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Review Article

Ecological data from observer programmes underpin ecosystem-based fisheries management

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Data required from fisheries monitoring programmes substantially expand as management authorities transition to implement elements of ecosystem-based fisheries management (EBFM). EBFM extends conventional approaches of managing single fishery effects on individual stocks of target species by taking into account the effects, within a defined ecosystem, of local to regional fisheries on biodiversity, from genotypes to ecological communities. This includes accounting for fishery effects on evolutionary processes, associated and dependent species, habitats, trophic food web processes, and functionally linked systems. Despite seemingly insurmountable constraints, through examples, we demonstrate how data routinely collected in most observer programmes and how minor and inexpensive expansions of observer data fields and collection protocols supply ecological data underpinning EBFM. Observer data enable monitoring bycatch, including catch and mortality of endangered, threatened and protected species, and assessing the performance of bycatch management measures. They provide a subset of inputs for ecological risk assessments, including productivity–susceptibility analyses and multispecies and ecosystem models. Observer data are used to monitor fishery effects on habitat and to identify and protect benthic vulnerable marine ecosystems. They enable estimating collateral sources of fishing mortality. Data from observer programmes facilitate monitoring ecosystem pressure and state indicators. The examples demonstrate how even rudimentary fisheries management systems can meet the ecological data requirements of elements of EBFM.

Keywords: bycatch, EAF, EBFM, ecological risk assessment, ecosystem approach to fisheries, ecosystem-based fisheries management, ecosystem modelling, endangered, threatened and protected species, habitat degradation, trophic interactions.

Introduction: EBFM extensions of conventional fisheries management

The socio-economic sustainability of marine capture fisheries and the state of marine ecosystems are unequivocally linked (Link, 2002; FAO, 2003). Sustaining target production levels of principal market species by marine capture fisheries requires the persistence of a selected state of ecosystem structure and functions. Recognizing this, the management of marine capture fisheries via an ecosystem approach has been prescribed in major international agreements for over four decades (FAO, 2003, 2009a; Garcia *et al.*, 2003).

Fisheries have direct impacts on target species, but can also have large impacts on incidentally caught species, some of which are particularly vulnerable to overexploitation (marine mammals, seabirds, sea turtles, elasmobranchs, other finfish) and can directly degrade habitat. Fisheries also can have broad, collateral effects, for example, through fishery effects manifested through food web linkages that can cause prolonged changes to ecosystem structure and processes or even permanent regime shifts (Kaiser and de Groot, 2000; Brodziak and Link, 2002; Hilborn *et al.*, 2004; Pikitch *et al.*, 2004). At the often neglected genotype level of biodiversity, fisheries can affect the evolutionary characteristics

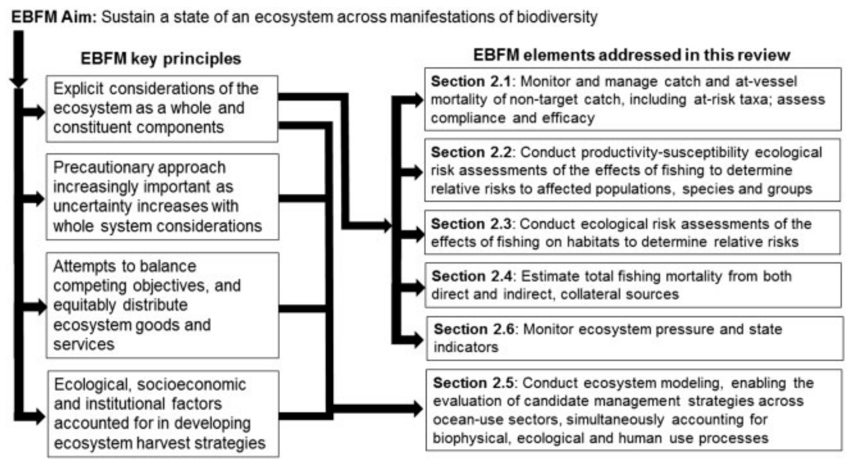

Figure 1. Summary of the aim and selected key principles of EBFM, and visualization of how the examples of observer data discussed in this review relate to these EBFM principles.

of exploited populations and ecosystems. This can occur, for example, from fishing gear selecting for large individuals of a population (Law, 2000; Fenberg and Roy, 2008; Heino and Deickmann, 2008) and from unsustainable fishing mortality of phylogenetically distinct species (Diniz, 2004; Redding and Mooers, 2006; Gilman *et al.*, 2011). Ecosystem-based fisheries management (EBFM), one component of an ecosystem approach to fisheries (EAF), augments conventional target species single-stock, single-fishery management approaches to address these direct and broader effects of fishing (Figure 1; FAO, 2003, 2009a; Garcia *et al.*, 2003).

EBFM aims to sustain a state of an ecosystem across manifestations of biodiversity, from genotypes to communities within the system, including effects from changes to functionally linked ecosystems. Additionally, EBFM aims to sustain the capacity of marine ecosystems to provide services, including fisheries yields, while balancing often competing objectives by equitably distributing these benefits (Link, 2002; FAO, 2003, 2009a; Garcia *et al.*, 2003; Pikitch *et al.*, 2004).

We intentionally have not used here the often employed goal statement of EBFM of sustaining ecosystem “integrity” or “health”. These terms falsely imply that there is an unequivocal target state of an ecological system and that ecological restoration can successfully return all systems altered by anthropogenic stressors to some selected historical, baseline, pre-disturbed reference state (Pitcher, 2001; Link, 2002; Suding *et al.*, 2004; Gilman *et al.*, 2011). Defining a desired ecosystem state is instead based on subjective preferences accounting for biological, ecological, socioeconomic, and governance objectives, where there are necessary trade-offs to balance goals for the persistence of manifestations of biodiversity and for the maintenance or enhancement of selected ecosystems services (Link, 2002; Gilman *et al.*, 2011). To clarify, the "persistence of a selected state of a marine ecosystem" means maintaining or restoring a system’s conditions (structure, processes, resilience) to meet management objectives, such as might be defined as part of an ecosystem-based harvest strategy (Table 1).

EBFM has been described as extending, not replacing or conflicting with conventional fisheries management approaches (FAO, 2003, 2009a; Garcia *et al.*, 2003). This is valid for many but not all EBFM elements. For example, elaborating upon an entry from Table 1, in some fisheries, pursuing single-species management objectives can fail to achieve multispecies objectives when multispecies trade-offs occur (Walters *et al.*, 2005; Smith *et al.*, 2011). Applying single-species maximum sustainable yield (MSY) fishing mortality rates as limit reference points to all species of an ecosystem could substantially alter ecosystem processes and structure (Mace, 2001; Hall *et al.*, 2006). The EBFM extension to single-species MSY-based reference points are multispecies and ecosystem-based reference points, which account for the broader community and ecosystem effects of fishery removals, and are selected based on acceptable levels of risk of protracted or irreparable changes to the ecosystem occurring (Rice, 2000; Pitcher, 2001; Link *et al.*, 2002; Rochet and Trenkel, 2003). The EBFM extension to single stock management through a comprehensive harvest strategy would be a multispecies or broader ecosystem-based harvest strategy, which includes the suite of elements identified in Table 1. Observer data are inputs to multispecies and ecotrophic models, which are used to derive system-level reference points and other harvest strategy components (Table 1 and Figure 1) (Link *et al.*, 2002).

The slow uptake of EBFM has been attributed to many factors. The rudimentary state of some fisheries governance systems, including institutional and financial limitations, lack of political will, basic monitoring systems, limited fisheries data and sparse knowledge of the direct and collateral ecological effects of fishing, have been identified as obstacles to transitioning to EBFM (Leslie *et al.*, 2008; Tallis *et al.*, 2010; Gilman *et al.*, 2014). The argument is that because many fisheries management authorities struggle to implement basic activities of traditional single-stock and single fisheries management systems owing to these limitations, taking on the more demanding EBFM requirements should not be pursued until these management authorities can first effectively implement fundamental conventional management tasks. The use of widely different and often abstract definitions of EBFM and broader EAF (Garcia *et al.*, 2003; US Commission on Ocean Policy, 2004; Hall and Mainprize, 2005) has been identified as an additional obstacle hampering EBFM implementation (Patrick and Link, 2015).

Here we counter these arguments by illustrating how management systems, even those that are rudimentary and governing data-poor fisheries, can successfully transition to meeting the ecological

Table 1. Examples of elements of an ecosystem approach to fisheries management that broaden, and in some cases, replace conventional approaches (Rice, 2000; Sainsbury *et al*., 2000; Collie and Gislason, 2001; Link, 2005; WCPFC, 2014a).

Conventional	EAF extension
Narrow scale, monitor, assess, and manage effects of a single fishery on single stocks of principal market species	**Nested scales**, monitor, assess, and manage effects of local to regional fisheries on all manifestations of biodiversity (from genotypes to communities) within a defined ecosystem, including effects on associated and dependent species, habitat effects, and effects on evolutionary processes, trophic processes and functionally linked systems.
Monitor, assess and manage **direct fishing mortality of stocks of principal market species**	Monitor, assess and manage effects of fishing on the **ecosystem as a whole**, and all constituent components, from genotypes to communities, within the system, including **all sources of fishing mortality**, including from indirect, collateral effects of fishing operations, **for all affected** target and associated and dependent species, effects on **habitat**, and broader effects (e.g. trophic connectivity, size structure, diversity).
Single stock assessment models for principal market species	**Multispecies and ecosystem models** to assess broad effects of management options (as well as define a system's reference state, patterns and trends in change).
Single-stock harvest strategy with the following stock-specific elements: *Management objectives* for the stock *Target reference point*, selected based on ecological and socio-economic considerations, to meet the adopted management objectives *Limit reference point*, selected to constrain harvesting within safe biological limits (e.g. prevent exceeding a point of recruitment impairment of the stock), usually based on spawning stock biomass and fishing mortality *Acceptable levels of risk* of exceeding the limit reference point and not achieving the target reference point *Monitoring strategy* to assess performance against reference points, i.e. assess the status of the stock to determine its status in relation to target and limit reference points *Harvest control rule (HCR)*: Pre-agreed management actions that are triggered when there is a change in stock status with respect to reference points. Designed to keep stocks near targets and to not exceed limits, reducing fishing mortality rates when reference points are approached, and increasing fishing mortality rates as stocks trend above TRPs. *Management strategy evaluation (MSE)* to evaluate the likely performance of alternative HCRs against operational management objectives, including risk assessment, such as uncertainty with stock assessments.	**Ecosystem-based harvest strategy** with the following ecosystem-level elements: *Management objectives* define the desired state of the ecosystem and its constituent parts in order to sustain a desired level of provision of ecosystem goods and services. *Target reference point*, selected based on ecological and socio-economic considerations, attempts to balance competing objectives and to equitably distribute ecosystem goods and services as defined by the adopted management objectives. *Limit reference point*, selected to avoid causing an irreversible change in state of the system. *Acceptable levels of risk* of breaching the limit reference point, and of causing a protracted or irreparable change in ecosystem state *Ecosystem indicators*: select and monitor a suite of indicators that track trends in manageable *pressures* that alter broad ecosystem-level functions and structure, suite of indicators of the *state* of ecosystem components, and indicators to measure the *response* of managers to alter the level of a pressure that has resulted in an unwanted change in ecosystem state. *Monitoring strategy* using the ecosystem indicators to assess whether the system is near the target and not approaching the limit reference point *HCR* defining pre-agreed management actions, triggered to stay near ecosystem-based targets and to not exceed ecosystem-level limits, applying a precautionary approach to address uncertainty and concomitant risks. *MSE*, to evaluate the performance of alternative ecosystem HCRs against operational management objectives, using multispecies or ecosystem modelling approaches.

data requirements of some elements of EBFM (Figure 1). We accomplish this through examples that illustrate how: (i) data routinely collected in many observer programmes supply ecological data needed to implement many EBFM components, and (ii) minor and inexpensive expansions of fisheries observer programme data fields and collection protocols supply data that are not typically collected observer programmes, which are needed to meet some of the ecological data requirements of EBFM. Some examples demonstrate how observer data support single-species approaches for at-risk bycatch, one EBFM element. Other examples demonstrate the use of observer data to monitor and manage fisheries' effects on habitat. Examples are also drawn from multispecies and ecosystem assessment methods. The data requirements of fisheries monitoring programmes have substantially expanded as management authorities throughout the world have begun to transition to implementing elements of EBFM, with various degrees of success (Pitcher *et al*., 2009; Gilman *et al*., 2014). Despite substantial constraints, we demonstrate that it is possible, now, for at-sea observer programmes to supply ecological data that support the implementation of EBFM.

Observer programme data collection to support EBFM elements

Monitoring bycatch and assessing the performance of bycatch management measures

Monitoring and managing fisheries bycatch, including of endangered, threatened, and protected (ETP) species, is one component of EBFM. Expanding from a focus on principal market species, responsibility to conserve incidental catch and associated and dependent species first became an international obligation under the 1982 Law of the Sea Convention (United Nations, 1982).

Data collection methods of both human and electronic observer programmes, including categories of information to collect, and methods employed to collect the data, can be designed to support robust statistical analyses of bycatch interactions (Hall, 1999; FAO, 2002; Gilman *et al.*, 2014a; Gilman and Hall, 2015). Management objectives for analyses of observer bycatch data, including desired levels of accuracy and precision of bycatch and survival rate estimates, determine the selection of observer data fields and collection protocols. Management objectives for the use of observer bycatch data will also determine coverage rates and how to provide for the random and representative allocation of sampling effort (Hall, 1999; FAO, 2002; Babcock *et al.*, 2003). Observer bycatch data collection methods require periodic adaptation as scientific requirements, regulations, market conditions, and fishing vessel equipment, gear and practices evolve over time.

In this section, we provide a sample of examples from pelagic longline and purse seine fisheries to illustrate some of the various applications of observer data to implement the EBFM component of monitoring and managing bycatch.

Fleet-wide bycatch levels and nominal catch rates

Observer data from a sample of fishing effort from fisheries with partial coverage rates are routinely raised to produce fleet-wide estimates of bycatch. For example, management authorities of the Hawaii longline tuna fishery and New Zealand Southern Bluefin Tuna fishery modelled observer data to estimate annual ETP fleet-wide catch quantities (McCracken, 2014; Ministry for Primary Industries, 2015). And, observer data have been used to estimate raised bycatch levels of European purse seine tuna fisheries of the Atlantic Ocean (Amande *et al.*, 2010) and of eastern Pacific Ocean large-scale purse seine vessels (IATTC, 2015a, b). The observed bycatch levels and observed sample of effort enable estimating observed nominal bycatch rates.

In the Hawaii longline swordfish fishery estimates of annual ETP catch levels have been based on 100% observer coverage since 2004 (NMFS, 2005; McCracken, 2014). From the observer data on ETP catch levels and total effort, nominal ETP species catch rates can be calculated (e.g. for seabirds, NMFS, 2016a).

These fishery-wide bycatch estimates support many EBFM applications. Bycatch estimates based on observed data are used by management authorities to determine when bycatch quotas are reached (e.g. annual sea turtle catch limits in the Hawaii longline swordfish fishery, NMFS, 2016b), contributing to the implementation of the EBFM element of accounting for fishery effects on associated and dependent species. Bycatch observer data can be assessed to determine if temporal trends in catch levels and nominal catch rates are occurring. For instance, observer data enabled Walsh *et al.* (2009) to determine that nominal catch rates of some shark species have significantly declined in recent years in the Hawaii longline tuna and swordfish fisheries. Analysis of observer data from the Hawaii longline tuna fishery documented increasing temporal trends in seabird raised annual catch levels and nominal catch rates (Gilman *et al.*, 2016a). Estimates of bycatch and mortality levels from observer programme data, including length frequency distributions, sex ratios, temporal and spatial distribution of bycatch and effort, and time series of bycatch-per-unit-of-effort are used as inputs to population and stock assessment models (e.g. Chaloupka, 2002; Dans *et al.*, 2003; Lewison and Crowder, 2003; ISC, 2016) and to multi-species and ecosystem models, so that emergent ecosystem properties can be assessed and ecosystem changes can be quantified (see section 'Quantitative model-based ecological risk assessments—examples from ecosystem models'). Therefore, these and numerous other national and regional fisheries observer programmes that already collect basic data on bycatch and effort support various single-species and broader multi-species and system-level EBFM approaches that make use of estimates of bycatch levels and rates (Figure 1).

Standardized catch and at-vessel mortality rates

Observer data can be fit to standardized catch and survival rate models for bycatch species. Findings enable estimating, for example, an index for temporal trends in relative abundance (e.g. Walsh and Clarke, 2011; Gilman *et al.*, 2012a; Rice, 2012). Catch and survival rate model outputs also enable the identification of categorical factors and covariates that significantly explained bycatch and at-vessel survival risk, which inform bycatch management. For example, the factor hook shape (circle- vs. J-shaped) significantly explained blue shark (*Prionace glauca*) catch risk in the Hawaii longline tuna fishery, based on fitting observer catch data to a standardized catch rate model that explicitly accounted for other significant factors and covariates (Gilman *et al.*, 2012a). The covariate season and geospatial location of fishing effort had significant effects on seabird catch risk in the Hawaii longline tuna fishery (Gilman *et al.*, 2016a) and on catch risk of some shark species in the Marshall Islands longline tuna fishery (Bromhead *et al.*, 2012). Sea surface temperature was found to significantly explain standardized blue shark at-vessel mortality rate in the Palau pelagic longline tuna fishery (Gilman *et al.*, 2015).

Simple, inexpensive and practical modifications to observer programme data fields and data collection protocols can be made to supply data prioritized for bycatch monitoring and management, including information on potentially significant factors and covariates that could be explicitly accounted for in standardized catch and survival rate models. This was exemplified by the Western and Central Pacific Fisheries Commission (WCPFC), a tuna regional fisheries management organization (RFMO), through implementation of recommendations of the Joint Tuna RFMO Technical Working Group-Bycatch on harmonizing observer bycatch data for pelagic longline fisheries (Gilman and Hall, 2015; ISSF, 2015; WCPFC, 2016). The WCPFC Regional Observer Programme is implemented in several Pacific small island developing states with relatively rudimentary fisheries management systems, as well as in developed countries with relatively sophisticated, comprehensive management frameworks. In 2016, WCPFC expanded observer fields and data collection protocols by the WCPFC Regional Observer Programme to support improved bycatch monitoring and management (WCPFC, 2016). For example, the WCPFC observer programme was amended to have longline observers record anatomical hooking position (mouth-hooked, deeply hooked, externally hooked) and record what terminal tackle remained attached to ETP species that were released alive, both of which affect survival rates (see section 'Collateral sources of fishing mortality').

WCPFC also amended their regional observer programme to have longline observers record the number of "shark lines" deployed per set (WCPFC, 2016). Catch and at-vessel survival rates of some species are significantly different on shark lines, which are branchlines that soak at or near the surface through attachment directly to floats or floatlines, vs. branchlines attached to the mainline that soak at deeper depths (Bromhead *et al.*, 2012; Gilman *et al.*, 2015). A WCPFC measure includes a ban on the use of shark lines as one option for longline fisheries (WCPFC, 2014b). Thus, in addition to supporting robust catch and survival rate modelling, observer data collection on shark line use also supports assessments of compliance with and efficacy of this measure (see sections 'Monitoring compliance with bycatch measures' and 'Inferring the effect of bycatch mitigation management measures'). Many of the new observer data fields were subsequently used to evaluate alternative strategies for managing sea turtle catch and mortality in regional longline fisheries (WCPFC and SPC, 2016).

As a final example, in 2004 the Hawaii longline observer programme began to have at-sea observers record the number of seabirds attending vessels during setting and gear haulback. These data improved the certainty of standardized seabird catch rate models by enabling the models to explicitly account for the effect of the density of seabirds attending the vessel on catchability (NMFS, 2010; Gilman *et al.*, 2016a). These examples demonstrate how both existing data routinely collected by observer programmes, and how small improvements to observer bycatch data quality support EBFM approaches that use outputs from standardized catch and survival rate models.

Analyses of tissue samples

Tissue samples of bycatch collected by observers may be used for genetic analyses to determine which populations the fishery affects (e.g. odontocetes, NMFS, 2012, 2015) and to assess population structure of a species caught in a fishery (e.g. oceanic whitetip shark *Carcharhinus longimanus*, Camargo *et al.*, 2016). Discussed in more detail in section 'Semi-quantitative ecological risk assessments (ERAs) of the effects of fishing on populations, stocks and species', analyses of other tissue samples (otoliths and other fish hard parts, stomachs, gonads) collected by observers can contribute to understanding life history attributes, information used to assess relative (see section 'Semi-quantitative ERAs of the effects of fishing on populations, stocks and species') and absolute population-level fishery effects, and used in robust ecosystem models (see section 'Quantitative model-based ecological risk assessments—examples from ecosystem models').

Monitoring compliance with bycatch measures

Various data collected by observers can support monitoring compliance with bycatch mitigation measures. For example, data on the date, time and vessel spatial position during fishing operations enable assessing compliance with temporal and spatial restrictions on fishing. Data on the time of day of fishing operations enable assessments of compliance by longline vessels with requirements for night setting to mitigate seabird bycatch and compliance with a prohibition on night setting by eastern Pacific Ocean purse seine vessels making dolphin-associated sets (AIDCP, 2009; NMFS, 2016a). Observer data also enable assessments of compliance with seasonal and area closures. For instance, compliance with a seasonal closure on purse seine sets on fish aggregating devices has been assessed by using observer catch data, including data on bycatch composition (Hare *et al.*, 2015).

Observer data on the fate of the catch (e.g. retained, released alive, discarded dead, shark fins retained, and carcass discarded) have been used to monitor compliance with retention bans and shark finning restrictions (e.g. Clarke *et al.*, 2013; Gilman *et al.*, 2015; Piovanno and Gilman, 2016). For example, Clarke *et al.* (2013) assessed compliance with a regional measure restricting shark finning by reviewing observer data from purse seine and pelagic longline fisheries operating in the western and central Pacific Ocean.

Observer data on gear designs and fishing methods also enable monitoring compliance with prescribed bycatch mitigation methods. For instance, observer data from the Fiji longline tuna fishery revealed high compliance with a ban on the use of shark lines (Piovanno and Gilman, 2016). Thus, existing observer programme designs enable assessment of compliance with bycatch measures, a component of EBFM (Figure 1).

Inferring effects of bycatch mitigation management measures

Data already being collected by many fisheries observer programmes support various approaches for assessing the efficacy of bycatch management measures. Nominal and standardized catch rates calculated from observer data from before and after bycatch mitigation regulations came into effect have been used to infer the performance of bycatch mitigation measures. Gilman *et al.* (2008) fit observer seabird catch data from the Hawaii longline tuna fishery to a standardized catch rate model to assess the change in catch risk following the introduction of regulations requiring employment of seabird bycatch mitigation methods. Observer programme data have been analysed to monitor changes in the proportion of caught sharks released alive and to estimate minimum mortality rates following the adoption of finning restrictions and bans on shark retention (e.g. Walsh *et al.*, 2009; Gilman *et al.*, 2015; Piovanno and Gilman, 2016).

Findings from controlled and comparative experiments are a critical first step to assess the efficacy and commercial viability (practicality, economic viability, safety) of a candidate bycatch mitigation method. Unlike in properly designed experimental studies, analyses of observer data do not experimentally manipulate-specific variables and control for others. As a result, estimated effects of individual variables from analyses of observer data are always confounded by other variables. However, during commercial operations, captain, and crew implementation of bycatch mitigation methods that rely on crew behaviour for compliance with prescribed implementation can differ from implementation during research experiments. In some cases this can result in substantial differences in the efficacy of the mitigation method (Gilman *et al.*, 2005; Cox *et al.*, 2007). Consequently, properly designed analyses of observer data that explicitly account for potentially significant explanatory factors and covariates provide one of the most reliable methods to assess the in-practice performance of bycatch mitigation methods.

In both experiments and analyses of observer programme data, bias from the presence on board of an observer (observer effect) can occur. This level of bias can and should be estimated (Hall, 1999; Liggens *et al.*, 1997; Babcock *et al.*, 2003) and accounted for in monitoring bycatch, including assessing the efficacy of mitigation methods, an additional element of EBFM.

Semi-quantitative ecological risk assessments of the effects of fishing on populations, stocks and species

Another EBFM element is assessing and managing risks that fisheries pose to high-risk populations and species (Figure 1 and

Table 1). 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 (MSC, 2010; Hobday *et al.*, 2007, 2011). This section provides examples of how observer data provide inputs for semi-quantitative ERA methods.

The objective of analysis of most semi-quantitative fisheries ERAs has been to determine population- and species-level relative risks from fishing mortality of taxonomic groups especially vulnerable to overexploitation (seabirds, sea turtles, marine mammals, and elasmobranchs), most employing productivity–susceptibility analyses (PSAs) (e.g. Stobutzki *et al.*, 2002; Waugh *et al.*, 2008; Cortes *et al.*, 2015). Few ERAs have holistically assessed relative risks from fishing operations across affected taxonomic groups or risks at other levels of marine biodiversity, including effects on genetic diversity and evolutionary processes resulting from selective fishery removals. ERAs have also largely not assessed broad community- and ecosystem-level fishery effects, nor assessed risks from collateral effects of fishing operations (Hobday *et al.*, 2011; Gilman *et al.*, 2014).

Findings from ERAs, an EBFM assessment approach which relies partly on observer data inputs, have been used by the Australian government to develop management responses to identified highest priority ecological risks to species, habitats and ecological communities. For example, an ERA for the Australia eastern tuna and billfish fishery identified nine high risk species. No target species, habitats or ecological communities were found to be high risk. Based on the ERA findings, the government adopted measures to manage bycatch (AFMA, 2012).

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 the level of fishing mortality. Attributes used for susceptibility include those that describe the overlap between a population, stock or species and a fishery spatially and temporally, the probability that the species interacts with fishing

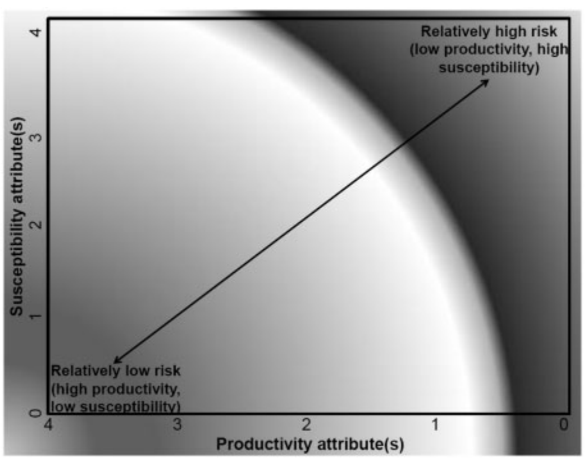

Figure 2. Sample productivity–susceptibility analysis plot, an example of a semi-quantitative ecological risk assessment of the effects of fishing on populations, stocks, species or habitat (adapted from Hobday *et al.*, 2011; Williams *et al.*, 2011).

vessels, the species' catchability, and the probability of injury and mortality as a result of a fishery interaction (Hobday *et al.*, 2011). Figure 2 presents a generic PSA plot where, for example, a species with high productivity and low susceptibility has lower risk from the fishery relative to species with lower productivity and higher susceptibility scores. A single risk score of assessed species may be determined by their position on the productivity and susceptibility axes, such as by calculating the Euclidian distance from the origin of the PSA plot, in order to provide a rank-order of relative risk (Kirby, 2006; Williams *et al.*, 2011; Nel *et al.*, 2012; Cortes *et al.*, 2015).

Some attributes that have been used to characterize relative productivity in PSAs can be estimated, in part, using observer data on length, weight and sex of the catch. Observer data from the collection of samples of fish hard parts (otoliths, scales, opercula, spines, vertebrae) to determine age; and collection and at-sea analysis of stomach contents and gonads also contribute to determining some PSA productivity attributes (e.g. by regional tuna fishery bodies, CCSBT, No Date; IOTC, 2010; Waugh *et al.*, 2012; IATTC, 2014; SPC and FFA, 2014; WCPFC, 2015); including:

- Intrinsic rate of increase (natural growth rate of a population)
- Age and size at first maturity
- Maximum age (lifespan) and maximum size
- Age-specific natural mortality
- Fecundity
- Trophic level
- Recruitment
- Reproductive strategy (e.g. broadcast spawner, egg layer, live bearer)
- Potential biological removal (marine mammals, Wade, 1998) and various adaptations (e.g. seabirds, Dillingham and Fletcher, 2011; sea turtles, Curtis and Moore, 2013).

Many attributes have been used to characterize relative susceptibility in PSAs (Stobutzki *et al.*, 2002; Nel *et al.*, 2012; Waugh *et al.*, 2012; Sharp *et al.*, 2013; Cortes *et al.*, 2015). Most of these attributes can be determined through fisheries observer data on gear characteristics, fishing methods and catch (see section 'Monitoring bycatch and assessing the performance of bycatch management measures'), such as:

- The degree of spatial (geo-spatial and depth) and temporal (time-of-day, season) overlap between a population, stock or species and a fishery, and the proportion of each age class that overlaps the fishery. This requires information on the geospatial, depth, and seasonal distribution of fishing effort and of each age class of the population/stock/species.
- Gear designs that affect species and size selectivity (e.g. gillnet mesh size, longline hook size).
- Catch rates and levels by species and age class.
- Observable components of total fishing mortality rates and levels by age class, including estimates of at-vessel mortality rates (proportion of the catch that is alive vs. dead at haulback, before being handled by crew), fate of the catch (proportion of the catch that is retained, released alive, or discarded dead after being brought onboard), post-release mortality rates (proportion of the catch that is released alive that subsequently die as a result of the fishery interaction; see section 'Collateral sources of fishing mortality'), and pre-catch loss rates (proportion of

the catch that dies from the fishing operation but is not brought onboard when the gear is retrieved; see section 'Collateral sources of fishing mortality').

- Components of total fishing mortality that are not readily detectable directly by onboard observers, but can be estimated using observer data, such as ghost fishing mortality (e.g. using observer records of abandoned, lost and discarded fishing gear, Gilman, 2015; Gilman *et al.*, 2016c), and causes of pre-catch mortality (e.g. comparing observer estimates of bird captures during setting vs. the number hauled aboard to determine a pre-catch loss rate, Gilman *et al.*, 2013; see section 'Collateral sources of fishing mortality').

In summary, observer data supply inputs to semi-quantitative ERAs, one tool supporting the management of fishery effects on highest risk populations and species, which is one element of EBFM (Figure 1). This includes inputs for attributes used to characterize productivity and susceptibility for assessments of relative risks posed by fisheries to species within taxonomic groups.

Identification of sites meeting definitions of Vulnerable Marine Ecosystems and conducting habitat PSAs

Fisheries observer data are used to monitor, assess, and manage fishery effects on habitat, another EBFM component (Figure 1). Fishing gear contact with the substrate can cause protracted changes to benthic community structure and functions (e.g. Kaiser and de Groot, 2000; Kaiser *et al.*, 2006; FAO, 2009b). Observer data are used by several RFMOs to implement measures to identify and protect benthic vulnerable marine ecosystems (Gilman *et al.*, 2012b, 2014). To manage fishery effects on vulnerable benthic communities (e.g. seamounts, hydrothermal vents, cold water coral reefs, and sponge fields), these RFMOs have adopted management measures that include explicit definitions to identify benthic areas as vulnerable marine ecosystems, and have adopted observer data collection protocols that enable implementation of these measures. The definitions are based on threshold catch rates of live corals and sponges. Areas that meet the definition may be immediately subject to a move-on provision, and later be considered for permanent closure to demersal fisheries (SEAFO, 2009; CCAMLR, 2010; NAFO, 2010; NEAFC, 2010).

As with semi-quantitative ERAs of the effects of fishing on populations, stocks and species (see section 'Semi-quantitative ecological risk assessments of the effects of fishing on populations, stocks and species') observer data are also used for some of the data inputs for risk assessments of fishery effects on benthic habitats (Figure 1) (Hobday *et al.*, 2011; Williams *et al.*, 2011). For example, observer data on the spatial distribution of fishing effort and depth of gear can be used to assess the degree of "encounterability" that the fishery has with different habitat types. Encounterability is one attribute for susceptibility to damage that can be used in a PSA for the habitat effects of fishing (Hobday *et al.*, 2011; Williams *et al.*, 2011). The relative degree of disturbance a fishery causes to a habitat type (e.g. does a single encounter cause high damage, or are many encounters required to cause damage), another PSA susceptibility attribute (Williams *et al.*, 2011), can also be assessed, in part, using observer data on characteristics of the fishing gear, such as the size, weight, design and mobility of demersal trawl gear. Observer data on habitat bycatch (e.g. deep sea coral fragments) have been used to assist in mapping habitat distributions and productivity attributes, such as substrate hardness, used in PSAs (Williams *et al.*, 2011).

Collateral sources of fishing mortality

Reliable ecosystem models, as well as conventional single stock assessment models (see section 'Quantitative model-based ecological risk assessments—examples from ecosystem models') require high certainty estimates of *total* fishing mortality, which includes collateral, not readily detectable sources (Broadhurst *et al.*, 2006; Gilman *et al.*, 2013; Uhlmann and Broadhurst, 2015). Collateral sources of fishing mortality include pre-catch, post-release, and ghost fishing losses. Fishing mortality can also occur from cumulative and interacting indirect effects of fishing, such as when repeated sub-lethal interactions result in mortality, when released catch is displaced from habitat used for shelter and dies from predation as the organism swims back to its preferred habitat, and from habitat degradation such as anoxia from discards and habitat loss caused by fishing gear (Broadhurst *et al.*, 2006; Gilman *et al.*, 2013). Ecosystem models such as the Atlantis model framework (see section 'Quantitative model-based ecological risk assessments—examples from ecosystem models') can be designed to account for habitat degradation, ghost fishing and other indirect fishing mortality sources.

There are several examples of fisheries in which components of collateral mortality are routinely monitored using observer data. Some observer programmes record information on the amount and location of terminal tackle remaining attached and other information used to estimate the probability of post-release survival. For example, observer data on pelagic longline gear remaining attached to organisms released alive, the anatomical location of hook and line, and the species, size, and sex of catch have been used to estimate the probability of post-release survival (e.g. sea turtles, Ryder *et al.*, 2006; false killer whales *Pseudorca crassidens*, NMFS, 2012).

Observer data on indicators of degree of injury (condition and vitality) of organisms that are released alive can inform estimates of the probability of post-release survival (e.g. condition codes used in the Pacific Community observer programme for pelagic longline and purse seine fisheries, SPC, 2014a, b). However, observer condition/vitality categorizations may be poor predictors of post-release survival (Broadhurst *et al.*, 2006; Musyl *et al.*, 2015).This may be because condition/vitality measures have inconsistent responses to different types of fishing stressors and because categorizations of wounds are largely subjective and thus introduce bias in mortality estimates (Davis, 2002).

Analyses of observer data have been used to assess the effect of various longline gear design factors on anatomical hooking position (Gilman *et al.*, 2016b; Gilman and Huang, 2017). This provides an indication of the degree of injury and concomitant probability of pre-catch, haulback and post-release survival (Swimmer and Gilman, 2012; Gilman *et al.*, 2013; Gilman and Hall, 2015; Parga *et al.*, 2015).

Observers in the New England herring trawl fishery estimate slipped pre-catch, so that higher certainty estimates of total fishing mortality can be used for stock assessment inputs (New England Fishery Management Council, 2011). Pre-catch losses of seabirds caught in longline and trawl fisheries have been estimated by comparing counts of bird captures during setting to the number retrieved during gear haulback, where results have been used by regulators to base management decisions on more certain estimates of the effect of the fishery on affected seabird populations (e.g. USFWS, 2004; Gilman *et al.*, 2013). As a final example, in some fisheries, observers record abandoned, lost and discarded fishing gear, which can be used to estimate ghost fishing mortality rates and quantities (Gilman, 2015; Gilman *et al.*, 2016c). Therefore, some observer programmes currently collect data needed to estimate collateral components of fishing mortality, supporting implementation of EBFM elements.

Quantitative model-based ecological risk assessments—examples from ecosystem models

Quantitative ERAs employ model-based analyses (Hobday *et al.*, 2011). Conventional single stock assessment methods are one form of a quantitative ERA, typically used for principal market species, to assess the status and temporal changes in stock status and predict stock responses to different management options (Hilborn and Walters, 1992). The EBFM extension, multispecies, and ecosystem models, create simplified versions of an ecosystem and can simulate the main dynamics, and use fisheries observer programme data for some model inputs.

Through synthesizing multidisciplinary datasets, multispecies, and ecosystem models can: (i) define a whole system's reference state (structure and processes); (ii) determine patterns and trends in ecosystem changes in response to pressures, including from fishing; and (iii) evaluate socio-economic and ecological effects from alternative management options [e.g. Ecopath with Ecoism (EwE): Polovina, 1984; Walters *et al.*, 1997; Christensen and Walters, 2004a; Atlantis: Fulton and Smith, 2004; Fulton *et al.*, 2004; models of intermediate complexity for ecosystem assessment [MICE]: Plaganyi *et al.*, 2014]. There is a wide range of ecosystem modelling frameworks, varying in their balance between realism, accuracy and complexity, each with different degrees of data requirements (Weijerman *et al.*, 2015). Different types of models are suitable for addressing different types of management-related questions (Weijerman *et al.*, 2015) where different modelling approaches require different model inputs.

A subset of the information critical for building robust ecosystem models is supplied by observer datasets (Figure 3), including data inputs on (Plaganyi, 2007; Travers *et al.*, 2007; Fulton *et al.*, 2007, 2014):

- Components of total fishing mortality (e.g. observer data on retained catch, catch discarded dead, and catch released alive, with various fields to estimate probability of post-release mortality; see sections 'Monitoring bycatch and assessing the performance of bycatch management measures' and 'Collateral sources of fishing mortality').
- Spatial and temporal relative species abundance index (e.g. fitting observer catch data to standardized catch rate models).
- Size structure of the catch (from observer catch, length, weight, and age data).
- Selectivity of fishery removals, including the relative catchability of different functional groups (e.g. from observer data on gear types and attributes, such as gillnet mesh size, and depth of the gear).
- Spatial and temporal location of fishing effort.
- Life history attributes, especially of higher trophic level species (see section 'Semi-quantitative ecological risk assessments of the effects of fishing on populations, stocks and species').
- Trophic linkages (through a diet matrix) (e.g. from analyses of observer data on stomach contents).
- Damage to benthic habitats (see section 'Identification of sites meeting definitions of vulnerable marine ecosystems and conducting habitat PSAs').
- Ecosystem state indicators that are sensitive to fishing pressure and for which trends in the indicator can be interpreted [e.g. mean trophic level of the catch (TLc), estimated using observer catch data on species, length, and weight) (see section 'Monitoring ecosystem pressure and state indicators'].

Outputs from multispecies and ecosystem models are being used in some fisheries management frameworks. An Atlantis ecosystem model developed for the southeast Australian Commonwealth scalefish and shark fishery provides an example of a complex ecosystem model used for quantitative management strategy evaluation (Fulton *et al.*, 2007; Smith *et al.*, 2007). The model used all of the data inputs listed above, relying in part on datasets from fishery observer programmes, as well as data on fleet-specific costs (e.g. ice, fuel, maintenance) and revenues, as the model coupled socio-economic dynamics with ecological and physical dynamics.

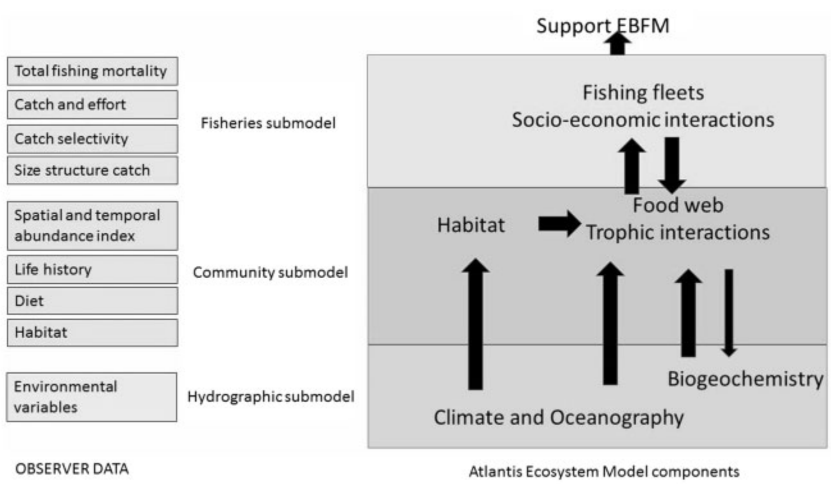

Figure 3. Conceptual model of the Atlantis model framework where fisheries observer data are inputs for many of the core component modules (adapted from a figure courtesy Beth Fulton, CSIRO).

A less complex EwE model developed for coral reef ecosystems in Indonesia (Ainsworth *et al.*, 2008) was based mainly on trophic linkages but also included damage to benthic habitat and various catch-specific data inputs (e.g. size structure of the catch, selectivity of fishery removals by gear type, ecosystem state indicators), which could be collected by fishery observers. This model was used to investigate research questions related to EBFM.

Findings from multispecies and ecosystem models have also been used to improve single-species assessments. For example, management authorities have used outputs from multispecies models that approximate predator–prey interactions to improve the certainty of predation mortality time series fitted to single stock assessment models (reviewed in Link *et al.*, 2011). Ecosystem model findings have been used to investigate the role of at-risk taxa in the predation or competition for prey of target species, such as seal predation of target groundfish stocks (Bundy, 2001; Chassot *et al.*, 2009; Link *et al.*, 2011). Therefore, data already being collected in most fisheries observer programmes provide fundamental inputs to quantitative ERAs, from conventional single stock assessments to ecosystem models.

Monitoring ecosystem pressure and state indicators

Fishery management authorities can select and monitor a suite of indicators that track trends in manageable fishing pressures that alter broad ecosystem-level functions and structure, indicators of the state of ecosystem components and indicators of the response of managers to alter the level of a pressure that has resulted in an unwanted change in ecosystem state (Jennings, 2005; Piet *et al.*, 2007). Monitoring these pressure, state and response indicators allows assessment of progress towards meeting ecosystem-level thresholds selected to meet ecological and socio-economic management objectives, guiding the adaptation of controls on fishery pressures, and communicating trends in indicators and management actions (Garcia *et al.*, 2000; Jennings, 2005; Rice and Rivard, 2007; Fay *et al.*, 2015; Weijerman *et al.*, 2016). The selection and monitoring of suites of pressure, state, and response ecosystem indicators is one core element of a robust ecosystem-based harvest strategy (Table 1).

To comprehensively detect the effects of fishing on the state of an ecosystem, including evaluating whether ecosystem-based harvest control rules are maintaining the ecosystem near an ecosystem-based target reference point and are achieving management objectives, it is necessary to monitor a suite of ecosystem state indicators. Collectively, the suite should span ecosystem attributes (Fulton *et al.*, 2005; Link, 2005; Rice and Rochet, 2005). Here, we present examples of how observer data can be used to monitor a sample of ecosystem state indicators that are sensitive to fishing pressure and for which trends in the indicator can be interpreted.

Temporal trends in the mean TLc, a univariate ecosystem state indicator, can be determined from observer catch data on species, length, and weight. A declining trend in the mean TLc of a fishery indicates that the fishery has been shifting to catching smaller organisms of the same target species and/or to lower trophic level species, which may have resulted from declining local abundance of top trophic level species (Cury and Christensen, 2005; Fulton *et al.*, 2005; Allain *et al.*, 2015). For example, a recent ecosystem modelling assessment using EwE for the Pacific warm pool ecosystem, which used fisheries observer programme data for some model inputs, observed an increasing trend in TLc. The authors hypothesized that this may have resulted from the spatial expansion of regional tuna fisheries and concomitant increased catches of high trophic level bycatch species (Allain *et al.*, 2015).

Allain *et al.* (2015) also observed a decreasing temporal trend in Kempton's Q index. This multivariate ecosystem state indicator is sensitive to fishing pressure and relies on observer data inputs. A decreasing trend indicates a reduction in the number (species richness) and biomass (species evenness) of upper trophic level (trophic levels >3) functional groups (Kempton and Taylor, 1976; Kempton, 2002; Allain *et al.*, 2015). Kempton's Q index, originally developed to describe species diversity, has been adapted for use with EwE (Christensen and Walters, 2004b; Ainsworth and Pitcher, 2006; Allain *et al.*, 2015). This ecosystem state indicator uses functional groups in place of individual species, either for all trophic levels of a system or just for high trophic level groups (Christensen and Walters, 2004b; Ainsworth and Pitcher, 2006).

Temporal trends in length frequency distributions of a species is another example of a univariate ecosystem state indicator sensitive to fishing pressure that can be monitored with observer data (Hilborn and Walters, 1992; Fulton *et al.*, 2004; Gilman *et al.*, 2012a). Temporal changes in mean and maximum lengths are considered part of a core set of ecosystem indicators for EBFM (Fulton *et al.*, 2004; Link, 2005). A shift in length frequency distribution (size structure) of a population towards smaller fish could be the result of selective removal by the fishery of large individuals of the population. This could alter the evolutionary characteristics of these populations by creating a driver favouring genotypes for maturation at an earlier age, smaller size and slower growth (Ward and Myers, 2005; Zhou *et al.*, 2010). Shifts in length frequency distributions of the catch can occur from factors other than fishing mortality. For example, a change or expansion in the spatial distribution of fishing effort, change in fishing methods and gear that affect size selectivity, and changes in environment conditions that affect, for instance, recruitment and growth rates can affect the length distributions of the catch. Many of these variables can also be monitored through observer data. Gilman *et al.* (2012a) modelled Hawaii longline observer programme data to examine temporal trends in expectile length distributions. Tuna and billfish mean lengths significantly increased owing to entire distributions of length classes having shifted towards larger fish. The authors hypothesized that changes in spatial and seasonal distributions of fishing effort, greater use of wider circle hooks, and possibly increased purse seine selective removals of juvenile tunas contributed to increased selectivity for larger tunas and billfishes (Gilman *et al.*, 2012a). Therefore, interpretations of temporal trends in length frequency distribution and other ecosystem state indicators that are sensitive to fishing pressure based on fishery-dependent data inputs need to explicitly account for variables that significantly explain the length of the catch. Observer programme data enable monitoring many of these variables.

Conclusions on transitioning to meet EBFM ecological data requirements

As authorities have transitioned to accounting for ecosystem considerations in planning and managing marine cross-sectoral activities, including capture fisheries, data requirements have greatly increased. As some fishery management systems struggle to meet basic ecological data requirements of conventional management approaches, the dilemma is whether they should focus on addressing the deficits before expanding monitoring programmes to supply ecological data which support elements of EBFM. Given the substantial data requirements to understand the structure and processes of marine ecosystems, and how fisheries are affecting these systems, monitoring system-wide effects of fisheries to determine if thresholds are being maintained to sustain a desired ecosystem state seems daunting. Despite these seemingly insurmountable constraints, we have demonstrated that some fisheries monitoring frameworks are already collecting observer data that underpin some of the ecological data requirements of EBFM, and provided examples of how small and inexpensive modifications to existing observer programmes can supply additional ecological data needed for some EBFM elements.

Even relatively rudimentary management systems can take steps towards meeting the ecological data requirements of EBFM that are inexpensive and feasible. Many fisheries observer programmes are designed to collect data to monitor bycatch and assess the performance of bycatch management measures. Observer data have provided inputs to PSA ERAs and are being used to identify vulnerable, sensitive habitat. Although there are few examples, observer data are also being to estimate collateral sources of fishing mortality. Observer data are critical inputs to multispecies and ecosystem models and used for monitoring ecosystem indicators, but as with monitoring collateral sources of fishing mortality, there are few management systems employing these EBFM elements.

Many fisheries with relatively rudimentary management systems, with deficits in basic governance elements and limited institutional and financial resources, have onboard observer programmes in place (see section 'Standardized catch and at-vessel mortality rates'). However, there is no at-sea observer coverage for a large proportion of marine capture fisheries, where, for example, over two-thirds of fisheries managed by multilateral RFMOs lack observer coverage (Gilman *et al.*, 2014). For the numerous fisheries management systems lacking onboard observer programmes for fisheries whose ecosystem effects cannot be accurately assessed without data from at-sea monitoring, implementing the opportunities highlighted here to enhance observer programmes to better meet ecological data requirements of EBFM first requires establishing fisheries at-sea observer programmes. This, however, can require substantial investment in building institutional and financial capacity (FAO, 2002).

Across disciplines, including fisheries science, there has been increasing awareness of the benefits of providing for the interoperability of datasets and metadata catalogues (Branton *et al.*, 2006; Gilman, 2011; ISSF, 2012, 2015). This enables the pooling of datasets within and across regions necessary to support large-scale spatial analyses that underpin some EBFM elements, facilitating meaningful comparisons between regions. It also allows training materials and courses for observers to be standardized (Gilman and Hall, 2015). Standardized fisheries observer programme data fields, data collection methods and dataset formatting are needed to enable interoperability of national and regional fisheries observer programme datasets. Metadata catalogues of fisheries observer programme datasets and other datasets of relevance to EBFM enable discovery of datasets relevant for a specific research study. To provide the requisite information to determine if pooling of various databases is suitable, standards for metadata would be useful, such as ensuring fields on data collection methods and estimates of positional error are included (Gilman, 2011; Gilman *et al.*, 2011). In addition to enabling the discovery and pooling of observer datasets, to augment research that informs EBFM, there is also a longstanding need to overcome legal confidentiality measures that typically restrict access to observer datasets.

While the examples drawn here demonstrate how observer programme data can meet the ecological data requirements of EBFM, given that EBFM is only one component of holistic ecosystem-based management, EBFM will be undermined if marine pressures from other than the fishing sector are not also effectively monitored and managed (Link and Browman, 2014). Effective marine ecosystem-based management requires transitioning from piecemeal management of human marine activities by sector, species or issue to cross-sectoral, spatially explicit planning that holistically governs across marine pressures (Pikitch *et al.*, 2004; Crowder and Norse, 2008). 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, climate change, habitat degradation, and the spread of invasive alien species, will increasingly require effective cross-sectoral collaboration (FAO, 2003; Gilman *et al.*, 2014). The fishing industry, unlike the majority of marine industries, relies directly on the production capacity of natural coastal and marine ecosystems and thus has the most at stake.

The data requirements of fisheries monitoring programmes have substantially increased as management authorities have begun to implement EBFM elements, with large dispersion in degrees of success (Pitcher *et al.*, 2009; Gilman *et al.*, 2014). We are cautiously optimistic that the transition to EBFM will continue with improved success as fisheries observer and other monitoring programmes gradually supply more of the ecological data that underpin EBFM.

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