gillnet eliminates a bag of slack webbing that entangles organisms such as sea turtles (Price and Van Salisbury 2007; Gilman et al. 2010). Making portions of gillnets more visible by using certain net colors, using thicker twine diameter, attaching corks or other materials within the net, and illuminating nets with chemical or battery-operated lightsticks can reduce catch rates of sea turtles, seabirds and marine mammals (Hayase and Yatsu 1993; Melvin et al. 2001a; Barlow and Cameron 2003; Werner et al. 2006; Wang et al. 2010, 2013; Mangel et al. 2014). As another example, Turtle Excluder Devices, sorting and shepherding devices used in shrimp and prawn trawl fisheries, can exclude sea turtles as well as elasmobranchs and other large fishes, while largely retaining smaller organisms, such as targeted shrimp, that fit through the grid, without increasing catch of other taxa of conservation concern (Broadhurst 2000; FAO 2010; Willems et al. 2016). Spatial and temporal restrictions on fishing at predictable bycatch hotspots in some cases do not create cross-taxa conflicts (however, see the section "Temporal and spatial fishery closures"). This includes both marine protected areas (MPAs) that protect static (e.g., shallow submerged features such as seamounts) (Morato et al. 2008) and dynamic sites (gyres, fronts) (e.g., dynamic spatial management of southern bluefin tuna (*Thunnus maccoyii*) bycatch, Hobday et al. 2010; real-time fleet communication to enable vessels to avoid fishing grounds with high bycatch, Gilman et al. 2006b; Little et al. 2015).

Drawing examples from pelagic longline, tuna purse seine, gillnet and trawl fisheries, we demonstrate how bycatch mitigation methods that effectively mitigate catch and mortality of one species or group of conservation concern can exacerbate catch and mortality of another of potentially greater conservation concern. The bycatch mitigation methods in the examples use gear technology, which involves changes in fishing methods and gear, and temporal and spatial fishery closures (for information on the full range of bycatch mitigation approaches, see: Hall 1996; Hall et al. 2000; FAO 2011; Gilman 2011). We also present theoretical examples of how employment of bycatch mitigation methods could inadvertently exacerbate FIE, reducing genetic diversity within affected populations. We describe examples of qualitative to model-based, quantitative methods for integrated ecological risk assessments (ERAs) of the effects of fishing which could be used to holistically assess the relative risks from fishing across affected taxonomic groups and across manifestations of biodiversity, from genetic diversity to the state of ecosystem-level components. Priority improvements in data quality and in understanding ecological effects of bycatch fishing mortality to support holistic semi-quantitative and model-based ecological risk assessments of the effects of bycatch removals are identified. Findings from integrated risk assessments are needed to design bycatch management frameworks and define best-practice bycatch mitigation methods that account for cross-taxa conflicts and result in intentional and

acceptable tradeoffs. These examples highlight the need to move beyond piecemeal, species-by-species approaches to holistic bycatch assessment and management. This will enable the relative risks to affected components of biodiversity, including genetic diversity, impacts on populations and species, and broader ecosystem-level effects resulting from alternative fisheries bycatch mitigation methods to be identified and accounted for in fisheries decision-making processes.

Gear technology

Pelagic longline hook shape

Several environmental NGOs and retailers have adopted sustainable sourcing policies that call for the use of circle hooks instead of J-shaped hooks by pelagic longline fisheries as a solution to sea turtle bycatch (Monterey Bay Aquarium 2015; ALDI 2016; Wegmans 2016; WWF 2016). This taxa-specific recommendation could inadvertently exacerbate bycatch rates of other taxa of conservation concern. Four predominant hook shapes are used in global pelagic longline fisheries: circle, J, tuna and teracima (Fig. 1) (Beverly 2009; IATTC 2011). The point of J-hooks is positioned parallel to the hook shank. Tuna and teracima hooks have a slightly curved shank, and like J-hooks, the shank does not protect the point, and as a result, are categorized as 'J-shaped' hooks (see Fig. 2 for fishing hook anatomy) (Beverly and Chapman 2007; Serafy et al. 2009). Circular shaped 'circle' hooks have points that are turned perpendicularly back toward the shank. This makes the point less exposed relative to J-shaped hooks.

When ingested, J-shaped hooks tend to hook deeply, in the esophagus and gut. Circle hooks with

Fig. 1 Four hook designs commonly used in pelagic longline fisheries, from left, circle, J, tuna and teracima hooks. (Beverly 2009; IATTC 2011)

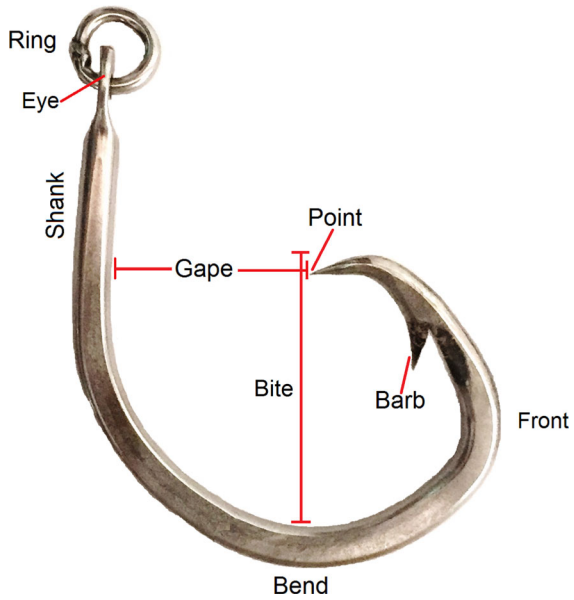

Fig. 2 Terminology for main elements of fishing hooks

little or no offset tend to catch in the corner of the mouth (Cooke and Suski 2004; Curran and Beverly 2012; Epperly et al. 2012; Clarke et al. 2014; Parga et al. 2015). Hooks can be 'straight', where the point is in the same plane as the shank, or 'offset', with the point bent away from the plane of the shank. There is likely a threshold offset angle above which a circle hook's gape would be sufficiently large to cause it to hook similarly to J-shaped hooks (Swimmer et al. 2010; Gilman and Hall 2015). Having a less exposed point, circle hooks also have a lower probability of foul-hooking (i.e., hooking externally in the body) than J-shaped hooks. Organisms that are foul-hooked or hooked in the mouth have higher at-vessel survival rates and possibly higher probability of pre-catch and post-release survival than those that are deeply hooked in the esophagus and gut (Borucinska et al. 2002; Chaloupka et al. 2004; Cooke and Suski 2004; Casale et al. 2008; Pacheco et al. 2011; Swimmer and Gilman 2012). Additionally, circle hooks' predominant hooking position in the mouth facilitates relatively easy access to remove all terminal tackle (hook, leader, weight, trailing branchline), potentially increasing post-release survival rates (Chaloupka et al. 2004; Cooke and Suski 2004; Godin et al. 2012; Serafy et al. 2012a; Swimmer and Gilman 2012; Parga et al. 2015).

Leatherback sea turtles (*Dermochelys coriacea*), the only extant sea turtle species to not have a hard

shell, are most frequently caught in pelagic longline gear by becoming foul-hooked and entangled. They have lower catch rates on circle than J-shaped hooks (Watson et al. 2005; Sales et al. 2010; Pacheco et al. 2011; Foster et al. 2012; Coelho et al. 2015; Santos et al. 2012, 2013; Gilman and Huang 2017). For hard-shelled and leatherback turtles that ingest a hook, circle hooks result in a lower proportion of turtles swallowing the hook deeply relative to J-shaped hooks (Andraka et al. 2013; Gilman and Huang 2017). Thus, relative to J-shaped hooks of the same minimum width, circle hooks benefit sea turtles (both hard-shelled species and leatherbacks) because a smaller proportion of those captured will be deeply hooked, and because they reduce leatherback sea turtle catch rates. However, use of circle hooks in place of similar sized J-shaped hooks will provide little benefit to hard-shelled turtles in deep-set longline fisheries where most turtles drown during the gear soak and are dead upon gear haulback (e.g., Gilman et al. 2006c; Clarke et al. 2014; Hall et al. 2017).

While hook shape has the same effect on shark anatomical hooking position and survival rates as for other taxonomic groups, some pelagic sharks have a higher probability of capture on circle versus J-shaped hooks (Gilman et al. 2016b; Reinhardt et al. 2017). Based on results of a meta-analysis from 30 studies, pooled shark species had a 1.2 times significantly higher risk of capture on circle hooks than on J and tuna hooks. Blue shark (*Prionace glauca*), the main shark species captured in global longline fisheries (Gilman 2011; Clarke et al. 2014; Oliver et al. 2015), was the predominant caught shark species in 17 of the 30 compiled studies in the meta-analysis (Gilman et al. 2016b). Consistent with the findings of Gilman et al. (2016b), a meta-analysis by Reinhardt et al. (2017) found significantly higher catch risk for six individual species of sharks, and no shark species had significantly lower relative risk of capture on circle hooks relative to J-shaped hooks. Due to the larger sample size plus the number of independent studies, correctly designed meta-analyses can provide estimates with increased precision and accuracy over estimates from single studies, with increased statistical power to detect a real effect (e.g. Borenstein et al. 2009; Musyl et al. 2011, 2015). Most of the studies compiled for the meta-analyses by Gilman et al. (2016b) and Reinhardt et al. (2017) employed designs with simultaneous variability in multiple potentially significant explanatory factors, including hook minimum width, such that the analyses were in effect measuring pooled relative catchability between wider circle and narrower J-shaped hooks. However, the small number of studies designed to enable an assessment of the single factor effect of hook minimum width have found higher probability of shark capture on circle hooks relative to J-shaped hooks of the same minimum width (Ward et al. 2009; Serafy et al. 2012b; Andraka et al. 2013; Caneco et al. 2014; Gilman et al. 2016b). While circle hooks increase shark catchability, they also reduce at-vessel mortality and the proportion of the catch that are deeply hooked relative to J-shaped hooks (Gilman et al. 2016b; Reinhardt et al. 2017).

Observations of lower shark probability of capture on J-shaped hooks relative to circle hooks may be partially due to a synergistic effect between hook shape and leader material, when monofilament leaders are used. When caught on circle hooks, sharks are most often hooked in the mouth and jaw, making it less likely that they can sever (bite through or abrade) a monofilament leader than when deeply hooked on a J-shaped hook (Afonso et al. 2012; Gilman et al. 2016b). Several studies have observed significantly lower shark catch rates on circle versus J-shaped hooks when wire leaders were used (Mejuto et al. 2008; Sales et al. 2010; Andraka et al. 2013), indicating that, when wire leaders are used, the single factor hook shape alone causes the observed difference in shark catch rates (Gilman et al. 2016b). The effect of hook shape on shark catch risk may be due in part to the way sharks consume their prey. Sharks chomp their prey (and baited hooks) repeatedly before swallowing, unlike teleosts who suck in and swallow their prey. While being munched by sharks, due to their shape, circle hooks roll and slide more so than J-shaped hooks, perhaps creating a higher probability that circle hooks will become oriented so that it lodges.

The relative magnitude of the effect of the single factor hook shape on catch rates is likely substantially larger for leatherback sea turtles (ca. 63% lower catch rate on circle vs. J-shaped hook when bait type was not variable, Watson et al. 2005; Sales et al. 2010; Foster et al. 2012, Coelho et al. 2015; Santos et al. 2012, 2013) than for sharks (ca. 20% higher relative risk of capture on circle vs. J-shaped hook, Gilman et al. 2016b). While the benefit to leatherback sea turtles of using circle hooks relative to J-shaped hooks exceeds the cost to sharks in terms of catch rate effect,

the effect on catch levels, fishing mortality levels and relative risks to affected populations must also be considered and requires fishery specific assessment.

Use of circle hooks to benefit sea turtles may result in conflicting effects with additional species of conservation concern. Reinhardt et al. (2017), pooling data from two records, found that Atlantic bluefin tuna (*Thunnus thynnus*) had a significantly higher relative risk of capture on circle hooks than J-hooks, as is the case for most principal market tunas (Gilman and Hall 2015). Overfishing is not occurring for either of the two Atlantic ocean stocks, but the status of biomass is unknown due to extensive uncertainty with historical catch data (ICCAT 2017). Unlike other principal market tuna species which have extensive spawning grounds and protracted spawning seasons, bluefin species spawn in limited areas and have relatively short spawning periods of 1–2 months, making them particularly vulnerable to stressors, including from fishing mortality and outcomes of climate change (Collette et al. 2011; Muhling et al. 2011). Effects of pelagic longline hook shape on other taxa of conservation concern are less certain (Gilman and Hall 2015). Circle hooks may reduce odontocete (toothed whale) catch rates relative to J-shaped hooks of a similar size (Gilman et al. 2006a; Forney et al. 2011). Circle hooks also result in a higher probability of odontocetes being able to escape by throwing the hook or disentangling from line, resulting in less serious injuries (Forney et al. 2011). Circle hooks increase pelagic stingray catch rates relative to rates on J-shaped hooks of a similar size (Andraka et al. 2013; Gilman et al. 2016b).

While not included as a case study here, it is worth briefly mentioning that pelagic longline bait type poses similar conflicts as hook shape: when using small species of fish (e.g., pelagic 'forage' fishes, such as mackerels and species with mackerel-like characteristics) for bait, the catch and deep hooking rates are higher for some shark species but lower for primarily hard-shelled species of sea turtles relative to when squid is used for bait (Gilman et al. 2016b; Gilman and Huang 2017). Fish bait also has higher catch rates of some seabird species relative to using squid (Li et al. 2012).

Pelagic longline time-of-day and gear depth

Measures prescribing gear soak depth and time-of-day of fishing operations, such as deeper pelagic longline sets to reduce epipelagic sea turtle bycatch (e.g., NMFS 2004; FAO 2010), and longline night setting to reduce seabird bycatch (e.g., Monterey Bay Aquarium 2015; ACAP 2014, 2016), inadvertently exacerbates bycatch rates of other taxa of conservation concern in some fisheries. Gear soak depth and time-of-day of fishing operations significantly explain species-specific pelagic longline catch rates (Beverly et al. 2009; Young et al. 2010; Musyl et al. 2003, 2011; Gilman and Hall 2015; Gilman et al. 2017a). There is a substantial body of literature from electronic tagging and longline catch studies that indicate pelagic fishes and turtles partition themselves vertically (by depth and temperature) as well as horizontally (geospatially) by prey availability and primary environmental variables of hydrostatic pressure, temperature, and dissolved oxygen (Beverly et al. 2009; Bernal et al. 2010; Musyl et al. 2003, 2011).

For example, based on cluster analysis using temperature distributions determined from electronic tags, Musyl et al. (2011) grouped pelagic guilds into epipelagic (silky shark *Carcharhinus falciformis*, oceanic whitetip shark *C. longimanus*, marlins *Makaira* spp.), mesopelagic I (blue shark, shortfin mako shark *Isurus oxyrinchus*) and mesopelagic II (bigeye tuna *Thunnus obesus*, swordfish *Xiphias gladius*, bigeye thresher shark *Alopias superciliosus*) thermal niche categories. Furthermore, there is partitioning of epipelagic sharks based on body size and latitude (Young et al. 2010; Musyl et al. 2011). These clusters enable predicting relative susceptibility to capture and identifying viable bycatch mitigation methods (Rey and Muñoz-Chápuli 1992; Musyl et al. 2011).

There are synergistic effects of soak depth and the time-of-day of fishing activities on catch rates. Pelagic species' vertical habitat preferences can vary temporally, within a day, due to diel vertical migration cycles, time of day of active foraging, and temporal variability in diving behavior (Sedberry and Loefer 2001; Nakano et al. 1997, 2003; Ward and Myers 2005; Beverly et al. 2009; Musyl et al. 2003, 2011). Some species of pelagic apex predatory fishes conduct diel vertical migration cycles, mirroring movements of prey organisms comprising the deep scattering layer

(Carey et al. 1990; Schaefer and Fuller 2002; Weng and Block 2004; Young et al. 2010; Musyl et al. 2003, 2011). For example, blue sharks make frequent deep dives during the daytime and forage primarily near the sea surface at night (Carey et al. 1990; Musyl et al. 2011), making them susceptible to capture primarily in shallow-set longline gear, but also in deep daytime sets. Thresher sharks infrequently come near the surface and thus are largely susceptible to capture only in deep-set longline gear (Carey et al. 1990; Nakano et al. 2003; Weng and Block 2004; Musyl et al. 2011). Sea turtles also exhibit diel vertical migration patterns where transitions between daytime and nighttime are not as pronounced. Turtles conduct relatively deeper dives during crepuscular hours to reach prey migrating vertically through the water column, but during the daytime, their prey might be too deep or not worth the energy investment for sea turtles to dive to reach (Hays et al. 2006). Depending on the vertical diel distribution pattern of a particular species, the time of day of gear setting and haulback can also significantly affect catch rates in deep set longline fisheries because the terminal tackle passes through a relatively large section of the water column for an extended time period (Kerstetter and Graves 2006).

These interacting effects of gear soak depth and time-of-day of fishing operations are illustrated in Fig. 3 for selected pelagic teleost and shark species. Based on pop-up satellite archival tagging (PSAT) data, Musyl et al. (2011) estimated that fishing longline hooks deeper than ca. 100 m during the day resulted in higher catch of target species of bigeye tuna and swordfish while avoiding epipelagic species of sharks relative to fishing shallow at night (Fig. 3). This fishing strategy would avoid nearly 98% and 86% of oceanic whitetip and silky shark interactions, respectively, would reduce blue shark encounters by an estimated 53% and shortfin mako by 46%. This would, however, increase the catch rate of deeper-habitat mesopelagic II species, including bigeye thresher sharks (Fig. 3), which have long generation times and slow growth relative to other elasmobranchs (Dulvy et al. 2008).

Consistent with findings of Musyl et al. (2011), Table 1 summarizes the effects of various combinations of soak depth and time-of-day of gear soak on catch rates of the Palau tropical western and central Pacific Ocean pelagic longline fishery, based on analyses of observer and logbook program data (Gilman et al. 2017a). To summarize, in general, daytime deep-set longline gear, with all hooks soaking > 100 m, can achieve economically viable catch rates of swordfish and some target tunas, and have substantially lower catch rates of olive ridley and other hard shelled sea turtle species (Polovina et al. 2003, 2004; Rice and Balazs 2008; Hall et al. 2012; Cambiè et al. 2013; Watson and Bigelow 2014), epipelagic sharks (Galeana-Villasenor et al. 2008; Musyl et al. 2011; Bromhead et al. 2012; Caneco et al. 2014; Watson and Bigelow 2014), and istiophorid billfishes (Beverly et al. 2009; Musyl et al. 2011; Watson and Bigelow 2014) relative to shallow-set gear, where the majority or all hooks soak at depths < 100 m. However, deeper-set daytime gear has higher catch rates of some mesopelagic II species such as thresher sharks (Beverly et al. 2009; Musyl et al. 2011).

Variability in time-of-day of foraging behavior observed in pelagic fishes and sea turtles (e.g., Bigelow et al. 2002; Ward et al. 2004; Musyl et al. 2003, 2011) also occurs for pelagic seabirds. In some regions, night setting, prescribed in some fisheries to protect albatrosses and other primarily diurnal foraging seabird species has led to higher bycatch of nocturnal foragers (e.g. northern fulmars *Fulmarus glacialis*, Melvin et al. 2001b).

There can also be variability in vertical habitat distributions by age class, body size and sex within species. As a result, the time-of-day and depth of fishing affects the length frequency distribution, mean length and sex ratios of the catch of individual species (Musyl et al. 2003, 2011; Coelho et al. 2012; Gilman et al. 2017a). Sex-skewed catches can in turn affect mean lengths for species with sexual size dimorphism (differences in size by sex of the same age class, i.e., differential growth by sex).

Gear soak depth also significantly affects at-vessel survival rates. Because shallower, warmer water depths generally have lower dissolved oxygen concentration, higher stress occurs for fish caught at shallower depths due to the combination of temperature, limited oxygen and higher metabolic rates (Moyes et al. 2006; Gallagher et al. 2014). However, turtles and marine mammals caught on deeper hooks are more likely to drown before gear retrieval if they cannot reach the surface to breathe during the gear soak, and are more likely to experience decompression

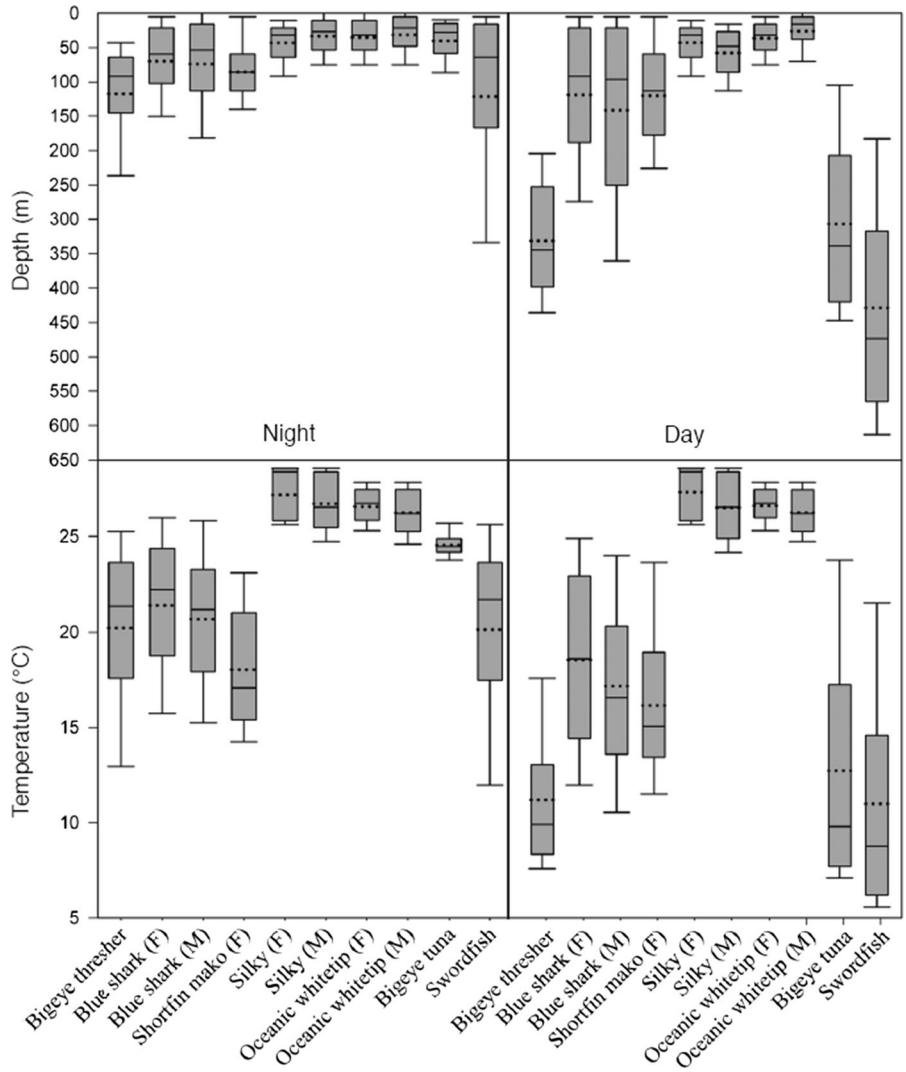

Fig. 3 Depth and temperature preferences for various pelagic teleost and shark species, determined from pop-up satellite archival tags. Boxes are 25% and 75% quartiles, solid horizontal line is the median (50% quartile), dashed horizontal line is the mean, and bars represent 10–90th deciles. M = male, F = female. [adapted from Musyl et al. (2011)]

sickness (the bends) if they survive the gear soak (Gilman et al. 2006c; Moore et al. 2009; García-Párraga et al. 2014).

Vertical habitat preferences also vary geospatially and temporally for individual species, and within species by age class, body size, and sometimes by sex. Therefore, the effect of pelagic longline soak depth on catch rates will vary spatially and temporally. The effect of soak depth on pelagic longline catch rates will vary both at relatively small spatial scales, such as whether gear is set near a shallow submerged feature or in the open ocean, and over very broad scales (100 s to 1000 s of km) due to changes in pelagic species' distributions, such as in response to large scale climate variability (e.g., inter-annual El Niño—Southern Oscillation and decadal Pacific Decadal Oscillation phases) and responses to outcomes of climate change (Gilman et al. 2016a). The effect of fishing depth will also vary due to regional and seasonal differences in variables that determine habitat preferences and migration corridors, including variability in the mixed-layer depth/thermocline, oxygen/temperature gradients, and other variables that determine habitat preferences, such as prey availability (Musyl et al. 2003, 2011).

Tuna purse seine set type

Tuna purse seine vessels make different set types, which can have substantially different catch

Table 1 Mean catch rates estimated from Palau pelagic longline observer and logbook program data for main market species and species/groups of conservation concern. (Gilman et al. 2017a)

	Catch rate (no./1000 hks)							
Soak category	BET	YFT	DOL	BILL	BSH	FAL	THR	LKH
Day shallow	0.9	7.8	1.1	0.4	0.4	1.0	0.1	0.2
Day intermediate	2.3	5.8	0.2	0.4	0.4	1.7	0.1	0.3
Day deep	2.4	2.9	0.2	0.3	0.3	0.2	0.4	0.0
Night shallow	1.9	1.6	0.4	1.7	1.7	2.2	0.4	0.5
Night intermediate	4.1	3.8	0.1	0.2	0.2	0.8	0.2	0.2
Night deep	3.7	2.4	0.0	0.0	ND	ND	ND	ND

Green: relatively high catch rate of market species (the first four species), and relatively low catch rate of shark and turtle species. Red: opposite of green. Yellow: intermediate catch rate

ND No data, *BET* bigeye tuna, *YFT* yellowfin tuna, *DOL* dolphinfish, *BILL* billfishes, *BSH* blue shark, *FAL* silky shark, *THR* thresher shark species, *LKH* olive ridley sea turtle

compositions. They make 'unassociated' or free swimming school sets (hereafter called school sets), where a school of tunas and other pelagic fishes is at or near the sea surface. Fishers detect the free swimming school when the fish disturb the sea surface and through the presence of seabirds foraging from above (Hall 1998; Hall and Roman 2013). Purse seine vessels also make sets on floating objects that aggregate pelagic organisms. This includes drifting logs, drifting algae, live and dead large marine organisms, marine debris (e.g. crates, pallets, nets), as well as artificial objects that are built and deployed by fishers and are designed specifically to aggregate pelagic fishes (Castro et al. 2002; Hall and Roman 2013). This latter category of artificial floating object is referred to as FADs (Gaertner et al. 2016). Drifting FADs are typically equipped with satellite buoys, and sometimes also with echo-sounders, to track the spatial position and estimate the biomass of fish aggregated at the device (Lopez et al. 2016; Gilman et al. 2018a). FADs can also be anchored, called rumpons in Indonesia, and payaos in the Philippines, usually used by small-scale fisheries, but in some regions (e.g., Papua New Guinea) are used by industrial purse seine fisheries (Beverly et al. 2012). Fishing vessels may also be used like a drifting FAD (e.g., seiners cooperating with bait boats, and purse seine vessels can be used to draw an aggregation away from a FAD) (Hall and Roman 2013). And in the Indian and Atlantic Oceans, purse seine vessels set on anchored supply vessels positioned over shallow seamount summits, similar to an anchored FAD (Amandè et al. 2008; Koehler and Moreno 2016). Primarily in the eastern Pacific, purse seiners make sets on schools of yellowfin tuna in association with dolphin pods, referred to as dolphin sets (Hall and Roman 2013).

The most appropriate approach to combine various set types to understand effects on the species composition and sizes of the catch is not always clear. Different school set types and different associated set types can have substantially different catch compositions (Dagorn et al. 2013; Hall and Roman 2013; ISSF 2017c; Peatman et al. 2017). Despite this, here we refer to the catch composition of combined school set types and combined associated set types. This enables simpler comparisons that are consistent with higher resolution purse seine set type categorizations employed by seafood certification and ranking programs and major seafood buyers (e.g., preferential sourcing of school tuna, ALDI 2016; certification under the Marine Stewardship Council standard differentiates between school and floating object sets, Acoura Marine 2017; Seafood Watch program ranks 'floating object purse seines' as red/avoid and 'unassociated purse seine' as yellow/good alternative, Monterey Bay Aquarium 2018), and because data

quality often prevents employing finer resolution categorizations of set types (Gilman et al. 2017c).

Conflicting ecological risks to bycatch species between purse seine sets on dolphins versus drifting FADs have been recognized for over 2 decades (Hall 1998). Measures adopted by fishers and management authorities to reduce dolphin mortality levels in eastern Pacific Ocean purse seine sets on dolphin pods (including conducting a backdown maneuver, deploying crew during backdown to help dolphins escape, using a Medina dolphin safety panel to reduce dolphin entanglement and increase sinking of the corkline during backdown, and fleet-wide and vessel-based dolphin mortality limits) have effectively achieved an order of magnitude reduction in annual, direct dolphin mortality levels. The number of dolphins killed per dolphin school set declined by about 98%, and the low annual dolphin mortality levels have been effectively maintained over the past 2 decades (Hall et al. 2001; Hall and Roman 2013). In 1990, when dolphin mortality levels had already declined by about 60% from a peak in 1986, in response to advocacy efforts by Earth Island Institute, an environmental NGO, some tuna canneries adopted policies to not purchase tuna caught in trips that included dolphin sets. As a result, U.S. purse seine vessels began making drifting FAD sets (Hall 1998). There was a temporary decline in the number of annual dolphin sets, which eventually reverted to pre-1990 levels (Hall et al. 2001; Hall and Roman 2013). However, the development of the drifting FAD fishery in the eastern Pacific resulting from the adoption of dolphin-safe policies has remained in place. The temporary benefit to affected dolphin populations from the dolphin-safe policy came at a lasting cost of an increase in bycatch of sharks and sea turtles, as well as in catch and discards of undersized tunas and non-marketable teleost species in drifting FAD sets (Hall 1998; Hall and Roman 2013).

Similar to the conflicting effects on bycatch rates between purse seine sets on dolphins versus drifting FADs, purse seine school and FAD, log and other associated set types have different catch compositions (Dagorn et al. 2013; Hall and Roman 2013; ISSF 2017c; Peatman et al. 2017). There is very large regional variability in purse seine catch rates by set type and taxonomic group (Table 2). Table 2 compares catch rates of selected species and groups in purse seine associated and school sets. Two rates are reported for some records when rates were available both in units of catch per 1000 sets, which may be an appropriate rate to use in fisheries subject to input controls (i.e., limits on effort, such as the annual number of sets or number of days fishing) and per 1000 t of landed target tunas (bycatch-to-catch ratio), which may be an appropriate rate to use in fisheries subject to output controls (i.e., limits on catch, such as annual catch levels of target tuna species).

Log and drifting FAD sets catch a larger number of species and a much larger biomass of nontarget species (i.e., species other than skipjack, yellowfin and bigeye tunas) than school sets (Hall 1998; Amandè et al. 2010; Hall and Roman 2013; Torres-Irineo et al. 2014; Gaertner et al. 2016; Peatman et al. 2017; Lezama-Ochoa et al. 2017). This is in part due to the slow drifting speed of FADs and logs relative to that of free-swimming tuna schools chasing prey and of schools of tunas and dolphins escaping speedboats. The majority of nontarget catch is comprised of teleost species that are not considered to be of conservation concern (e.g., kawakawa *Euthynnus affinis* and other small species of tunas, ocean triggerfish *Canthidermis maculata*, mackerel scad *Decapterus macarellus*, mahi-mahi *Coryphaena hippurus*, rainbow runner *Elagatis bipinnulata* and other carangids) (Dagorn et al. 2013; Hall and Roman 2013; ISSF 2017c). In most regions, a large proportion of the nontarget teleost catch is discarded (Hall and Roman 2013; ISSF 2017c). However, some tuna RFMOs are considering requiring full retention of these species.

Not accounting for entanglements in FAD subsurface appendages (which can be addressed by using non-entangling materials; ISSF 2015), shark catch rates, expressed as the weight of caught sharks per 1000 sets, were an order of magnitude higher in sets on drifting FADs and logs than in school sets (Atlantic and Indian Oceans, Amandè et al. 2008, 2010). When the shark catch rate is expressed as the weight of caught sharks per weight of retained market tunas, shark catch rates in school and associated sets are the same order of magnitude (ISSF 2017c). School sets have higher catch rates of mobulids (manta and devil rays, *Mobula* spp.) (Amandè et al. 2008, 2010; Dagorn et al. 2013; Hall and Roman 2013), of which several species are categorized as IUCN Endangered, Vulnerable and Near Threatened (Croll et al. 2016; IUCN 2017).

Table 2 Catch rates of selected species and groups in purse seine tuna fishery associated and school sets. Associated sets refers to sets on both drifting FADs and logs unless specified otherwise (Williams et al. 2009; Amandè et al. 2008, 2010; Dagorn et al. 2013; Hall and Roman 2013; ISSF 2017c)

Species/Group	Area	Catch rate	
		Associated sets	School sets
Bigeye tuna	Global	9% of catch	2% of catch
	Atlantic Ocean	14% of catch	5% of catch
	Eastern Pacific Ocean	28% of catch	1% of catch
	Indian Ocean	8% of catch	6% of catch
	Western and central Pacific Ocean	4% of catch	1% of catch
Yellowfin tuna	Global	16% of catch	35% of catch
	Atlantic Ocean	17% of catch	76% of catch
	Eastern Pacific Ocean	15% of catch	43% of catch
	Indian Ocean	25% of catch	72% of catch
	Western and central Pacific Ocean	14% of catch	22% of catch
Billfishes (mainly marlins, sailfish)	Atlantic Ocean	46.8 t/1000 sets	41.9 t/1000 sets
		1.0 t/1000 t landed tunas	0.4 t/1000 t landed tunas
	Eastern Pacific Ocean	1.4 t/1000 t landed tunas	0.7 t/1000 t landed tunas
	Indian Ocean	22.8 t/1000 sets	11.0 t/1000 sets
		0.6 t/1000 t landed tunas	0.1 t/1000 t landed tunas
	Western and central Pacific Ocean	0.4 t/1000 t landed tunas	0.6 t/1000 t landed tunas
Sharks (mainly silky and oceanic whitetip)	Atlantic Ocean	29.6 t/1000 sets	2.7 t/1000 sets
		4.6 t/1000 t landed tunas	8.5 t/1000 t landed tunas
	Eastern Pacific Ocean	2.3 t/1000 t landed tunas	1.2 t/1000 t landed tunas
	Indian Ocean	121.5 t/1000 sets	6.5 t/1000 sets
		5.1 t/1000 t landed tunas	1.1 t/1000 t landed tunas
	Western and central Pacific Ocean	1.4 t/1000 t landed tunas	0.6 t/1000 t landed tunas
Rays (mainly manta and devil)	Atlantic Ocean	5.7 t/1000 sets	30.3 t/1000 sets
		0.3 t/1000 t landed tunas	2.2 t/1000 t landed tunas
	Eastern Pacific Ocean	35 no./1000 sets	231 no./1000 sets
		0.0 t/1000 t landed tunas	0.2 t/1000 t landed tunas
	Indian Ocean	5.0 t/1000 sets	6.4 t/1000 sets
		0.1 t/1000 t landed tunas	0.7 t/1000 t landed tunas
	Western and central Pacific Ocean	0.1 t/1000 t landed tunas	0.2 t/1000 t landed tunas

Table 2 continued

Species/Group	Area	Catch rate	
		Associated sets	School sets
Sea turtles (mainly olive ridley, other hard-shelled species)	Atlantic Ocean	74.1 no./1000 sets	59.8 no./1000 sets
	Eastern Pacific Ocean	1.8 no./1000 sets	0.7 no./1000 sets
	Indian Ocean	15.9 no./1000 sets	0.8 no./1000 sets
	Western and central Pacific Ocean	2.8 no./1000 sets (drifting FADs)	6.1 no./1000 sets

The number of sea turtles captured in tuna purse seine sets (ca. hundreds per year globally) is extremely small relative to other gear types (e.g., gillnet, trawl and pelagic longline, Lewison and Crowder 2007; Gilman 2011). When captured, more than 90% are released alive (Williams et al. 2009; Amandè et al. 2008, 2010; Gilman 2011; Dagorn et al. 2013; Hall and Roman 2013). Sea turtle mortality also occurs from entanglement in FAD appendages and surface structures but data are unavailable to estimate mortality levels from these interactions. Turtle catch rates are higher in combined (drifting FAD and log) associated sets than in school sets in the Atlantic, Eastern Pacific and Indian Oceans (Amandè et al. 2008, 2010; Hall and Roman 2013). However, based on data from the western and central Pacific Ocean, turtle capture rates are higher in school sets than drifting FAD sets (Table 2) (Williams et al. 2009). Not shown in Table 2, turtle capture rates are highest in sets on large marine animals (e.g., whales, whale sharks *Rhincodon typus*, large mobulid rays), and are higher in log sets than in both school and drifting FAD sets (Williams et al. 2009). Leatherback turtles, an IUCN Vulnerable species, are caught ca. 90% more frequently in school than associated sets, as they tend not to aggregate at floating objects (Hall and Roman 2013).

Billfishes catch rates are the same order of magnitude in associated and school sets (Table 2). There were lower billfishes catch rates in associated sets in the western and central Pacific, and higher in associated sets in the other three regions (Amandè et al. 2008, 2010; ISSF 2017c). Some marlin stocks are overexploited (e.g., western central north Pacific Ocean striped marlin, ISC 2015).

Associated sets have higher skipjack and bigeye tuna catch rates than school sets, while school sets have higher yellowfin tuna catch rates (Dagorn et al. 2013; Harley et al. 2015). Drifting FAD sets also have higher catch of small, juvenile yellowfin and bigeye tunas than school sets (Fonteneau et al. 2013; Restrepo et al. 2017). Some bigeye and yellowfin tuna stocks are overfished (ISSF 2017b).

Environmental NGOs have prescribed seafood sourcing practices for canned tuna and several major retailers and food service companies have adopted canned tuna sourcing policies that identify canned tuna sourced from purse seine FAD sets as less ecologically sustainable than school sets, due to relatively higher bycatch rates in FAD sets (e.g., WWF 2011; ALDI 2016; Giant Eagle 2016; Wegmans 2016). These recommendations were very likely not based on assessments that accounted for the population effects of increased fishing mortality levels of species and groups with higher catch rates in purse seine school sets than in drifting FAD sets, including sea turtles, mobulids and yellowfin tuna.

This comparison does not account for other ecological risks, including habitat damage, fishing mortality from entanglement in in-use and derelict FAD surface and subsurface structures (when non-entangling designs are not employed), and potential effects of FADs on the behavior and ecology of species that associate with floating objects, possibly causing collateral population-level and broader ecosystem-level effects (Marsac et al. 2000; Hallier and Gaertner 2008; Amandè et al. 2010; Dagorn et al. 2013; Hall and Roman 2013; Sempo et al. 2013; Maufroy et al. 2015; Gaertner et al. 2016; Escalle et al. 2017; Gilman et al. 2018a).

Gillnet mesh size

Gillnets of a particular mesh size are relatively size-selective for most species of fishes, but can have poor species selectivity (Hamley 1975; Kirkwood and

Walker 1986; Holgård and Lassen 2002; Valdemarsen and Suuronen 2003; He and Pol 2010). Fish tend to get caught in one of the meshes in the gill region of its body (Bjordal 2002). The fish swims part way through a mesh, struggles to free itself, and the twine of the mesh slips under the fish's opercula (gill covers) preventing escape; i.e., the fish becomes 'gilled' (Bjordal 2002). Less frequently, a fish can also become wedged around the largest part of its body inside a gillnet mesh (Fujimori and Tokai 2001; He and Pol 2010). Parts of a fishes' body (fins, teeth, spine or other projection) may also become snagged on or tangled in the twine, where mesh size is not an important factor in explaining catchability (Murphy and Willis 1996; Price and Van Salisbury 2007; He and Pol 2010).

Of various gillnet gear design factors that affect size selectivity, including mesh size, netting color, twine material, number of filaments in the twine, hanging ratio, net dimensions, and twine diameter, mesh size has the largest effect for most species (He and Pol 2010). Mesh size has been regulated to balance the efficiency of catch rates of target species with the mitigation of catch rates of unwanted catch of juvenile and large adults of target and non-target species, including endangered, threatened and protected bycatch species (Price and Van Salisbury 2007; Gilman et al. 2010). However, conflicts can result from a prescribed gillnet mesh size. For example, as mesh size increases, catchability of sea turtles, marine mammals, some seabirds and other large organisms generally increases while catchability of small, juvenile age classes of target fish species decreases (Dagys and Zydelis 2002; Price and Van Salisbury 2007; Murray 2009; Orphanides 2010). Mesh size can also have conflicting effects on different elasmobranch species: The U.S. northwest Atlantic Ocean monkfish (*Lophius americanus*) set gillnet fishery had a 74% significantly lower spiny dogfish (*Squalus acanthias*) but 126% significantly higher thorny skate (*Amblyraja radiata*) bycatch rates in 35.6 cm (14 inch) than 25.4 cm (10-inch) mesh gillnets (Salerno et al. 2010).

Gillnet acoustic pingers

Acoustic pingers are used to reduce bycatch rates of some species of small cetaceans and seabirds in drift and set gillnets, and bycatch of some species of small cetaceans in pelagic and mid-water trawls, traps and other gears (e.g., Lien et al. 1992; Melvin et al. 2001a; Koschinski et al. 2006; Leeney et al. 2007; Berrow et al. 2009; Morizur et al. 2008; Carretta and Barlow 2011; Dawson et al. 2012). For example, a coastal drift gillnet with acoustic pingers attached to the float line and ends of each net significantly reduced the bycatch rate of common murres (*Uria aalge*) by 50% relative to control gear without pingers (Melvin et al. 2001a). Coastal set gillnets with active pingers attached to the float line had a 86% significantly lower Franciscana dolphin (*Pontoporia blainvillei*) catch rate than control nets (Bordino et al. 2002). In a controlled experiment in the Gulf of Maine coastal set gillnet fishery, there was a 92% significantly lower harbour porpoise (*Phocoena phocoena*) bycatch rate in gear with pingers than in control gear (Kraus et al. 1997). Analyses of observer data from the Gulf of Maine set gillnet fishery found a 60% lower porpoise bycatch rate in nets with pingers spaced at $\leq$ 92 m apart than in nets lacking pingers (Palka et al. 2008). Fisher compliance with prescribed pinger use can be one obstacle to achieving the magnitude of pinger bycatch mitigation efficacy observed in experiments (Dawson et al. 2012). For some species, some characteristics of the sound emitted by pingers may be unpleasant, causing the animals to avoid the area near the pingers, or the pinger may alert some species to the presence of the gear and enable them to avoid capture (Dawson et al. 2012).

However, for bottlenose dolphins (*Turisops truncatus*) and some species of pinnipeds, pinger broadcasts become a 'dinner bell', alerting and attracting them to the location of the gear, increasing their local abundance and rate of depredation of catch from the fishing gear, providing a food subsidy (Kraus et al. 1997; Melvin et al. 2001a; Bordino et al. 2002; Dawson et al. 2012; Stansbury et al. 2015). For example, Bordino et al. (2002) observed a significantly higher rate of sea lion (*Otaria flavescens*) depredation in set gillnets with pingers than in control gear, and the sea lion depredation rate on nets with pingers increased throughout the study period. Similarly, Melvin et al. (2001a) observed a significantly higher abundance of harbour seals attending a drift gillnet with pingers than gillnets lacking pingers. This increased abundance and depredation activity by pinnipeds in the vicinity of the fishing gear in turn increases their catch risk (Hamer et al. 2011; Dawson et al. 2012), which may offset population-level

benefits resulting from the food subsidy from depredation. For example, bottlenose dolphins have been captured in trammel nets and gillnets containing relatively high-intensity (loud) pingers designed to deter depredation (Northridge et al. 2003; Dawson et al. 2012). It is as yet unclear if pingers can be designed to avoid and minimize detection by species of pinnipeds and small cetaceans that depredate the catch from fishing gear, such as by using a frequency that they cannot detect, emitting randomized signals, or emitting a range of frequencies (Dawson et al. 2012).

Temporal and spatial fishery closures

There are numerous examples of how marine area closures can result in the spatial or temporal displacement of bycatch of an individual species or age classes (e.g., Powers and Abeare 2009; Diamond et al. 2010; Murray et al. 2000; Suuronen et al. 2010; SPC 2010; Sibert et al. 2012). Although less well documented, there are also examples of closed areas designed to reduce the bycatch of one species of conservation concern increasing the bycatch of another, and increasing fisheries degradation of benthic communities at other sites.

For example, area closures for a U.S. north Pacific Ocean demersal trawl fishery effectively met the objective of reducing the bycatch rate of king crab (*Paralithodes camtschaticus*), but resulted in an increased bycatch rate of Pacific halibut (*Hippoglossus stenolepis*) due to the spatial displacement of effort (Abbott and Haynie 2012). The fishery, which targets rock sole (*Lepidopsetta polyxysra*) roe and Pacific cod (*Gadus microcephalus*), had seasonal king crab and Pacific halibut bycatch limits due to concerns over the conservation status of king crab and the allocation of Pacific halibut to coastal fisheries that target this species. An additional unintended consequence of the closures was a change in fisher behavior to increase catch rates of cod and decrease catch rates of rock sole (Abbott and Haynie 2012).

Baum et al. (2003) modelled the effects on catch rates from the closure of areas to the U.S. north Atlantic swordfish longline fishery. A portion of the fishing grounds called the Northeast Distant statistical area was closed for about 3 years, staring in 2000, to reduce loggerhead (*Caretta caretta*) and leatherback (*Dermochelys coriacea*) sea turtle bycatch (NMFS 2001). The spatial displacement of effort was modeled to simulate either a change in effort to achieve constant levels of swordfish catch, or to maintain a constant level of effort. Both scenarios for the closure of the Northeast Distant area were predicted to reduce catch rates of loggerhead and leatherback sea turtles and two species of sharks of relatively low conservation concern (blue and shortfin mako sharks, categorized as Low Risk by the IUCN Red List of Threatened Species), but increase catch rates of 10 shark species, including two categorized as Vulnerable by the IUCN Red List (blacktip *Carcharhinus limbatus* and dusky *C. obscurus* sharks) (IUCN 2017). The model simulations predicted mixed effects on catch rates of teleosts, with increased catch rates of some species of discarded teleosts (Baum et al. 2003).

In 2001, to reduce pressure on the depleted North Sea cod (*Gadus morhua*) stock, the European Commission temporarily closed to certain fishing fleets a large cod spawning area in the central North Sea. Rijnsdorp et al. (2001) observed that the closed areas resulted in the displacement of beam trawling effort to areas with increased catch rates of sensitive demersal and benthic species. Beam trawling effort redistributed to the western North Sea, where according to Rijnsdorp et al. (2001), restricted-range populations of rays and benthic vulnerable marine ecosystems occur. Similarly, Dinmore et al. (2003) predicted that continuation of the seasonal cod area closures, and concomitant continued displacement of trawling effort to areas that had not been trawled prior to the seasonal area closures, would increase cumulative impacts to benthic invertebrate production and cause localized reductions in benthic biomass.

Are bycatch mitigation methods drivers of fisheries-induced evolution?

FIE has unequivocally been demonstrated using transcriptomics on specific heritable growth traits for fish populations in the laboratory (van Wijk et al. 2013) and, by inference, genomic diversity in the wild (e.g., Hauser et al. 2002; Hutchinson et al. 2003; Thériault et al. 2008). But the full impact and scope of FIE has not yet been investigated (Heino et al. 2015). Employment of a bycatch mitigation method may inadvertently cause or strengthen a fishery's

intraspecific heritable trait-selective mortality, resulting in FIE. These changes in genetic diversity within species can be protracted or permanent (Kuparinen and Merila 2007; Heino et al. 2015) where causes of changes in traits from human predation outpaces most natural causes (Darimont et al. 2009). The relative catchability of individuals of a population susceptible to capture by a fishing gear type is explained, in part, by various heritable traits that show inter-individual variability. This includes, for example, behavioral traits for shyness/boldness; life-history traits such as age and size at maturation and growth rate (size-at-age); physiological traits such as metabolic rate (which may be correlated with shyness/boldness), visual acuity and swimming performance; and morphological traits such as mouth dimensions (e.g., gape width and height) and body shape (Heino et al. 2015; Lennox et al. 2017; Hollins et al. 2018). We are unaware of any empirical research on bycatch mitigation method effects on FIE, and more generally, FIE in taxa other than teleosts and FIE resulting from selection on physiological, morphological and behavioral traits are not well understood (Heino et al. 2015; Hollins et al. 2018). As a result of these gaps, here we present only theoretical examples of potential ways that employment of bycatch mitigation methods might cause or increase a fishery's intraspecific selectivity of heritable traits that are variable within an affected population, in particular for populations with small effective sizes.

Changes in passive (static) fishing gear designs to mitigate unwanted bycatch can affect the gear's visual and olfactory detectability. The capture process for passive fishing gears (e.g., gillnets, longlines, traps, hook-and-line gears) relies, in part, on the movement of organisms into the gear, where, among many other factors, detectability of the gear affects fishing efficiency (Bjordal 2002; He and Pol 2010). For instance, replacing wire with monofilament for leaders used in pelagic longline branchlines conducted to reduce shark catch rates has resulted in higher catch rates of some teleosts, perhaps due to lower detectability of the monofilament leaders (Ward et al. 2008; Gilman et al. 2016b). This change in gear design might increase the probability of catching turtles with traits for relatively poorer vision: While the continuum of vision capacity of all individuals of a turtle population might enable the entire population to detect a wire leader next to a baited hook, only turtles with traits for acute vision or flicker fusion (to detect contrast) might be able to see monofilament leaders, eliciting avoidance and concomitant lower probability of capture than if they could not detect this component of the branchlines. This would increase the gear's selective removal (increasing the relative risk of capture) of the subset of the population with heritable traits for relatively poor vision (Gilman et al. 2016b).

Prescribing a minimum hook size to reduce sea turtle catchability (Gilman and Huang 2017) increases the size selectivity of some components of the catch (Gilman et al. 2018b), favoring traits for faster life histories, resulting in, for example, a reduction in the average size at first reproduction (Zhou et al. 2010). The larger the hook, the larger an organism's mouth dimensions needs to be to fit it in its mouth (Erzini et al. 1997; Gilman et al. 2018b). As a result, for species that tend to be caught by ingesting a baited hook, hooks with a larger minimum width reduce the relative catchability of smaller length classes within a species (Lokkeborg and Bjordal 1992; Cooke et al. 2005). This same concept could be applied to other gear types where a change in gear design or fishing method to mitigate problematic bycatch alters the length frequency distribution of the catch, including for example from changes in gillnet mesh size, discussed in the section "Gillnet mesh size".

The capture process for mobile fishing gears (e.g., trawls, purse seines) involves actively encircling or towing the gear into the path of the target species (Bjordal 2002). Some pelagic trawl captains, on very rare occasions when they detect a dolphin or porpoise in the net, reduce their tow speeds to attempt to allow the marine mammals to escape. A slower tow speed might increase intraspecific selectivity of fishes and marine mammals susceptible to capture, as it may reduce relative catch risk and hence favor genotypes possessed by members of a population for physiological traits such as for anaerobic swimming (i.e., sprinters), other swimming performance traits, and perhaps behavioral traits supporting escapement with slower trawl tow speeds (e.g., Gabr et al. 2007; Hollins et al. 2018).

Measures that reduce tuna purse seine sets on FADs and increase school sets in order to reduce shark bycatch rates (refer to the section "Tuna purse seine set type") might increase selectivity for individuals of a species with heritable traits favoring maintaining schooling behavior when experiencing 'predation' by

purse seine gear. This could occur if the intraspecific variability in behavior of remaining in or leaving a school during a predator attack (Pitcher 1995; Pavlov and Kasumyan 2000) occurs when experiencing predation when in a free swimming school but not when aggregated in a shoal in a fixed location such as under a drifting or anchored FAD. Under natural conditions, individuals possessing traits for atypical behavior of discontinuing schooling behavior when the school is being preyed upon would have the highest predation risk (Pitcher 1995; Pavlov and Kasumyan 2000), but when an unassociated school is being pursued by a purse seine, this minority trait would improve their survival probability (assuming that they survive following escape in a suboptimal school size, such as by merging with another school).

Vessels fish deeper to avoid surface-habitat bycatch species, such as by pelagic trawls to avoid dolphin bycatch (e.g., Thompson et al. 2013), and pelagic longlines to avoid sea turtles and epipelagic species of elasmobranchs (FAO 2010; Gilman 2011; Musyl et al. 2011; and see the section "Pelagic longline time-of-day and gear depth"). This increases the relative catchability and selective fishing mortality of individuals of a population possessing physiological traits that enable them to make use of deeper habitat where lower temperatures and oxygen concentrations occur (Musyl et al. 2011; Hollins et al. 2018).

Spatio-temporal bycatch mitigation measures can also result in unintended FIE. This would occur if a time-area fisheries closure displaces fishing effort to times (time-of-day, season) and sites where a narrower proportion of the variations of a trait that exists for an affected population occurs. For example, because many marine species exhibit temporal and spatial variability in habitat use by size (e.g., sharks, Stevens and McLoughlin 1991; teleosts, Solmundsson et al. 2015), an MPA that displaces fishing effort to an area with a narrower length frequency distribution of a species could result in an increase in size selectivity. More specifically, if small size classes of a species occur primarily in nearshore habitats while larger individuals are distributed offshore, an MPA that restricts fishing to offshore waters that increases selection for larger organisms would favor heritable traits for faster life histories (e.g., earlier maturation) (Heino et al. 2015). MPAs could, however, be designed to reduce FIE (e.g., Hall 2002; Dunlop et al. 2009).

If employment of a bycatch mitigation method inadvertently causes intraspecific heritable trait-based selectivity, this would reduce genetic diversity of affected populations by reducing the occurrence of phenotypes for traits associated with higher catchability. Reduced intra-population genetic diversity reduces a population's fitness, resistance and resilience to natural pressures and concomitant persistence, and can compromise natural selection and the ability to evolve in response to environmental changes and stressors, also increasing the risk of population extirpation (local extinction) as a result of the reduction in adaptive genetic variation (Saccheri et al. 1998; Westemeier et al. 1998; Reed and Frankham 2003; Ehlers et al. 2008; Evans and Sheldon 2008). Not accounting for FIE-influenced changes in life-history characteristics in stock assessments can compromise the robustness of model estimates (Audzijonyte et al. 2016). Similarly, bycatch mitigation measures may inadvertently exacerbate fishing mortality of phylogenetically distinct species, compromising the potential for evolution at community and ecosystem levels (Kareiva and Marvier 2003; Redding and Moores 2006; Gilman et al. 2014a, b). These examples highlight the importance of assessing and managing fisheries bycatch impacts across levels of biodiversity, including genetic diversity, impacts on individual populations and species, and broader community- and ecosystem-level effects (e.g., McNeely et al. 1990), discussed in the next section.

How to transition to integrated bycatch assessment and management

The examples presented here illustrate how some bycatch mitigation methods reduce catch and mortality of one species or group of conservation concern but exacerbate catch and mortality of another. Theoretically, employment of some bycatch mitigation methods might inadvertently reduce genetic diversity within populations, causing or exacerbating FIE. Bycatch mitigation can also affect evolutionary processes of communities by inadvertently increasing direct or collateral fishing mortality of phylogenetically distinct species. Some bycatch mitigation measures, through food web links, cause or strengthen drivers for broad community- and ecosystem-level changes in functions, structure and resilience, with

concomitant effects on the provision of targeted ecosystem services, including fishery yields (Pace et al. 1999; Stevens et al. 2000; Ferretti et al. 2010). These examples support moving from piecemeal to holistic bycatch assessment and management so that the relative risks across affected components of biodiversity resulting from fisheries bycatch are identified and accounted for in fisheries decision-making processes. Intentional cross-taxa bycatch tradeoffs could be designed to balance exploitation, distributing fishing mortality across populations at sustainable levels, preserving the state of the ecosystem characteristic of unexploited conditions (Hall 1996; Zhou et al. 2010; Garcia et al. 2012).

Ecological risk assessments

The transition to holistic bycatch assessments would be supported by adapting Productivity-Susceptibility Analyses (PSAs) and other methods for semi-quantitative and quantitative, model-based ERAs of the effects of fisheries bycatch to assess relative risks to affected populations, cumulative effects from regional fisheries, and to assess broader ecosystem-level effects. ERAs of the effects of fisheries bycatch need to move beyond single taxa species-level assessments to broader, more holistic assessments of risks across levels of biodiversity, including genetic diversity, impacts on individual populations and species, and broader ecosystem-level effects. Methods for ERA of the effects of fishing have recently been developed for the continuum of data-poor to data-rich fisheries. ERA methods include rapid, first order, qualitative evaluations, semi-quantitative assessments, and model-based quantitative assessments (Dambacher et al. 2009; Hobday et al. 2007, 2011; Weijerman et al. 2015).

The objectives of analysis of most semi-quantitative fisheries ERAs have been to determine population- and species-level relative risks from fishing mortality of individual taxonomic groups of conservation concern, most employing PSAs (e.g., Stobutzki et al. 2002; Waugh et al. 2008; Cortes et al. 2015). PSAs assess productivity through use of attributes for intrinsic factors, such as demographic characteristics of a population, stock or species. These productivity attributes provide an indicator of relative resistance to fishing mortality and resilience or ability to recover from depletion. Susceptibility considers extrinsic factors that influence catch risk, such as encounterability, selectivity and post-release mortality, where some attributes are modified by the use of mitigation strategies (Patrick et al. 2009; Hobday et al. 2011; Gilman et al. 2017b). Species with high productivity and low susceptibility have lower risk from a fishery than species with lower productivity and higher susceptibility scores (Fig. 4).

PSAs and other semi-quantitative ERAs of the effects of fisheries bycatch provide a first order assessment of relative risk, and support prioritizing the focus of more rigorous quantitative assessments that estimate absolute population-level effects from fishery removals (Hobday et al. 2011). PSAs identify the relative risk to individual populations from the effects of fisheries bycatch. Populations identified in the PSAs as having the highest risk could then undergo robust quantitative modeling to simulate absolute risks to individual populations that would result from implementing alternative bycatch management measures. This would identify any cross-taxa conflicts, information needed to holistically manage bycatch.

The objectives of analysis of model-based quantitative methods for ERAs of the effects of fishing include to assess absolute risk, e.g., of population extirpation, irreparable harm to a stock, habitat loss, and loss in genetic diversity (Hobday et al. 2011; Zhou et al. 2011). Conventional single stock assessment methods are the most common form of a quantitative ERA, used to assess the status and temporal changes in stock status and predict stock responses to different

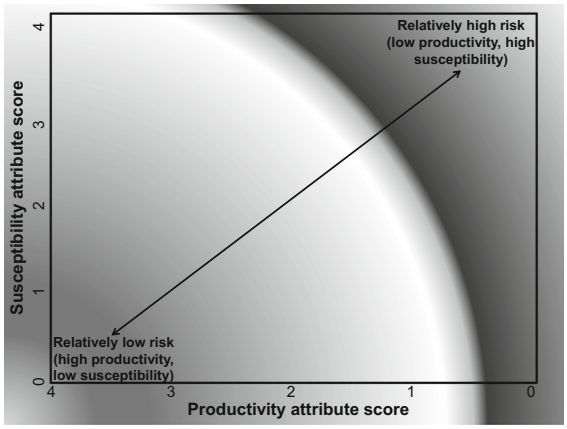

Fig. 4 Template productivity-susceptibility analysis plot, an example of a semi-quantitative ecological risk assessment of the effects of fishing on populations, stocks, species or habitat. [adapted from Hobday et al. 2011; Williams et al. (2011)]

management options (Hilborn and Walters 1992). They are typically used for principal market species but more recently have also been used for stocks of sharks and other incidentally caught species of conservation concern, including through new assessment methods designed for data-poor stocks (e.g., Fu et al. 2016).

Numerous analytical approaches have been used in quantitative ERAs of the effects of fishing to define threshold population sizes and fishing mortality rates beyond which populations are at risk of irreparable harm or extirpation, with a range of data requirements and concomitant range in certainty in outcomes (Wade 1998; Musick 1999; Chaloupka 2003; Lewison and Crowder 2003; Dulvy et al. 2004; Hobday et al. 2011; Zhou et al. 2011). Findings from population models can identify the relative risks from alternative bycatch mitigation methods on absolute population-level effects, guiding management responses. For example, there are now numerous adaptations of a model that does not require detailed demographic information, designed to determine mortality thresholds for marine mammals, known as Potential Biological Removal (PBR) (Wade 1998), including for seabirds (Dillingham and Fletcher 2011; Richard and Abraham 2013) and sea turtles (Curtis et al. 2015; Casale and Heppell 2016). A more rigorous population simulation modeling approach than PBR that also can be applied given minimal data compares relative competing risks of various mortality sources, including from fisheries bycatch (Chaloupka 2003). Examples of biological reference points that define a threshold beyond which increased fishing effort and mortality risks irreparably damaging a fish stock include F_{crash}, the fishing mortality rate that will drive a population to 1/1000 of virgin biomass, and B_{crit}, the minimum viable population size below which population extirpation is imminent (Mace 1994; Zhou et al. 2011; Gilman et al. 2014a).

As applied in stock-specific harvest strategies, multispecies and ecosystem-based limit reference points can also be defined as thresholds for undesirable conditions for an ecosystem. When stock-specific limits are exceeded, this triggers pre-agreed controls designed to rebuild the stock, thus eliminating the risk of serious or irreversible harm or extirpation. Similarly, when ecosystem-level thresholds are exceeded, controls would be triggered that alter the level of pressures that caused the unwanted change in ecosystem state (Sainsbury et al. 2000; ICES 2001; Collie and Gislason 2001; Gilman et al. 2014b, 2017b).

Global biodiversity research and conservation activities have generally focused on species-level criteria (see review by Gilman et al. 2011). Methods for ERAs of the effects of bycatch fishing mortality that comprehensively consider biodiversity across its hierarchical manifestations are needed. Multispecies and ecosystem models are additional examples of quantitative ERAs that can be designed to determine ecosystem changes in response to pressures, including from bycatch removals, and simulate ecosystem effects of alternative bycatch management approaches (e.g., Ecopath with Ecosim: Polovina 1984; Walters et al. 1997; Christensen and Walters 2004; Atlantis: Fulton and Smith 2004; Fulton et al. 2004; Models of Intermediate Complexity for Ecosystem assessment: Plaganyi et al. 2014; and qualitative modeling: Dambacher et al. 2009, 2015). Different ecosystem models vary in their balance between realism, accuracy and complexity, and have a wide range of data requirements (Weijerman et al. 2015). For example, the potential ecological and economic benefits of eliminating bycatch (the "maximum dexterity", or "clean fleet" scenario) were explored using simulations with food web models of ecosystems in northern British Columbia, Canada and in Raja Ampat, Indonesia (Pitcher and Ainsworth 2010). In both systems, increased economic returns were suggested by long-term simulations encompassing 100% or 80% bycatch reductions.

ERAs should also assess risks from the effects of bycatch removals on intraspecific genetic diversity and phylogenetic diversity resulting from selective fishery removals. The relative risks of intra-population FIE from alternative bycatch management methods could be predicted (e.g., Kenchington et al. 2003; Kuparinen and Merila 2007; Heino et al. 2015; Hollins et al. 2018). Similarly, evaluation of effects of bycatch management options on genetic diversity among species should also be conducted. An ERA of the effect of a pelagic longline fishery by Gilman et al. (2014a) compared species' PD_{50} indices, which is the expected phylogenetic diversity (PD) loss if the species goes extinct, assuming all other species have a 50% probability of persistence. This index of phylogenetic uniqueness enables an assessment of species' relative importance for maintaining unique

evolutionary history and for the potential continuation of evolutionary processes (Faith 2008; Sgro et al. 2011). However, evolutionary histories are not available for all taxonomic groups, and, discussed below, there is no agreed standardized way to compare the relative taxonomic distinctness of species from unrelated groups.

Data quality

Robust assessments of the relative risks to individual populations from fisheries bycatch require adequate data quality. Improvements in some fisheries monitoring and research programs are needed to identify cross-taxa conflicts resulting from bycatch mitigation methods to support holistic bycatch management. Fisheries management systems, even those that are rudimentary and governing data-poor fisheries, can successfully transition to meeting many of the data requirements of integrated bycatch assessment and management (Gilman et al. 2017b). Cross-taxa conflicts likely occur in many gear types, but have not been documented due to inadequate monitoring or gaps in research. For example, data gaps need to be filled to determine if increasing the fishing depth of pelagic trawls to reduce dolphin bycatch (e.g., Thompson et al. 2013) increases bycatch of mesopelagic species, such as thresher sharks. Similarly, data assessing fishers' response to bans on wire leaders in pelagic longline fisheries to reduce shark catch rates are needed to determine if this resulted in increased seabird catchability. When required to replace wire with monofilament leaders, pelagic logline fishers may respond by attaching branchline weights further from the hook due to safety concerns—if a branchline breaks during hauling, which frequently occurs when sharks are caught and sever leaders not made of wire or other durable material, the weight can fly at the vessel at high velocity when the weight is located near the hook—reducing the baited hook sink rate and increasing seabird bycatch rates (Gilman et al. 2016b). Monitoring methods require adjustments in some fisheries to enable the identification of cross-taxa conflicts resulting from alternative bycatch mitigation methods. For example, data fields and data collection protocols by onboard observers require expansion (e.g., Gilman and Hall 2015).

Data also need to be collected to determine which populations of species of conservation concern are captured in a fishery, a prerequisite to conducting quantitative ERAs that estimate absolute population-level effects from bycatch removals. As an example, there is minimal information on which cetacean species are captured in most Pacific Island pelagic longline fisheries, and little knowledge of the structure and conservation status of populations of cetacean species that overlap with these fisheries (SPREP 2007; Gilman et al. 2013). Minimally invasive tissue biopsy samples could be collected and analyzed to determine which populations are caught in a fishery.

Even when information on which populations are captured in a fishery is available, data requirements for robust population models are frequently not met. For many bycatch species of conservation concern, improved information is required on key life history characteristics, information to determine population structure, and accurate estimates of all sources of mortality, including total fishing mortality by age class and sex (e.g., NMFS 2012; Camargo et al. 2016; Gilman et al. 2017b).

Similarly, ERAs on the effects of bycatch require improvements in understanding of effects on genetic diversity within and between species. Increased knowledge is needed of the phenotypic ranges of intraspecific diversity of traits that explain susceptibility to fisheries capture (Hollins et al. 2018). FIE in species of conservation concern (i.e., taxa other than teleosts) and FIE driven by selectivity for physiological, morphological and behavioral traits (i.e., traits other than life-history attributes) require improvements in understanding (Heino et al. 2015; Hollins et al. 2018). Expansions of data collected by fisheries monitoring programs are also required. For instance, monitoring temporal trends in genetic diversity within populations requires sufficient time series of genetic markers for traits affected by selective fishery removals, which requires long-term collection of tissue samples for the extraction of DNA (Kenchington et al. 2003; Heino et al. 2015).

Improvements are needed in many data inputs for models of ecosystem-level effects of fishing and evaluation of bycatch management options (one component of fisheries management strategy evaluation), including from bycatch removals (e.g., Weijerman et al. 2015). Monitoring methods require adjustment to supply data needed for fisheries managers to track temporal changes in indicators of the state of ecosystem components that are sensitive to

bycatch fishing mortality. Ecosystem state indicators that are sensitive to bycatch fishing mortality include, for example: size structure within and across trophic levels, the number and biomass of functional groups within trophic levels, mean trophic level of the catch, the biomass of bycatch species' prey, the biomass of vulnerable species, the ratio of the biomass of pelagic to demersal fishes, and response diversity (the variety of responses to pressures and environmental changes by components of a facet of biodiversity—genotypes within a species, species within a functional group, functional groups within a trophic level, etc.) (Elmqvist et al. 2003; Fulton et al. 2005; Link 2005; Suding and Hobbs 2009; Coll et al. 2016).

Decision support tool for integrated bycatch assessment and management

Progress towards holistic bycatch management could be expedited through developing and having fisheries management authorities use decision support tools for bycatch assessment and management. The tools would benefit fisheries managers who struggle with conventional, piecemeal, bycatch assessment and management, as an integrated approach would be more efficient through employing a single, comprehensive bycatch program in place of separate taxa-specific programs. The tool's identification of which risk assessment methods are feasible given available data inputs would reduce complexity. The tool could augment the ability of fisheries management authorities to account for findings of integrated bycatch ERAs that identify cross-taxa conflicts resulting from alternative bycatch mitigation approaches. Use of the decision support tool could reduce the likelihood of having political attention respond to taxa-specific advocacy and instead maintain a holistic focus. The decision support tool would need to address several complex issues where there is no unequivocal, most rigorous approach, including how to determine the relative risk to species of different taxonomic groups, how to compare indices of phylogenetic uniqueness between species of unrelated groups, and how to account for uncertainty in ecosystem model simulations of alternative bycatch management approaches (Isaac et al. 2007; Pitcher and Ainsworth 2010; Gilman et al. 2014b). The tool could identify approaches to increase the likelihood of fisher uptake of integrated bycatch management measures, where fisher-led methods to reduce discarded bycatch can improve food security through a resilient fishing strategy "across the food web" (Pitcher and Lam 2010). It would provide managers with requisite information to enable them to then make agonizing choices when designing holistic, integrated bycatch management measures so that unavoidable tradeoffs are intentional and acceptable. Existing tools for management strategy evaluation could be employed for this purpose. An Atlantis ecosystem model developed for the southeast Australian Commonwealth scalefish (*Seriolella brama*) and shark fishery, a complex ecosystem model used for quantitative management strategy evaluation (Smith et al. 2007; Fulton et al. 2014), is an example of an existing management strategy evaluation method that could be employed as a decision support tool for integrated bycatch management. Integrated bycatch management strategy evaluation would simulate what effect alternative bycatch management measures would have on catch rates and levels of each species susceptible to capture in the fishery, providing information to support decisions on what are acceptable tradeoffs.

In addition to being designed to account for cross-taxa conflicts of alternative bycatch management measures, the bycatch decision support tool could adapt wetlands and terrestrial sequential mitigation hierarchy (e.g., EPA 2015) to provide decision support for broader management of fisheries bycatch (Wilcox and Donlan 2007; Finkelstein et al. 2008; UNEP-WCMC 2016). The sequence of bycatch management steps are: avoidance, minimization and offsets or compensatory mitigation. The tool could be designed to account for the fishery-specific context, including the existing management system, such as resources for surveillance and monitoring, and current conservation and management measures (input, output, temporal- and spatial- restrictions, gear and fishing method restrictions). And, the tool could evaluate and rank alternative bycatch mitigation methods, and combinations of methods, against criteria for efficacy, the economic viability, practicality, safety and ability to facilitate compliance monitoring of each bycatch mitigation method, in addition to a criterion on cross-taxa conflicts resulting from each method (WPRFMC 2018).

A decision support tool should be designed so that it could be effectively used across types of fisheries, from data-poor to data-rich fisheries, from artisanal/

small scale to industrial/large scale fisheries, and from fisheries with relatively rudimentary management systems with relatively low institutional and financial resources to fisheries with relatively robust management systems and ample resources. The tool could be designed to enable employment from local to regional fisheries, where the latter approach enables comprehensively assessing and managing cumulative impacts.

In summary, tools would support integrated ecological risk assessment and management of the effects of fisheries bycatch could:

- Characterize and benchmark existing knowledge of bycatch ecological risks, contemporary bycatch management framework, and fishing practices and gear designs that affect bycatch
- Guide end users to select ecological risk assessment methods that are feasible to implement given available data quality
- Evaluate and rank bycatch mitigation methods following sequencing hierarchy, and considering cross-taxa conflicts, bycatch mitigation efficacy, practicality, crew safety, economic viability and feasibility to monitor compliance
- Conduct bycatch management strategy evaluation, enabling the adoption of acceptable tradeoffs where unavoidable
- Be designed to be as simple as possible to maximize the likelihood of successful on-the-ground implementation
- Be adaptable for use across fisheries, from data-poor to data-rich, artisanal/small-scale to industrial/large scale, and from those with rudimentary to robust management frameworks, and from local (stage 2, Fig. 5) to regional scale (stage 3, Fig. 5).

Approaches to bycatch management can be described as progressing along a continuum from conventional narrow-scale, fishery-by-fishery and species-by-species, to nested-scale ecosystem-based fisheries management (EBFM), to cross-sectoral marine ecosystem-based management (EBM) (Fig. 5). The capacity to progress along this continuum will vary by fishery, country and region (Pitcher et al. 2009; Gilman et al. 2017b). Most fisheries management frameworks that manage bycatch are at the first stage of Fig. 5, where bycatch is managed fishery-by-fishery and species-by-species. The

Fig. 5 Transitioning from conventional narrow-scale and species-by-species bycatch management to nested-scale EBFM, to cross-sectoral marine EBM. (Sainsbury et al. 2000; Link 2005; Gilman et al. 2017b)

decision support tool would aim to get fisheries to stage 3, with integrated bycatch management at a nested-scale, across regional fisheries. EBFM augments conventional target species single-stock, single-fishery management approaches to address broader effects of fishing (FAO 2003; Garcia et al. 2003). The EBFM extension to single-stock reference points is ecosystem-based reference points, which account for ecosystem effects of fishing activities (Rice 2000; Pitcher 2001; Rochet and Trenkel 2003). Similarly, EBFM extends single stock management through ecosystem-based harvest strategies. EBFM is one component of holistic marine EBM, which requires cross-sectoral planning and management of marine pressures (Pikitch et al. 2004; Crowder and Norse 2008; Link and Browman 2014). Successful mitigation of the main global drivers of change and loss in marine biodiversity that adversely affect the fishing industry but are largely caused by other industry sectors, including marine pollution, outcomes of climate change, habitat degradation, and invasive alien species (Leadley et al. 2010) will increasingly require effective cross-sectoral collaboration. These developments would lead, hopefully, from old-style management that tends to be utopian, narrow, and prescriptive, to a new era of practical, holistic, adaptive and industry-supported strategies (Pitcher and Lam 2010).

Acknowledgements This study was supported by The Nature Conservancy Indo-Pacific Tuna Program. We are grateful for constructive comments provided by two anonymous peer reviewers. This article is based roughly on a 2017 presentation made by EG and FP at the Marine Biodiversity, Exploitation and Conservation research lab (MARBEC) and a blog by EG published on the International Seafood Sustainability Foundation website.

References

Abbott J, Haynie A (2012) What are we protecting? Fisher behavior and the unintended consequences of spatial closures as a fishery management tool. Ecol Appl 22:762–777

ACAP (2014) Demersal and pelagic longline night setting. Bycatch Mitigation Fact Sheet 5. Agreement on the Conservation of Albatrosses and Petrels, Hobart

ACAP (2016) ACAP summary advice for reducing the impact of pelagic longline fisheries on seabirds. Agreement on the Conservation of Albatrosses and Petrels, Hobart

Acoura Marine (2017) MSC sustainable fisheries certification. PNA Western and Central Pacific Skipjack and Yellowfin, Unassociated/Non FAD Set, Tuna Purse Seine Fishery. Acoura Marine, Edinburgh

Afonso AS, Santiago R, Hazin H, Hazin FHV (2012) Shark bycatch and mortality and hook bite-offs in pelagic longlines: Interactions between hook types and leader materials. Fish Res 131–133:9–14

ALDI (2016) ALDI US seafood buying policy. ALDI, Batavia

Amandè JM, Ariz J, Chassot E, Chavance P, Delgado de Molina A, Gaertner D, Muroa H, Pianet R, Ruiz J (2008) By-catch and discards of the European purse seine tuna fishery in the Indian Ocean. Estimation and characteristics for the 2003–2007 period. Ecosystem and By-Catch Working Group, Bangkok, Thailand, 2008. IOTC-2008-WPEB-12. Indian Ocean Tuna Commission

Amandè MJ, Ariz J, Chassot E, De Molina AD, Gaertner D, Murua H et al (2010) Bycatch of the European purse seine tuna fishery in the Atlantic Ocean for the 2003–2007 period. Aquat Living Resour 23:353–362

Andraka S, Mug M, Hall M et al (2013) Circle hooks: developing better fishing practices in the artisanal longline fisheries of the Eastern Pacific Ocean. Biol Conserv 160:214–223

Audzijonyte A, Fulton E, Haddon M, Helidoniotis F, Hobday A, Kuparinen A, Morrongiello J, Smith A, Upston J, Waples R (2016) Trends and management implications of human-influenced life-history changes in marine ectotherms. Fish Fish 17:1005–1028

Barlow J, Cameron G (2003) Field experiments show that acoustic pingers reduce marine mammal bycatch in the California drift gill net fishery. Mar Mammal Sci 19:265–283

Baum J, Myers R, Kehler D, Worm B, Harley S, Doherty P (2003) Collapse and conservation of shark populations in the Northwest Atlantic. Science 299:389–392

Bernal D, Sepulveda C, Musyl M, Brill R (2010) The eco-physiology of swimming and movement patterns of tunas, billfishes, and large pelagic sharks. In: Domenici P, Kapoor B (eds) Fish locomotion: an eco-ethological perspective. Science Publishers, Enfield, pp 436–483

Berrow S, Cosgrove R, Leeuney R, O'Brien J, McGraith D, Dalgard J, Le Gall Y (2009) Effect of acoustic deterrents on the behaviour of common dolphins. J Cetacean Res Manag 10:227–233

Beverly S (2009) Longline terminal gear identification guide. Secretariat of the Pacific Community, Noumea

Beverly S, Chapman L (2007) Interactions between sea turtles and pelagic longline fisheries. WCPFC-SC3-EB SWG/IP-

01. Western and Central Pacific Fisheries Commission, Kolonia, Federated States of Micronesia
Beverly S, Curran D, Musyl M, Molony B (2009) Effects of eliminating shallow hooks from tuna longline sets on target and non-target species in the Hawaii-based pelagic tuna fishery. Fish Res 96:281–288
Beverly S, Griffiths D, Lee R (2012) Anchored fish aggregating devices for artisanal fisheries in South and Southeast Asia: benefits and risks. RAP Publication 2012/20. FAO Regional Office for Asia and the Pacific, Bangkok
Bigelow K, Hampton J, Miyabe N (2002) Application of a habitat-based model to estimate effective longline fishing effort and relative abundance of Pacific bigeye tuna (*Thunnus obesus*). Fish Oceanogr 11:143–155
Bjordal A (2002) The use of technical measures in responsible fisheries: regulation of fishing gear. In Cochrane K (ed) A fishery manager's guidebook: management measures and their application. FAO Fisheries Technical Paper 424, pp 21–48. Food and Agriculture Organization of the United Nations, Rome
Bordino P, Kraus S, Albareda D, Fazio A, Palmerio A, Mendez M, Botta S (2002) Reducing incidental mortality of Franciscana dolphin *Pontoporia blainvillei* with acoustic warning devices attached to fishing nets. Mar Mammal Sci 18:833–842
Borenstein M, Hedges L, Higgins J, Rothstein H (2009) Introduction to meta-analysis. Wiley Press, West Sussex, p 421
Borucinska J, Kohler N, Natanson L, Skomal G (2002) Pathology associated with retained fishing hooks in blue sharks, *Prionace glauca* (L.), with implications for their conservation. J Fish Dis 25:515–521
Branch T, Lobo A, Purcell S (2013) Opportunistic exploitation: an overlooked pathway to extinction. Trends Ecol Evol 28:409–413
Broadhurst MK (2000) Modifications to reduce bycatch in prawn trawls: review and framework for development. Rev Fish Biol Fish 10:27–60
Bromhead D, Clarke S, Hoyle S, Muller B, Sharples P, Harley S (2012) Identification of factors influencing shark catch and mortality in the Marshall Islands tuna longline fishery and management implications. J Fish Biol 80:1870–1894
Camargo S, Coelho R, Chapman D et al (2016) Structure and genetic variability of the oceanic whitetip shark, *Carcharhinus longimanus*, determined using mitochondrial DNA. PLoS ONE 11:e0155623
Cambiè G, Muiño R, Mingozzi T, Freire J (2013) From surface to mid-water: Is the swordfish longline fishery "hitting rock bottom"? A case study in southern Italy. Fish Res 140:114–122
Caneco B, Donovan C, Harley S (2014) Analysis of WCPO longline observer data to determine factors impacting catchability and condition on retrieval of oceanic white-tip, silky, blue, and thresher sharks. WCPFCSC10- 2014/EB-WP-01. Western and Central Pacific Fisheries Commission, Kolonia, Federated States of Micronesia
Carey F, Scharold J, Kalmijn A (1990) Movements of blue sharks (*Prionace glauca*) in depth and course. Mar Biol 106:329–342
Carretta J, Barlow J (2011) Long-term effectiveness, failure rates, and "dinner bell" properties of acoustic pingers in a gillnet fishery. Mar Technol Soc J 45:7–19
Casale P, Heppell S (2016) How much sea turtle bycatch is too much? A stationary age distribution model for simulating population abundance and potential biological removal in the Mediterranean. Endanger Species Res 29:239–254
Casale P, Freggi D, Rocco M (2008) Mortality induced by drifting longline hooks and branchlines in loggerhead sea turtles, estimated through observation in captivity. Aquat Conserv 18:945–954
Castro J, Santiago J, Santana-Ortega A (2002) A general theory of fish aggregation to floating objects: an alternative to the meeting point hypothesis. Rev Fish Biol Fish 11:255–277
Chaloupka M (2003) Stochastic simulation modelling of loggerhead sea turtle population dynamics given exposure to competing mortality risks in the western south Pacific. In: Bolten AB, Witherington B (eds) Loggerhead sea turtles. Smithsonian Institution Press, Washington, pp 274–294
Chaloupka M, Parker D, Balazs G (2004) Modeling postrelease mortality of loggerhead sea turtles exposed to the Hawaii-based pelagic longline fishery. Mar Ecol Prog Ser 280:285–293
Christensen V, Walters C (2004) Ecopath with Ecosim: methods, capabilities and limitations. Ecol Model 172:109–139
Clarke S, Sato M, Small C, Sullivan B, Inoue Y, Ochi D (2014) Bycatch in longline fisheries for tuna and tuna-like species: a global review of status and mitigation measures. FAO Fisheries and Aquaculture Technical Paper 588. Food and Agriculture Organization of the United Nations, Rome
Coelho R, Fernandez-Carvalho J, Lino P, Santos M (2012) An overview of the hooking mortality of elasmobranchs caught in a swordfish pelagic longline fishery in the Atlantic Ocean. Aquat Living Resour 25:311–319
Coelho R, Santos M, Fernandez-Carvalho J, Amorim S (2015) Effects of hook and bait in a tropical northeast Atlantic pelagic longline fishery: part I—incidental sea turtle bycatch. Fish Res 164:302–311
Coll M, Shannon L, Lleisner K et al (2016) Ecological indicators to capture the effects of fishing on biodiversity and conservation status of marine ecosystems. Ecol Ind 60:947–962
Collette B, Carpenter K, Polidoro B, Juan-Jorda M, Boustany A, Die D, Elfes C, Fox W, Graves J, Harrison L et al (2011) High value and long life—double jeopardy for tunas and billfishes. Science 333:291–292
Collie J, Gislason H (2001) Biological reference points for fish stocks in a multispecies context. Can J Fish Aquat Sci 58:2167–2176
Cooke S, Suski C (2004) Are circular hooks and effective tool for conserving marine and freshwater recreational catch-and-release fisheries? Aquat Conserv 14:299–326
Cooke S, Barthel B, Suski C, Siepker M, Philipp D (2005) Influence of circle hook size on hooking efficiency, injury, and size selectivity of bluegill with comments on circle hook conservation benefits in recreational fisheries. N Am J Fish Manag 25:211–219
Cortes E, Domingo A, Miller P, Forselledo R, Mas F, Arocha F, Campana S et al (2015) Expanded ecological risk assessment of pelagic sharks caught in Atlantic pelagic longline fisheries. Collect Vol Sci Pap ICCAT, vol 71, pp 2637–2688
Croll D, Dewar H, Dulvy N, Fernando D, Francis M, Galván-Magaña F, Hall M, Heinrichs S, Marshall A, Mccauley D,

Newton K (2016) Vulnerabilities and fisheries impacts: the uncertain future of manta and devil rays. Aquat Conserv 26:562–575

Crowder L, Norse E (2008) Essential ecological insights for marine ecosystem-based management and marine spatial planning. Mar Policy 32:772–778

Curran D, Beverly S (2012) Effects of 16/0 circle hooks on pelagic fish catches in three South Pacific albacore longline fisheries. Bull Mar Sci 88:485–497

Curtis K, Moore J, Benson S (2015) Estimating limit reference points for western Pacific leatherback turtles (*Dermochelys coriacea*) in the U.S. West Coast EEZ. PLoS ONE 10:e0136452

Dagorn L, Holland K, Restrepo V, Moreno G (2013) Is it good or bad to fish with FADs? What are the real impacts of the use of drifting FADs on pelagic marine ecosystems? Fish Fish 14:391–415

Dagys M, Zydelis R (2002) Bird bycatch in fishing nets in Lithuanian coastal waters in wintering season 2001–2002. Acta Zool Litu 12:276–282

Dambacher J, Gaughan D, Marie-Joelle R, Rossignol P, Trenkel V (2009) Qualitative modelling and indicators of exploited ecosystems. Fish Fish 10:305–322

Dambacher J, Rothlisberg P, Loneragan N (2015) Qualitative mathematical models to support ecosystem-based management of Australia's northern prawn fishery. Ecol Appl 25:278–298

Darimont C, Carlson S, Kinnison M, Paquet P, Reimchen T, Wilmers C (2009) Human predators outpace other agents of trait change in the wild. PNAS 106:952–954

Dawson S, Northridge S, Waples D, Read A (2012) To ping or not to ping; the use of active acoustic devices in mitigating interactions between small cetaceans and gillnet fisheries. Endanger Species Res 19:201–221

Diamond S, Kleisner K, Duursma D, Wang Y (2010) Designing marine reserves to reduce bycatch of mobile species: a case study using juvenile red snapper (*Lutjanus campechanus*). Can J Fish Aquat Sci 67:1335–1349

Dillingham P, Fletcher D (2011) Potential biological removal of albatrosses and petrels with minimal demographic information. Biol Conserv 144:1885–1894

Dinmore T, Duplisea D, Rackham B, Maxwell D, Jennings S (2003) Impact of a large-scale area closure on patterns of fishing disturbance and the consequences for benthic communities. ICES J Mar Sci 60:371–380

Dulvy N, Ellis J, Goodwin N, Grant A, Reynolds J, Jennings S (2004) Methods of assessing extinction risk in marine fishes. Fish Fish 5:255–276

Dulvy N, Baum J, Clarke S, Compagno L, Cortes E, Domingo A, Fordham S, Fowler S, Francis M, Gibson C, Martinez J, Musick J, Soldo A, Stevens J, Valenti S (2008) You can swim but you can't hide: the global status and conservation of oceanic pelagic sharks and rays. Aquat Conserv 482:459–482

Dunlop E, Baskett M, Heino M, Dieckmann U (2009) Propensity of marine reserves to reduce the evolutionary effects of fishing in a migratory species. Evol Appl 2:371–393

Ehlers A, Worm B, Reusch T (2008) Importance of genetic diversity in eelgrass Zostera marina for its resilience to global warming. Mar Ecol Prog Ser 355:1–7

Elmqvist T, Folke C, Nystrom M, Peterson G, Bengtsson J, Walker B, Norberg J (2003) Response diversity, ecosystem change, and resilience. Front Ecol Environ 1:488–494

EPA (2015) Wetlands compensatory mitigation. U.S. Environmental Protection Agency, Washington

Epperly S, Watson J, Foster D, Shah A (2012) Anatomical hooking location and condition of animals captured with pelagic longlines: the grand banks experiments 2002–2003. Bull Mar Sci 88:513–527

Erzini K, Goncalves J, Bentes L, Lino P (1997) Fish mouth dimensions and size selectivity in a Portuguese longline fishery. J Appl Ichthyol 13:41–44

Escalle L, Brouwer S, Phillips J, Pilling G, PNA (2017) Preliminary analyses of PNA FAD tracking data from 2016 and 2017. WCPFC-SC13-2017/MI-WP-05. Federated States of Micronesia, Western and Central Pacific Fisheries Commission, Kolonia

Evans S, Sheldon B (2008) Interspecific patterns of genetic diversity in birds: correlations with extinction risk. Conserv Biol 22:1016–1025

Faith D (2008) Threatened species and the potential loss of phylogenetic diversity: Conservation scenarios based on estimated extinction probabilities and phylogenetic risk analysis. Conserv Biol 22:1461–1470

FAO (1995) Code of conduct for responsible fisheries. Food and Agriculture Organization of the United Nations, Rome

FAO (1999a) International plan of action for reducing incidental catch of seabirds in longline fisheries. Food and Agriculture Organization of the United Nations, Rome

FAO (1999b) International plan of action for the conservation and management of sharks. Food and Agriculture Organization of the United Nations, Rome

FAO (2003) Fisheries management 2. The ecosystem approach to fisheries. FAO Technical Guidelines for Responsible Fisheries No. 4 (Suppl. 2). Food and Agriculture Organization of the United Nations, Rome

FAO (2010) Guidelines to reduce sea turtle mortality in fishing operations. In: Gilman E, Bianchi G (eds) FAO technical guidelines for responsible fisheries. Food and Agriculture Organization of the United Nations, Rome. ISBN 978-92-106226-5

FAO (2011) International guidelines on bycatch management and reduction of discards. Food and Agriculture Organization of the United Nations, Rome

Ferretti F, Worm B, Britten G, Heithaus M, Lotze H (2010) Patterns and ecosystem consequences of shark declines in the ocean. Ecol Lett 13:1055–1071

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

Fonteneau A, Chassot E, Bodin N (2013) Global spatio-temporal patterns in tropical tuna purse seine fisheries on drifting fish aggregating devices (DFADs): taking a historical perspective to inform current challenges. Aquat Living Resour 26:37–48

Forney K, Kobayashi D, Johnston D, Marchetti J, Marsik M (2011) What's the catch? Patterns of cetacean bycatch and depredation in Hawaii-based pelagic longline fisheries. Mar Ecol 32:380–391

Foster D, Epperly S, Shah A, Watson J (2012) Evaluation of hook and bait type on the catch rates in the western north Atlantic ocean pelagic longline fishery. Bull Mar Sci 88:529–545

Fu D, Rouz M, Clarke S, Francis M, Dunn A, Hoyle S (2016) Pacific-wide sustainability risk assessment of bigeye thresher shark (*Alopias superciliosus*). National Institute of Water and Atmospheric Research, Wellington

Fujimori Y, Tokai T (2001) Estimation of gillnet selectivity curve by maximum likelihood method. Fish Sci 67:644–654

Fulton E, Smith A (2004) Lessons learnt from a comparison of three ecosystem models for Port Philip Bay, Australia. Afr J Mar Sci 26:219–243

Fulton E, Fuller M, Smith A, Punt A (2004) Ecological indicators of the ecosystem effects of fishing: final report. Australian Fisheries Management Authority Report R99/1546. Australian Fisheries Management Authority, Canberra

Fulton E, Smith A, Punt A (2005) Which ecological indicators can robustly detect effects of fishing? ICES J Mar Sci 62:540–551

Fulton E, Smith A, Smith D, Johnson P (2014) An integrated approach is needed for ecosystem based fisheries management: insights from ecosystem-level management strategy evaluation. PLoS ONE 9:e84242

Gabr M, Fujimori Y, Shimizu S, Miura T (2007) Trawling experiment in a circular water tank to assess the effects of towing speed, light intensity, and mesh shape on active escape of undersized fish. Fish Sci 73:557–564

Gaertner D, Ariz J, Bez N, Clermidy S, Moreno G, Murua H, Soto M, Marsac F (2016) Results achieved within the framework of the EU research project: catch, effort, and ecosystem impacts of FAD-fishing (CECOFAD). IOTC-2016-WPTT18-35. Mahé, Seychelles, Indian Ocean Tuna Commission

Galeana-Villasenor I, Galvan-Magana F, Gomez-Aguilar R (2008) Influence of hook type and fishing depth on longline catches of sharks and other pelagic species in the northwest Mexican Pacific. Rev Biol Mar Oceanog 43:99–110

Gallagher A, Orbesen E, Hammerschlag N, Serafy J (2014) Vulnerability of oceanic sharks as pelagic longline bycatch. Glob Ecol Conserv 1:50–59

Garcia S, Zerbi A, Aliaume C, Do Chi T, Lasserre G (2003) The ecosystem approach to fisheries. FAO Fisheries Technical Paper No. 443. Food and Agriculture Organization of the United Nations, Rome

Garcia S, Kolding J, Rice J et al (2012) Reconsidering the consequences of selective fisheries. Science 335:1045–1047

García-Párraga D, Crespo-Picazo J, Bernaldo de Quirós Y, Cervera V, Martí-Bonmati L, Díaz-Delgado J, Arbelo M, Moore M, Jepson P, Fernández A (2014) Decompression sickness ('the bends') in sea turtles. Dis Aquat Organ 111:191–205

Giant Eagle (2016) Giant eagle tuna procurement policy. Giant Eagle, Indianapolis

Gilman E (2011) Bycatch governance and best practice mitigation technology in global tuna fisheries. Mar Policy 35:590–609

Gilman E, Hall M (2015) Potentially significant variables explaining bycatch and survival rates and alternative data collection protocols to harmonize tuna RFMOs' Pelagic Longline Observer Programmes. Appendix 1 to WCPFC-SC11-2015/EB-IP-05. Western and Central Pacific Fisheries Commission, Kolonia, Pohnpei, Federated States of Micronesia

Gilman E, Huang H (2017) Review of effects of pelagic longline hook and bait type on sea turtle catch rate, anatomical hooking position and at-vessel mortality rate. Rev Fish Biol Fish 27:43–52

Gilman E, Brothers N, Kobayashi D (2005) Principles and approaches to abate seabird bycatch in longline fisheries. Fish Fish 6:35–49

Gilman E, Brothers N, McPherson G, Dalzell P (2006a) Review of cetacean interactions with longline gear. J Cetacean Res Manag 8:215–223

Gilman E, Dalzell P, Martin S (2006b) Fleet communication to abate fisheries bycatch. Mar Policy 30:360–366

Gilman E, Zollett E, Beverly S, Nakano H, Shiode D, Davis K, Dalzell P, Kinan I (2006c) Reducing sea turtle bycatch in pelagic longline gear. Fish Fish 7:2–23

Gilman E, Gearhart J, Price B, Eckert S, Milliken J, Wang J, Swimmer Y, Shiode D, Abe O, Peckham S, Chaloupka M, Hall M, Mangel J, Alfaro-Shigueto J, Dalzell P, Ishizaki A (2010) Mitigating sea turtle bycatch in coastal passive net fisheries. Fish Fish 11:57–88

Gilman E, Dunn D, Read A, Hyrenbach KD, Warner R (2011) Designing criteria suites to identify discrete and networked sites of high value across manifestations of biodiversity. Biodivers Conserv 20(14):3363–3383

Gilman E, Suuronen P, Hall M, Kennelly S (2013) Causes and methods to estimate cryptic sources of fishing mortality. J Fish Biol 83:766–803

Gilman E, Owens M, Kraft T (2014a) Ecological risk assessment of the Marshall Islands longline tuna fishery. Mar Policy 44:239–255

Gilman E, Passfield K, Nakamura K (2014b) Performance of regional fisheries management organizations: ecosystem-based governance of bycatch and discards. Fish Fish 15:327–351

Gilman E, Allain V, Collette B, Hampton J, Lehodey P (2016a) Climate change effects on principal market tunas, a review. In: Laffoley D, Baxter J (eds) Explaining ocean warming: causes, scale, effects and consequences. IUCN, Gland, pp 254–270. ISBN 978-8317-1806-4

Gilman E, Chaloupka M, Swimmer Y, Piovano S (2016b) A cross-taxa assessment of pelagic longline bycatch mitigation measures: conflicts and mutual benefits to elasmobranchs. Fish Fish 17:748–784

Gilman E, Musyl M, Itano D, Heberer C, Zimring M, Williams P (2017a) Effects of pelagic longline time-of-day and gear soak depth on species- and size-selectivity. The Nature Conservancy, San Francisco. https://tinyurl.com/PLL-time-depth. Accessed 1 Oct 2017

Gilman E, Suuronen P, Chaloupka M (2017b) Discards by global tuna fisheries. Mar Ecol Prog Ser 582:231–252

Gilman E, Weijerman M, Suuronen P (2017c) Ecological data from observer programs underpin ecosystem-based fisheries management. ICES J Mar Sci 74:1481–1495

Gilman E, Bigler B, Muller B, Moreno G, Largacha E, Hall M, Poisson F, Toole J, He P, Chiang W (2018a) Stakeholder views on methods to identify the ownership and track the position of drifting fish aggregating devices used by tuna purse seine fisheries with reference to the FAO Draft Guidelines on the Marking of Fishing Gear. FAO Fisheries Circular 1163. Food and Agriculture Organization of the United Nations, Rome

Gilman E, Chaloupka M, Musyl M (2018b) Effects of pelagic longline hook size on species- and size-selectivity and survival. Rev Fish Biol Fish 28:417–433

Godin A, Carlson J, Burgener V (2012) The effect of circle hooks on shark catchability and at-vessel mortality rates in longlines fisheries. Bull Mar Sci 88:469–483

Goni R (1998) Ecosystem effects of marine fisheries: an overview. Ocean Coast Manag 40:37–64

Gray C, Kennelly S (2018) Bycatch of endangered, threatened and protected species in marine fisheries. Rev Fish Biol Fish 28:521. https://doi.org/10.1007/s11160-018-9520-7

Hall M (1996) On bycatches. Rev Fish Biol Fish 6:319–352

Hall M (1998) An ecological view of the tuna-dolphin problem: impacts and trade-offs. Rev Fish Biol Fish 8:1–34

Hall S (2002) The use of technical measures in responsible fisheries: area and time restrictions. In Cochrane K (ed) A fishery manager's guidebook: management measures and their application, pp 49–74. FAO Fisheries Technical Paper 424. Food and Agriculture Organization of the United Nations, Rome

Hall M (2015) More on bycatches: changes, evolution, and revolution. In: Kruse G, An H, Dicosimo J, Eishens C, Gislason G, McBride D, Rose C, Siddon C (eds) Fisheries bycatch: global issues and creative solutions. University of Alaska Fairbanks, Alaska Sea Grant

Hall M, Roman M (2013) Bycatch and non-tuna catch in the tropical tuna purse seine fisheries of the world. FAO Fisheries and Aquaculture Technical Paper No. 568. Food and Agriculture Organization of the United Nations, Rome

Hall M, Alverson D, Metuzal K (2000) By-catch: problems and solutions. Mar Pollut Bull 41:204–219

Hall M, Campa M, Gomez M (2001) Solving the tuna-dolphin problem in the eastern Pacific purse-seine fishery. In: Borgese E, Chircop A, McConnell M (eds) Ocean Yearbook, vol 17. University of Chicago Press, Chicago, pp 60–92

Hall M, Swimmer Y, Parga M (2012) No "silver bullets" but plenty of options. Working with artisanal fishers in the Eastern Pacific to reduce incidental sea turtle mortality in longline fisheries. In: Seminoff J, Wallace B (eds) Sea turtles of the Eastern Pacific: advances in research and conservation. The University of Arizona Press, Tucson, pp 136–153. ISBN 978-0-8165-1158-7

Hall M, Gilman E, Minami H, Mituhasi T, Carruthers E (2017) Mitigating bycatch in tuna fisheries. Rev Fish Biol Fish 27:881–908

Hallier J, Gaertner D (2008) Drifting fish aggregation devices could act as an ecological trap for tropical tuna species. Mar Ecol Prog Ser 353:255–264

Hamer D, Ward T, Shaughnessy P, Clark S (2011) Assessing the effectiveness of the Great Australian Bight Marine Park in protecting the endangered Australian sea lion *Neophoca cinerea* from bycatch mortality in shark gillnets. Endanger Species Res 14:203–216

Hamley J (1975) Review of gillnet selectivity. J Fish Board Can 32:1943–1969

Harley S, Tremblay-Boyer L, Williams P, Pilling G, Hampton J (2015) Examination of purse seine catches of bigeye tuna. WCPFC-SC11-2015/MI-WP-07. Kolonia, Federated States of Micronesia, Western and Central Pacific Fisheries Commission

Hauser L, Adcock G, Smith P, Ramírez J, Carvalho G (2002) Loss of microsatellite diversity and low effective population size in an overexploited population of New Zealand snapper (*Pagrus auratus*). PNAS 99:11742–11747

Hayase S, Yatsu A (1993) Preliminary report of a squid subsurface driftnet experiment in the North Pacific during 1991. INPFC Bull 53:557–576

Hays G, Hobson V, Metcalfe J, Righton D, Sims D (2006) Flexible foraging movements of leatherback turtles across the North Atlantic Ocean. Ecology 87:2647–2656

He P, Pol M (2010) Fish behavior near gillnets: capture processes and influencing factors. In: He P (ed) Behavior of marine fishes. Capture processes and conservation challenges. Wiley-Blackwell, Ames, pp 183–203. ISBN 978-0-8138-1536-7

Heino M, Pauli B, Dieckmann U (2015) Fisheries-induced evolution. Annu Rev Ecol Evol Syst 46:461–480

Hilborn R, Walters C (1992) Quantitative fisheries stock assessment: choice, dynamics and uncertainty. Chapman and Hall, London

Hobday A, Smith A, Webb H, Daley R, Wayte S, Bulman C, Dowdney J et al (2007) Ecological risk assessment for the effects of fishing: methodology. Report R04/1072. Australian Fisheries Management Authority, Canberra

Hobday A, Hartog J, Timmiss T, Fielding J (2010) Dynamic spatial zoning to manage southern bluefin tuna (*Thunnus maccoyii*) capture in a multi-species longline fishery. Fish Oceanogr 19:243–253

Hobday A, Smith A, Stobutzki I, Bullman C, Daley R, Dambacher J, Deng R et al (2011) Ecological risk assessment for the effects of fishing. Fish Res 108:372–384

Holgård H, Lassen H (2002) Manual on estimation of selectivity for gillnet and longline gears in abundance surveys. FAO Fisheries Technical Paper 397. Food and Agriculture Organization of the United Nations, Rome. ISBN 92-5-104461-9

Hollins J, Thambithurai D, Köeck B, Crespel A, Bailey D, Cooke S, Lindström J, Parsons K, Killen S (2018) A physiological perspective on fisheries-induced evolution. Evol Appl 11:561–576. https://doi.org/10.1111/eva.12597

Hutchinson W, van Oosterhout C, Rogers S, Carvalho G (2003) Temporal analysis of archived samples indicates marked genetic changes in declining North Sea cod (*Gadus morhua*). Proc R Soc Lond B Biol Sci 270:2125–2132

IATTC (2011) Hooks used in artisanal longline fisheries of the Eastern Pacific Ocean. Inter-American Tropical Tuna Commission, La Jolla

ICCAT (2017) Report of the Standing Committee on Research and Statistics (SCRS), Madrid, Spain, 2 to 6 October 2017. International Commission for the Conservation of Atlantic Tunas, Madrid

ICES (2001) Report of the working group on ecosystem effects of fishing activities. CM/ACME: 09. International Council for the Exploration of the Seas, Copenhagen

Isaac N, Turvey S, Collen B, Waterman C, Baillie J (2007) Mammals on the EDGE: conservation priorities based on threat and phylogeny. PLoS ONE 2:e296

ISC (2015) Stock assessment update for striped marlin (*Kajikia audax*) in the Western and Central North Pacific Ocean through 2013. Billfish Working Group. International Scientific Committee for Tuna and Tuna-like species in the North Pacific Ocean

ISSF (2015) ISSF guide for non-entangling FADs. International Seafood Sustainability Foundation, Washington

ISSF (2017a) RFMO bycatch mitigation and monitoring. In: ISSF. Status of the World Fisheries for Tuna. ISSF Technical Report 2017-02, pp 94–96. International Seafood Sustainability Foundation, Washington

ISSF (2017b) Status of the world fisheries for tuna. ISSF Technical Report 2017-02. International Seafood Sustainability Foundation, Washington

ISSF (2017c) A summary of bycatch issues and ISSF mitigation activities to date in purse seine fisheries, with emphasis on FADs. ISSF Technical Report 2017-06. International Seafood Sustainability Foundation, Washington

IUCN (2017) The IUCN red list of threatened species. Version 2017-2. International Union for the Conservation of Nature, Gland. www.iucnredlist.org. Accessed 18 Oct 2017

Jorgensen C, Enberg K, Dunlop E, Arlinghaus R, Boukal D, Brander K, Ernande B, Gardmark A, Johnston F, Matsumura S, Pardoe H (2007) Ecology-managing evolving fish stocks. Science 318:1247–1248

Kareiva P, Marvier M (2003) Conserving biodiversity coldspots. Am Sci 91:344–351

Kenchington E, Heino M, Nielsen E (2003) Managing marine genetic diversity: time for action? ICES J Mar Sci 60:1172–1176

Kerstetter D, Graves J (2006) Effects of circle versus J-style hooks on target and non-target species in a pelagic longline fishery. Fish Res 80:239–250

Kirkwood G, Walker T (1986) Gill net mesh selectivities for gummy shark, *Mustelus antarcticus* Günther, taken in south-eastern Australian waters. Aust J Mar Fresh Res 37:689–697

Kitchell J, Essington T, Boggs C, Schindler D, Walters C (2002) The role of sharks and longline fisheries in a pelagic ecosystem of the central Pacific. Ecosystems 5:202–216

Koehler H, Moreno G (2016) Survey on the treatment of support/supply/tender vessels that service drifting FADs in tuna RFMOs. ISSF Technical Report 2016-11. International Seafood Sustainability Foundation, Washington

Koschinski S, Culik B, Trippel E, Ginzkey L (2006) Behavioral reactions of free—ranging harbour porpoises *Phocoena phocoena* encountering standard nylon and $BaSO_4$ mesh gillnets and warning sound. Mar Ecol Prog Ser 313:285–294

Kraus S, Read A, Anderson E, Baldwin K, Solow A, Spradlin T, Williamson J (1997) Acoustic alarms reduce porpoise mortality. Nature 388:525

Kuparinen A, Merila J (2007) Detecting and managing fisheries-induced evolution. Trends Ecol Evol 22:652–659

Larsen F, Eigaard O, Tougaard J (2007) Reduction of harbour porpoise (*Phocoena phocoena*) bycatch by iron-oxide gillnets. Fish Res 85:270–278

Leadley P, Pereira H, Alkemade R, Fernandez-Manjarres J, Proenca V, Scharlemann J, Walpole M (eds) (2010) Biodiversity scenarios: projections of 21st century change in biodiversity and associated ecosystem services. A Technical Report for the Global Biodiversity Outlook 3. CBD Technical Series No. 50. Convention on Biological Diversity Secretariat, Montreal. ISBN 92-9225-219-4

Leeney R, Berrow S, McGrath D, O'Brien J, Cosgrove R, Godley B (2007) Effects of pingers on the behaviour of bottlenose dolphins. J Mar Biol Assoc UK 87:129–133

Lennox R, Alos J, Arlinghaus R, Horodysky A, Klefoth T, Monk C, Cooke S (2017) What makes a fish vulnerable to capture by hooks? A conceptual framework and a review of key determinants. Fish Fish 18:986–1010. https://doi.org/10.1111/faf.12219

Lewison R, Crowder L (2003) Estimating fishery bycatch and effects on a vulnerable seabird population. Ecol Appl 13:743–753

Lewison R, Crowder L (2007) Putting longline bycatch of sea turtles into perspective. Conserv Biol 21:79–86

Lezama-Ochoa N, Murua H, Hall M, Roman M, Ruiz J, Vogel N, Caballero A, Sancristobal I (2017) Biodiversity and habitat characteristics of the bycatch assemblages in fish aggregating devices (FADs) and school sets in the eastern Pacific Ocean. Front Mar Sci 4:265. https://doi.org/10.3389/fmars.2017.00265

Li Y, Browder J, Jiao Y (2012) Hook effects on seabird bycatch in the United States Atlantic pelagic longline fishery. Bull Mar Sci 88:559–569

Lien J, Barney W, Todd S, Seton R, Guzzwell J (1992) Effects of adding sounds to cod traps on the probability of collisions by humpback whales. In: Thomas J, Kastelein R, Supin A (eds) Marine mammal sensory systems. Springer, Boston

Link J (2005) Translating ecosystem indicators into decision criteria. ICES J Mar Sci 62:569–576

Link J, Browman H (2014) Integrating what? Levels of marine ecosystem-based assessment and management. ICES J Mar Sci 71:1170–1173

Little A, Needle C, Hilborn R, Holland D, Marshall C (2015) Real-time spatial management approaches to reduce bycatch and discards: experiences from Europe and the United States. Fish Fish 16:576–602

Lokkeborg S, Bjordal A (1992) Species and size selectivity in longline fishing: a review. Fish Res 13:311–322

Lopez J, Moreno G, Boyra G, Dagorn L (2016) A model based on data from echo-sounder buoys to estimate biomass of fish species associated with fish aggregating devices. Fish Bull 114:166–179

Mace P (1994) Relationships between common biological reference points used as thresholds and targets of fisheries management strategies. Can J Fish Aquat Sci 51:110–122

Mangel J, Alfaro-Shigueto J, Swimmer Y, Wallace G (2014) Tests of visual cues and sub-surface nets as bycatch mitigation measures in small-scale gillnet fisheries in Peru. In: ACAP. Sixth meeting of the seabird bycatch working group of ACAP, pp 10–12. Agreement on the Conservation of Albatrosses and Petrels, Hobart, Australia

Marsac F, Fonteneau A, Menard F (2000) Drifting FADs used in tuna fisheries: an ecological trap? In: Le Gall J, Cayre P, Taquet M (eds) Pêche thonière et dispositifs de concentration de poissons, vol 28. Actes Collogues-IFREMER, Sete, pp 15–35

Maufroy A, Chassot E, Joo R, Kaplan D (2015) Large-scale examination of spatio-temporal patterns of drifting fish aggregating devices (dFADs) from tropical tuna fisheries of the Indian and Atlantic oceans. PLoS ONE 10:e0128023

McNeely J, Miller K, Reid W, Mittermeier R, Werner T (1990) Conserving the World's Biological Diversity. World Conservation Union, World Resources Institute, Conservation International, World Wildlife Fund—US, and the World Bank, Washington

Mejuto J, Garcia-Cortes B, Ramos-Cartelle A (2008) Trials using different hook and bait types in the configuration of the surface longline gear used by the Spanish swordfish (*Xiphias gladius*) fishery in the Atlantic Ocean. Collect Vol Sci Pap ICCAT, vol 62, pp 1793–1830

Melvin E, Parrish J, Conquest L (2001a) Novel tools to reduce seabird bycatch in coastal gillnet fisheries. In: Melvin E, Parrish J (eds) Seabird bycatch: trends, roadblocks, and solutions. AK-SG-01-01i. University of Alaska Sea Grant, Fairbanks, pp 161–184

Melvin E, Parrish J, Dietrich K, Hamel O (2001b) Solutions to seabird bycatch in Alaska's demersal longline fisheries. Washington Sea Grant Program, University of Washington, Seattle

Monterey Bay Aquarium (2015) Seafood watch criteria for fisheries. Seafood Watch Program, Monterey Bay Aquarium, Monterey

Monterey Bay Aquarium (2018) About tuna. Seafood Watch Program, Monterey Bay Aquarium, Monterey. https://www.seafoodwatch.org/seafood-recommendations/groups/tuna/overview. Accessed 20 Feb 2018

Moore M, Bogomolni A, Dennison S, Early G, Garner M, Hayward B, Lentell B, Rotstein D (2009) Gas bubbles in seals, dolphins, and porpoises entangled and drowned at depth in gillnets. Vet Pathol 46:536–547

Morato T, Varkey D, Damaso C, Machete M, Santos M, Prieto R, Santos R, Pitcher T (2008) Evidence of a seamount effect on aggregating visitors. Mar Ecol Prog Ser 357:23–32

Moreno G, Restrepo V, Dagorn L, Hall M, Murua J, Sancristobal I, Grande M, Le Couls S, Santiago J (2016) Workshop on the use of biodegradable Fish Aggregating Devices (FAD). ISSF Technical Report 2016-18A. International Seafood Sustainability Foundation, Washington

Morizur Y, Le Gall Y, Van Canneyt O, Gamblin C (2008) Tests d'efficacité du repulsive acoustique CETASAVER à bord des chalutiers commerciaux français—(Effectiveness of acoustic deterrent CETASAVER testing on board French commercial trawlers). IFREMER. Centre de Brest, Sciences et Technologie Halieutiques

Moyes C, Fragoso N, Brill R, Musyl M (2006) Predicting postrelease survival in large pelagic fish. Trans Am Fish Soc 135:1389–1397

Muhling B, Lee S, Lamkin J, Liu Y (2011) Predicting the effects of climate change on bluefin tuna (*Thunnus thynnus*) spawning habitat in the Gulf of Mexico. ICES J Mar Sci 68:1051–1062

Murphy B, Willis D (1996) Fisheries techniques. American Fisheries Society, Bethesda

Murray K (2009) Characteristics and magnitude of sea turtle bycatch in US mid-Atlantic gillnet gear. Endanger Species Res 8:211–224

Murray K, Read A, Solow A (2000) The use of time/area closures to reduce bycatches of harbour porpoises: lessons from the Gulf of Maine sink gillnet fishery. J Cetacean Res Manag 2:135–141

Musick J (ed) (1999) Life in the slow lane: ecology and conservation of long-lived marine animals. Symposium 23. American Fisheries Society, Bethesda

Musyl M, Brill R, Boggs C, Curran D, Kazama T, Seki M (2003) Vertical movements of bigeye tuna (*Thunnus obesus*) associated with islands, buoys, and seamounts near the main Hawaiian Islands from archival tagging data. Fish Oceanogr 12:152–169

Musyl M, Brill R, Curran D, Fragoso N, McNaughton L, Nielsen A, Kikkawa B, Moyes C (2011) Postrelease survival, vertical and horizontal movements, and thermal habitats of five species of pelagic sharks in the central Pacific Ocean. Fish Bull 109:341–361

Musyl M, Moyes C, Brill R, Mourato B, West A, McNaughton L, Chiang W, Sun C (2015) Postrelease mortality in istiophorid billfish. Can J Fish Aquat Sci 72:1–19

Nakano H, Okazaki M, Okamoto H (1997) Analysis of catch depth by species for tuna longline fishery based on catch by branchlines. Bull Natl Res Inst Far Seas Fish 24:43–62

Nakano H, Matsunaga H, Okamoto H, Okazaki M (2003) Acoustic tracking of bigeye thresher shark *Alopias superciliosus* in the eastern Pacific Ocean. Mar Ecol Prog Ser 265:255–261

NMFS (2001) Atlantic highly migratory species; pelagic longline fishery; sea turtle protection measures. National Marine Fisheries Service. Fed Regist 66:36711–36714

NMFS (2004) Fisheries off West Coast States and in the Western Pacific; Western Pacific Pelagic Fisheries; Pelagic longline fishing restrictions, seasonal area closure, limit on swordfish fishing effort, gear restrictions, and other sea turtle take mitigation measures. Fed Reg 69:17329–17354

NMFS (2005) Fisheries off west coast states and in the western Pacific; pelagic fisheries; additional measures to reduce the incidental catch of seabirds in the Hawaii pelagic longline fishery. Fed Reg 70:75075–75080

NMFS (2014) International fisheries; western and central Pacific fisheries for highly migratory species; fishing restrictions regarding the oceanic whitetip shark, the whale shark, and the silky Shark. Fed Reg 80:8807–8816

NMFS (2015) Fisheries off west coast states and in the western Pacific; western Pacific pelagic fisheries; sea turtle mitigation measures. Fed Reg 70:69282–69285

NMFS (2016) Magnuson-Stevens Fishery Conservation and Management Act Provisions; implementation of the Shark Conservation Act of 2010. Fed Reg 81:42285–42289

NMFS (2012) Taking of marine mammals incidental to commercial fishing operations; false killer whale take reduction plan. Fed Reg 77:71260–71286

Northridge S, Vernicos D, Raitsos-Exarchopolous D (2003) Net depredation by bottlenose dolphins in the Aegean: first attempts to quantify and to minimise the problem. IWC SC/55/SM25, International Whaling Commission, Cambridge

Oliver S, Braccini M, Newman S, Harvey E (2015) Global patterns in the bycatch of sharks and rays. Mar Pollut Bull 54:86–97

Orphanides C (2010) Protected species bycatch estimating approaches: estimating harbor porpoise bycatch in U.S. northwestern Atlantic gillnet fisheries. J Northwest Atl Fish Sci 42:55–76

Pace M, Cole J, Carpenter S, Kitchell J (1999) Trophic cascades revealed in diverse ecosystems. Trends Ecol Evol 14:483–488

Pacheco J, Kerstetter D, Hazin F, Hazin H, Segundo R, Graves J, Carvalho F, Travassos P (2011) A comparison of circle hook and J hook performance in a western equatorial Atlantic Ocean pelagic longline fishery. Fish Res 107:39–45

Palka D, Rossman M, Vanatten A, Orphanides C (2008) Effect of pingers on harbour porpoise (*Phocoena phocoena*) bycatch in the US Northeast gillnet fishery. J Cetacean Res Manag 10:217–226

Parga M, Pons M, Andraka S, Rendon L, Mituhasi T, Hall M, Pacheco L, Segura A, Osmond M, Vogel N (2015) Hooking locations in sea turtles incidentally captured by artisanal longline fisheries in the Eastern Pacific Ocean. Fish Res 164:231–237

Patrick W, Spencer P, Ormseth O, Cope J, Field J, Kobayashi D, Gedamke T, Cortes E, Bigelow K, Overholtz W, Link J, Lawson P (2009) Use of productivity and susceptibility indices to determine stock vulnerability, with example applications to six US fisheries. NOAA Technical Memorandum NMFS-F/SPO-101. National Marine Fisheries Service, Alaska Fisheries Science Center, Seattle

Pavlov D, Kasumyan A (2000) Patterns and mechanisms of schooling behavior in fish: a review. J Ichthyol 40:S163–S231

Peatman T, Allain V, Caillot S, Wiliams P, Smith N (2017) Summary of purse seine fishery bycatch at a regional scale, 2003–2016. WCPFC-SC13-2017/ST-WP-05. Western and Central Pacific Fisheries Commission, Kolonia, Federated States of Micronesia

Pikitch E, Santora C, Babcock E, Bakun A, Bonfil R, Conover D, Dayton P et al (2004) Ecosystem-based fishery management. Science 305:346–347

Pitcher T (1995) The impact of pelagic fish behavior on fisheries. Sci Mar 59:295–306

Pitcher T (2001) Fisheries managed to rebuild ecosystems? Reconstructing the past to salvage the future. Ecol Appl 11:601–617

Pitcher T, Ainsworth C (2010) Resilience to change in two coastal communities: using the maximum dexterity fleet. Mar Policy 34:810–814

Pitcher T, Lam M (2010) Fishful thinking: rhetoric, reality and the sea before us. Ecol Soc 15:12. http://www.ecologyandsociety.org/vol15/iss2/art12/

Pitcher T, Kalikoski D, Short K, Varkey D, Pramod G (2009) An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries. Mar Policy 33:223–232

Plaganyi E, Punt A, Hillary R, Morello E, Thebaud O, Hutton T, Pillans R et al (2014) Multispecies fisheries management and conservation: tactical applications using models of intermediate complexity. Fish Fish 15:1–22

Polovina J (1984) Model of a coral reef ecosystems. I. The ECOPATH model and its application to French Frigate Shoals. Coral Reefs 3:1–11

Polovina J, Balazs G, Howell E, Parker D (2003) Dive-depth distribution of loggerhead (*Caretta caretta*) and olive ridley (*Lepidochelys olivacea*) sea turtles in the central North Pacific: might deep longline sets catch fewer turtles? Fish Bull 101:189–193

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

Powers J, Abeare S (2009) Fishing effort redistribution in response to area closures. Fish Res 99:216–225

Price B, Van Salisbury C (2007) Low-profile gillnet testing in the deep water region of Pamlico sound, NC. North Carolina Department of Environment and Natural Resources, Division of Marine Fisheries, Morehead City

Redding D, Moores A (2006) Incorporating evolutionary measures into conservation prioritization. Conserv Biol 20:1670–1678

Reed D, Frankham R (2003) Correlation between fitness and genetic diversity. Conserv Biol 17:230–237

Reinhardt J, Weaver J, Latham P et al (2017) Catch rate and at-vessel mortality of circle hooks versus J-hooks in pelagic longline fisheries: a global meta-analysis. Fish Fish 19:413–430. https://doi.org/10.1111/faf.12260

Restrepo V, Dagorn L, Justel-Rubio A (2017) Questions and answers about FADs and bycatch. Version 2. ISSF Technical Report 2017-04. International Seafood Sustainability Foundation, Washington

Rey J, Muñoz-Chápuli R (1992) Intra and interspecific association of large pelagic fishes inferred from catch data of surface longline. Environ Biol Fish 35:95–103

Rice J (2000) Evaluating fishery impacts using metrics of community structure. ICES J Mar Sci 57:682–688

Rice M, Balazs G (2008) Diving behavior of the Hawaiian green turtle during oceanic migrations. J Exp Mar Biol Ecol 356:121–127

Richard Y, Abraham E (2013) Application of potential biological removal methods to seabird populations. New Zealand Aquatic Environment and Biodiversity Report No. 108. Ministry for Primary Industries, Wellington

Rijnsdorp A, Piet G, Poos J (2001) Effort allocation of the Dutch beam trawl fleet in response to a temporarily closed area in the North Sea. In: ICES. ICES. International Council for the Exploration of the Sea Meeting 2001/N:01, pp 17–25. International Council for the Exploration of the Sea, Copenhagen

Rochet M, Trenkel V (2003) Which community indicators can measure the impact of fishing? A review and proposals. Can J Fish Aquat Sci 60:86–99

Saccheri I, Kuussaari M, Kankare M, Vikman P, Fortelius W, Hanski I (1998) Inbreeding and extinction in a butterfly metapopulation. Nature 392:491–494

Sainsbury K, Punt A, Smith A (2000) Design of operational management strategies for achieving fishery ecosystem objectives. ICES J Mar Sci 57:731–741

Salerno D, Eayrs S, Pol M, Lee S, Baukus A (2010) Analysis of size selectivity and bycatch in the gillnet fishery for monkfish. Gulf of Maine Research Institute, Portland

Sales G, Giffoni B, Fiedler F, Azevedo V, Kotas J, Swimmer Y, Bugoni L (2010) Circle hook effectiveness for the mitigation of sea turtle bycatch and capture of target species in a Brazilian pelagic longline fishery. Aquat Conserv 20:428–436

Santos M, Coelho R, Fernandez-Carvalho J, Amorim S (2012) Effects of hook and bait on sea turtle catches in an equatorial Atlantic pelagic longline fishery. Bull Mar Sci 88:683–701

Santos M, Coelho R, Fernandez-Carvalho J, Amorim S (2013) Effects of 17/0 circle hooks and bait on sea turtles bycatch in a Southern Atlantic swordfish longline fishery. Aquat Conserv 23:732–744

Schaefer K, Fuller D (2002) Movement, behavior, and habitat selection of bigeye tuna (*Thunnus obesus*) in the eastern equatorial Pacific, ascertained through archival tags. Fish Bull 100:765–788

Sedberry G, Loefer J (2001) Satellite telemetry tracking of swordfish, *Ziphias gladius*, off the eastern United States. Mar Biol 139:355–360

Sempo G, Dagorn L, Robert M, Deneubourg J (2013) Impact of increasing deployment of artificial floating objects on the spatial distribution of social fish species. J Appl Ecol 50:1081–1092

Serafy J, Kerstetter D, Rice P (2009) Can circle hook use benefit billfishes? Fish Fish 10:132–142

Serafy J, Cooke S, Diaz G, Graves J, Hall M, Shivji M, Swimmer Y (2012a) Circle hooks in commercial, recreational, and artisanal fisheries: research status and needs for improved conservation and management. Bull Mar Sci 88:371–391

Serafy J, Orbesen E, Snodgrass D, Beerkircher L, Walter J (2012b) Hooking survival of fishes captured by the United States Atlantic pelagic longline fishery: impact of the 2004 circle hook rule. Bull Mar Sci 88:605–621

Sgro C, Loew A, Hoffman A (2011) Building evolutionary resilience for conserving biodiversity under climate change. Evol Appl 4:326–337

Sibert J, Senina I, Lehodey P, Hampton J (2012) Shifting from marine reserves to maritime zoning for conservation of Pacific bigeye tuna (*Thunnus obesus*). PNAS 109:18221–18225

Smith A, Fulton E, Hobday A, Smith D, Shoulder P (2007) Scientific tools to support the practical implementation of ecosystem-based fisheries management. ICES J Mar Sci 64:633–639

Solmundsson J, Jónsdóttir I, Björnsson B, Ragnarsson S, Tómasson G, Thorsteinsson V (2015) Home ranges and spatial segregation of cod *Gadus morhua* spawning components. Mar Ecol Prog Ser 520:217–233

SPC (2010) Review of the implementation and effectiveness of CMM 2008-01. WCPFC7-2010/15. Secretariat of the Pacific Community, Oceanic Fisheries Programme, Noumea

SPREP (2007) Pacific Islands Regional Marine Species Programme 2008–2012. Secretariat of the Pacific Regional Environment Programme, Apia

Stansbury A, Gotz T, Deecke V, Janik V (2015) Grey seals use anthropogenic signals from acoustic tags to locate fish: evidence from a simulated foraging task. Proc R Soc Lond B Biol Sci 282:20141595. https://doi.org/10.1098/rspb.2014.1595

Stevens J, McLoughlin K (1991) Distribution, size and sex composition, reproductive biology and diet of sharks from northern Australia. Aust J Mar Fresh Res 42:151–199

Stevens J, Bonfil R, Dulvy N, Walker P (2000) The effects of fishing on sharks, rays and chimaeras (Chondrichthyans) and implications for marine ecosystems. ICES J Mar Sci 57:476–494

Stobutzki I, Miller M, Heales D, Brewer D (2002) Sustainability of elasmobranchs caught as bycatch in a tropical prawn (shrimp) trawl fishery. Fish Bull 100:800–821

Suding K, Hobbs R (2009) Threshold models in restoration and conservation: a developing framework. Trends Ecol Evol 24:271–279

Suuronen P, Jounela P, Tschernij V (2010) Fishermen responses on marine protected areas in the Baltic cod fishery. Mar Policy 34:237–243

Swimmer Y, Gilman E (eds) (2012) Report of the sea turtle longline fishery post-release mortality workshop, 15–16 Nov 2011. NOAA Technical Memorandum NMFS-PIFSC-34. National Marine Fisheries Service, Pacific Islands Fisheries Science Center, Honolulu

Swimmer Y, Arauz R, Wang J, Suter J, Musyl M, Bolanos A, Lopez A (2010) Comparing the effects of offset and non-offset circle hooks on catch rates of fish and sea turtles in a shallow longline fishery. Aquat Conserv 20:445–451

Thériault V, Dunlop E, Dieckmann U, Bernatchez L, Dodson J (2008) The impact of fishing-induced mortality on the evolution of alternative life-history tactics in brook charr. Evol Appl 1:409–423

Thompson F, Berkenbusch K, Abraham E (2013) Marine mammal bycatch in New Zealand trawl fisheries, 1995-96 to 2010-11. New Zealand Aquatic Environment and Biodiversity Report No. 105. New Zealand Ministry for Primary Industries, Wellington

Thorpe T, Frierson D (2009) Bycatch mitigation assessment for sharks caught in coastal anchored gillnets. Fish Res 98:102–112

Torres-Irineo E, Amandè M, Gaertner D, de Molina A, Murua H, Chavance P et al (2014) Bycatch species composition over time by tuna purse-seine fishery in the eastern tropical Atlantic Ocean. Biodivers Conserv 23:1157–1173

Tuck G, Thomson R, Barbraud C, Delord K, Louzao M, Herrera M, Weimerskirch H (2015) An integrated assessment model of seabird population dynamics: Can individual heterogeneity in susceptibility to fishing explain abundance trends in Crozet wandering albatross? J Appl Ecol 52:950–959

UNEP-WCMC (2016) Marine no net loss: a feasibility assessment of implementing no net loss of biodiversity in the sea. United Nations Environment Programme, Cambridge

Valdemarsen J, Suuronen P (2003) Modifying fishing gear to achieve ecosystem objectives. In: Sinclair M, Valdimarsson G (eds) Responsible fisheries in the marine ecosystem. FAO and CABI International Publishing, London, pp 321–341

van Wijk S, Taylor M, Creer S, Dreyer C, Rodrigues F, Ramnarine I, van Oosterhout C, Carvalho G (2013) Experimental harvesting of fish populations drives genetically based shifts in body size and maturation. Front Ecol Environ 11:181–187

Wade P (1998) Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. Mar Mammal Sci 14:1–37

Walters C, Christensen V, Pauly D (1997) Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. Rev Fish Biol Fish 7:139–172

Wang J, Fisler S, Swimmer Y (2010) Developing visual deterrents to reduce sea turtle bycatch in gill net fisheries. Mar Ecol Prog Ser 408:241–250

Wang J, Barkan J, Fisler S, Godinez-Reyes C, Swimmer Y (2013) Developing ultraviolet illumination of gillnets as a method to reduce sea turtle bycatch. Biol Let 9:20130383. https://doi.org/10.1098/rsbl.2013.0383

Ward P, Myers R (2005) Shifts in open-ocean fish communities coinciding with the commencement of commercial fishing. Ecology 86:835–847

Ward P, Myers R, Blanchard W (2004) Fish lost at sea: the effect of soak time on pelagic longline catches. Fish Bull 102:179–195

Ward P, Lawrence E, Darbyshire R, Hindmarsh S (2008) Large-scale experiment shows that nylon leaders reduce shark bycatch and benefit pelagic longline fishers. Fish Res 90:100–108

Ward P, Epe S, Kreutz D, Lawrence E, Robins C, Sands A (2009) The effects of circle hooks on bycatch and target catches in Australia's pelagic longline fishery. Fish Res 97:253–262

Watson J, Bigelow K (2014) Trade-offs among catch, bycatch, and landed value in the American Samoa longline fishery. Conserv Biol 28:1012–1022

Watson J, Epperly S, Foster D, Shah A (2005) Fishing methods to reduce sea turtle mortality associated with pelagic longlines. Can J Fish Aquat Sci 62:965–981

Waugh S, Baker G, Gales R, Croxall J (2008) CCAMLR process of risk assessment to minimise the effects of longline fishing mortality on seabirds. Mar Policy 32:442–454

Wegmans (2016) Wegmans tuna procurement policy. July 2016. Recommendations for ecologically sustainable sourcing of all tuna products. Wegmans, Rochester

Weijerman M, Fulton E, Janssen A, Kuiper J, Leemans R, Robson B, van de Leemput A et al (2015) How models can support ecosystem-based management of coral reefs. Prog Oceanogr 138:559–570

Weng K, Block B (2004) Diel vertical migration of the bigeye thresher shark (*Alopias superciliosus*), a species possessing orbital retia mirabilia. Fish Bull 102:221–229

Werner T, Kraus S, Read A, Zollett E (2006) Fishing techniques to reduce the bycatch of threatened marine animals. Mar Technol Soc J 40:50–68

Westemeier R, Brawn J, Simpson S, Esker R, Jansen R, Walk J, Kershner E, Bouzat J, Paige K (1998) Tracking the long–term decline and recovery of an isolated population. Science 282:1695–1698

Wilcox C, Donlan C (2007) Compensatory mitigation as a solution to fisheries bycatch biodiversity conservation conflicts. Front Ecol Environ 5:325–331

Willems T, Depestele J, De Backer A, Hostens K (2016) Ray bycatch in a tropical shrimp fishery: do bycatch reduction devices and turtle excluder devices effectively exclude rays? Fish Res 175:35–42

Williams P, Kirby D, Beverly S (2009) Encounter rates and life status for marine turtles in WCPO longline and purse seine fisheries. WCPFC-SC5-2009/EB-WP-07. Western and Central Pacific Fisheries Commission, Kolonia, Federated States of Micronesia

Williams A, Dowdney L, Smith A, Hobday A, Fuller M (2011) Evaluating impacts of fishing on benthic habitats: a risk assessment framework applied to Australian fisheries. Fish Res 112:154–167

WPRFMC (2018) Expert Survey for ranking of alternative seabird bycatch mitigation methods. Workshop to review seabird bycatch mitigation measures for the Hawaii Pelagic Longline Fisheries. Prepared by Gilman E. Western Pacific Regional Fishery Management Council, Honolulu. https://tinyurl.com/seabird-bycatch-ranking. Accessed 20 Sept 2018

WWF (2011) WWF statement on Fish Aggregation Devices (FADs) in tuna fisheries. World Wildlife Fund Smart Fishing Initiative, Hamburg

WWF (2016) WWF position. In: 12th regular session of the scientific committee of the Western Central Pacific Fisheries Commission. WWF, Wellington

Young J, Lansdell M, Campbell R, Cooper S, Juanes F, Guest M (2010) Feeding ecology and niche segregation in oceanic top predators off eastern Australia. Mar Biol 157:2347–2368

Zhou S, Smith A, Punt A, Richardson A, Gibbs M, Fulton E, Pascoe S, Bulman C, Bayliss P, Sainsbury K (2010) Ecosystem-based fisheries management requires a change to the selective fishing philosophy. PNAS 107:9485–9489

Zhou S, Smith A, Fuller M (2011) Quantitative ecological risk assessment for fishing effects on diverse data-poor non-target species in a multi-sector and multi-gear fishery. Fish Res 112:168–178