Towards Ecosystem-Based Management: identifying operational food-web indicators for marine ecosystems

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Abstract

Modern approaches to Ecosystem-Based Management and sustainable use of marine resources must account for the myriad impacts (interspecies, human and environmental pressures) affecting marine ecosystems. The network of feeding interactions between co-existing species and populations (food webs) are an important aspect of all marine ecosystems and biodiversity. Here we describe and discuss a quantitative process to evaluate the selection of operational food-web indicators for use in evaluating marine ecosystem status. This process brought together experts in food-web ecology, marine ecology, and resource management, to identify available indicators that can be used to inform marine management. Standard evaluation criteria (availability of data, quality of data, conceptual basis, communicability, relevancy to management) were implemented to identify and evaluate practical food-web indicators ready for operational use and indicators that hold reasonable promise for future use in policy and management. It was recognized that structure and functioning of food webs were the major attributes for which indicators were required and that resilience of food webs was a key aspect of ecosystem behavior and environmental status. Over 60 potential food-web indicators were evaluated and the final selection of operational food-web indicators includes: the primary production required to sustain a fishery, the productivity of seabirds (or similar charismatic megafauna), zooplankton indicators based on community biomass, size structure and productivity, integrated trophic indicators (including mean trophic level, mean size, etc.), and the biomass of trophic guilds. It was emphasised that more efforts should be made to determine suitable reference points in terms of threshold identification for achieving Good Environmental Status, as well as a greater level of integration in the development of indicators for international use.

1. Introduction

Balancing the long-term maintenance of both biological diversity and human well-being is key to sustainable resource management (e.g. Garcia et al., 2015, 2012; Link, 2010; Ostrom et al., 1999; Pretty, 2003; Rockstrom et al., 2009). As such, ecosystem approaches to resource management that address ecological and human interactions are an essential tool for conservation. While there are number of differing definitions for Ecosystem-Based Management (EBM), there is agreement about the need to move towards a more holistic environmental management approach that recognizes the full array of interactions within an ecosystem (Christensen et al., 1996; Link, 2010, 2005; McLeod et al., 2005). Currently, activities stemming from EBM are used to support a number of management actions in multiple ecosystems. In terrestrial habitats, EBM has been applied to management a number of times (e.g. Caldwell, 1970; Slocombe, 1998, 1993) and localized EBM efforts for shallow coastal habitats have also been undertaken (Kershner et al., 2011; Tallis et al., 2010). Globally, a push for EBM in marine ecosystems has been made to balance the trade-offs inherent in managing these complex ecosystems (Link, 2010). For example, EBM is central to NOAA's Integrated Ecosystem Assessments (IEAs: Levin et al., 2009), Fisheries and Oceans Canada has implemented aspects of EBM in the Canada Oceans Act (Curran et al., 2012), there has been a strong shift towards EBM in Australian fisheries driven by a number of policy directions and initiatives (Smith et al., 2007), the European Union's Marine Strategy Framework Directive (MSFD) has developed an overarching plan to reach and maintain Good Environmental Status (GES;Rogers et al., 2010) and EBM is the recognized mechanism to implement the Convention on the Conservation of Antarctic Living Marine Resources (Constable, 2011; Constable et al., 2000). Thus there is a diverse and widespread effort to continue to better manage marine ecosystems by taking into account all pressures, responses and dynamics simultaneously.

Many aspects of ecosystem dynamics are reflected in food webs, the networks formed by the trophic interactions between species in ecological communities. Historically, food web studies developed from simply recording biological data through to a phase where patterns in the data were identified and catalogued. Much work has since focused on interpreting data and patterns, using either phenomenological or mechanistic models in food webs (Rossberg, 2012). Among representations of food webs in the literature are simple directed graphs (topological webs), flow diagrams (energy budgets), representations aggregated by size or trophic level, and complex dynamic models (Link et al., 2005; Piroddi et al., 2015). Depending on the representation, different structural and dynamic properties of food webs emerge from the data. The relationships between these emergent patterns are the subjects of much ongoing research (de Ruiter et al., 2005; Link et al., 2015; Rossberg, 2013).

Ecological indicators are important to EBM because they serve as proxies for several ecological processes (e.g. growth dynamics, energy flow) and are representations of ecosystem state (e.g. biodiversity, resilience). In particular, food-web indicators have become increasingly important as they represent ecosystem services about which policy makers and stakeholders are concerned. The global uses of these indicators to better inform management of living marine resources has continued to increase over time (Coll et al., 2008; Fay et al., 2013; Jackson et al., 2001; Large et al., 2013; Large et al., 2015a; Levin et al., 2009). By addressing much of the inherent complexity of marine ecosystems, food-web indicators are one of the primary interfaces between policy and science. A critical step in the policy process is to agree on food-web indicators that are compelling, intuitive, understandable and defensible to all stakeholders, but also capture key food-web states and processes that underlie critical and complex ecosystem dynamics. Important instances of such indicators are those addressing emergent properties of food webs, which can be predicted without understanding in detail the intricate processes operating in these complex systems. This predictability is reflected in the existence of simplified models or representations of food webs addressing specific emergent properties (ICES, 2013a; Rossberg, 2013). Examples are representations of food webs as food chains passing energy and biomass from lower to higher trophic levels, representations in form of dynamically interacting aggregated groups of species, representations as graphs with arrows (feeding interaction) linking nodes (species), where a small number of top predators are supported by increasing numbers of species at lower-trophic levels (de Ruiter et al., 2005) or complementarily, representation of the distribution of community biomass over body sizes (Kerr and Dickie, 2001). It is important to take into account these properties in selecting food-web indicators to develop pragmatic indicators applicable to describe ecosystems at regional or larger scales.

For operational use, primary requirements are that food-web (or for that matter, any) indicators be sensitive, have a basis in theory and be measurable (Dale and Beyeler, 2001; Kershner et al., 2011; Link, 2010; Rice and Rochet, 2005a). Those indicators that are well studied and linked with emergent properties can address cumulative impacts, integrated dynamic responses, detect indirect and unintended consequences and can help to evaluate trade-offs in managing ecosystems. Globally, a set of best-practices is coalescing around indicator selection: a plethora of indicator selection criteria have been developed that identify key facets of indicators (Fulton et al., 2005; Garcia et al., 2000; Greenstreet and Rogers, 2006; Greenstreet et al., 2011; ICES, 2013a, 2013b; Institute for European Environmental Policy (IEEP), 2005; Link, 2005; Methratta and Link, 2006a; Piet and Jennings, 2005; Rice and Rochet, 2005b; Rochet and Rice, 2005; Shin and Shannon, 2009; Shin et al., 2010a, 2010b)

While there have been some efforts to develop operational ecological indicators to evaluate ecosystem status (ICES, 2015), the task of selecting specific food-web indicators has been difficult for a number of reasons. Food-web ecology is a relatively new area of research (compared to more established community ecology and population ecology) with rapidly emerging information and methods (Link et al., 2015; Longo et al., 2015; Thompson et al., 2012). In light of new methodologies, historical data is often unsuitable to calculate the necessary metrics to use potential food-web indicators for evaluating ecosystem status. Like many other types of ecological indicators, selection of a specific set of food-web indicators can imply that some aspects of marine food webs are valued more than others. Therefore, a well-balanced selection process for indicators is required that encompasses all currently known properties of marine food webs with the necessary data to be confidently used by both management and stakeholders.

This study aims to provide a list of operational food-web indicators that can be used to quantify the emergent properties of food webs in marine ecosystems. The context for this work was the EU's MSFD need to delineate GES with regards to food webs (Descriptor 4; ICES, 2014; Rogers et al., 2010), but was conducted cognizant of broader potential applications to assess ocean status. Here, we develop a strategy using the best available knowledge from scientific experts and a quantitative methodology for evaluating food-web indicators for implementation in EBM. We also discuss the future development of these indicators for practical use as reference points in management.

2. Methods

To address ongoing global requirements (Europe, North America and elsewhere), three objectives related to food-web indicators were explored:

- To determine a defined process for selecting and developing food-web indicators.
- To develop a short list of suggested food-web indicators related to management contexts (EBM) in Europe and globally.

- To establish future direction for operationalizing food-web indicators.

This approach led to a two-part set of efforts to a) identify and evaluate operational food-web indicators that can currently be used and b) identify food-web indicators that hold promise in the future for management, but that require further development. This guidance would allow for increased clarity in selecting food-web indicators coherently within and across regions and lead to more defined response and pressure targets for control rules in EBM. As part of this broader effort, this project was developed as part of the ICES workshop to develop food-web indicators for operational use in EBM (ICES, 2014). The workshop brought together international experts in food webs, marine ecology and management to identify appropriate food-web indicators for current use.

2.1. Food-web indicators

An initial set of 40 food-web indicators were selected from a list of over 60 candidate indicators presented by the workshop experts. Presentations covered all marine functional groups and all attributes of food webs that were considered necessary for a comprehensive evaluation. Duplicate and technically inappropriate indicators were eliminated from the pool of candidate indicators. The remaining 40 food-web indicators were grouped depending on three main food-web attributes which they addressed: functional indicators linked to energy flow, functional indicators linked to ecosystem resilience and structural indicators linked to diversity and 'canary' species (for more detailed descriptions see Appendix A).

2.2. Selection criteria

A list of 5 criteria and 13 sub-criteria (Table 1) was initially synthesized from a set of criteria determined by previous working groups of experts examining ecological indicators (ICES, 2015; Kershner et al. 2011). These criteria were adapted to broadly examine the functionality of the food-web indicators that could be operational within the global context (useful for several countries and regions).

Each indicator was evaluated against the selection criteria and scored as 0, 1 or 2, where 0 = not met, 1 = partly met, and 2 = fully met. A Delphi method was used whereby sets of indicators were scored by small groups based on consensus, following a discussion establishing common understanding of the indicators themselves and how to apply the criteria to the indicators. Each of the 13 sub-criteria was scored equally and no weighting was applied. Scores were presented as percentages of the total score available (maximum score by the number of categories; i.e. 2 x 13 = 26). Indicators were ranked within the agreed attributes of food webs (Functioning – energy flows, Resilience - ability to recover from perturbation, Structure - species organization). Particular issues or concerns with individual scores were highlighted for subsequent discussions. These were then examined so that all scores were adjusted through consensus-based discussions. This process was used to quantify the usefulness of indicators and to aid in the final selection.

2.3. Wider consideration for selecting food-web indicators

In addition to the specific criteria for each food-web indicator, a broader set of features was considered through consensus of the experts involved when evaluating the final recommended suite of indicators. The indicators were categorized into two groups, one set that may be currently implemented and one that holds promise for future development. Key considerations were:

Relative ranks within the major food-web indicator attributes informed the choice of indicators, but were not adhered to in a strictly quantitative manner.

Coverage of all attributes of food webs. To the extent practicable, all three main categories of food-web indicator attributes were represented.

Coverage of all functional groups found within a food web. Recognizing that much indicator development has occurred for upper trophic level contexts, we ensured that lower trophic level taxa were not omitted, even though as a group they may have scored lower than more commonly or routinely monitored upper trophic levels.

Major indicator classes (structure, function and resilience) were as well represented as possible to ensure that important facets of food webs were included.

Current operability was effectively based on an *ad hoc* review (or weighting) of operability issues related to data availability, management relevance and existence of baselines, targets or related reference points, which although were selection criteria, were deemed critical enough to warrant additional consideration.

Links to other indicator uses were considered to ensure that we emphasized food-web indicators that are unique to describing food webs. Other indicator uses include biodiversity, fisheries, eutrophication and sea floor integrity.

3. Results

Within each attribute, indicators tended to cluster into groups with similar underlying ecological theory. When selecting priority indicators for further development it was therefore considered necessary to review the full list of indicators and ensure that those that clustered together, but with lower scores, were also taken into consideration to maintain a diversity of indicator formulations.

The rank scores were obtained from the unweighted sum of all 13 evaluation sub-criteria (Table 2a, b). When the evaluation was re-run separately using only the first six sub-criteria in Table 1 (linked to practical aspects of indicator measurement), and the next seven criteria (linked to aspects of indicator implementation), there was relatively little difference in the final overall outcome. This suggests that the rank scores were robust to variability in criteria selection and were minimally influenced by single criteria evaluations.

3.1. Energy flow indicators

A relatively large number of indicators were identified which had clear links to functional aspects of food webs (Table 2a). Production or biomass ratios for various parts of the food web detect gross structural changes in the energy flow through a food web which may have been caused by, for example, harvesting of key species, or disruption of distributional overlap between predators and prey through climatic factors.

Total Mortality *Z* (Fishing mortality + natural mortality or production to biomass ratio), is commonly used in the ecosystem modelling community (Ecopath with Ecosim: Christensen and Pauly, 2008; Pauly et al., 2000). Despite the relatively high score this was not the most easily interpretable indicator of food web functioning. This was evident in the low score for the communication criteria (Table 2b). Ecosystem exploitation was considered useful to describe the harvesting pattern of exploited ecosystems. It is an indicator of the pressure of the fisheries on the food web.

Primary Production Required (PPR) to sustain a fishery has a solid conceptual basis (Pauly and Christensen, 1995). However, the difficulty of explaining the concept to the lay public contributed to a moderate score for this indicator. Moreover, this indicator does require estimates of transfer efficiency (TE), which is generally assumed to be 10-15% between trophic levels. Note that indicators of transfer efficiency themselves were not selected as indicators for use immediately due to lack of data to systematically estimate TE. Monitoring intermediate marine productivity and chlorophyll *a* fronts by satellite using remote observation was considered effective to estimate indicators of energy-flow in food webs.

Four fairly similar indicators based on trophic level were evaluated (the mean trophic level of the catch, the mean trophic index of the fish community, the mean trophic level of the community and the trophic balance index). Each has a slightly different formulation, but all require good quality and regularly updated data on dietary relationships, time series of survey catch or landings from broad regional seas to avoid local population or fleet effects, and accurate, agreed upon and regularly updated assessments of the trophic levels of the ingested food. Similarly the Trophic Balance Index, describing the fishing pattern of local métiers, can be useful in the context of assessing food web effects of fisheries harvesting, but has limited application for other pressures.

Low scores allocated to indicators such as the disturbance index, loss in production index, mean transfer efficiency and Finn Cycling Index were due to uncertainty over the quality of the technical assessment (data needs and rigor) and the likely ease of implementation. However, some of the indicators may warrant further investigation.

3.2. Resilience indicators

It was interesting to note that the six indicators that had a link to resilience of the food web were generally scored lower than many other indicators (Table 2b). This may be because they are more conceptually complex. It was considered that the top three in this category, the Mean number of trophic links per species, Ecological Network Analysis derived indicators, and the Gini-Simpson dietary diversity index, all held promise as food-web indicators, but the group of experts felt that these would not be recommended as suitable for implementation in the short-term. The conceptual and technical difficulty of measuring food-web resilience and ability to recover from perturbation partly explains the low scores allocated to the assessment criteria in the area of cost-effectiveness of data gathering, although they all have strong support in the literature.

The indicators for this attribute that scored poorly (Herbivory: Detritivory Ratio, Ecological Network Indices, System Omnivory Indices) will take more time to develop. The complexity of their formulation also suggests that, even if further developed, they may be difficult to explain in a management context. More importantly, these indicators need regular diet time series data encompassing the entire food web, which have not been made widely available even to support applied multispecies fishery assessments.

3.3. Structural indicators

Several indicators in this category obtained relatively high scores, suggesting that managers may want to use these indicators to help interpret patterns observed particularly at higher trophic levels. Another important consideration is the role of aggregated sets of structural indicators, such as those related to phytoplankton, zooplankton, forage fish, scavengers and birds, which together have important implications for food-web resilience (e.g. low or high biodiversity) as well as structure of the individual components. Many structural indicators are describing the same ecosystem components in multiple ways (Table 2a, b) and due to the multi-faceted uses of these indicators (in addition to characterizing food webs) the data are likely to be collected and available.

Higher-scoring indicators were those which informed trends in absolute biomass, production, or ratios of both, for a number of guild-level ecosystem components, especially higher predators. For those structural indicators that aggregate across multiple components, it was generally thought preferable to have indicators comprising absolute values rather than ratios, as these data would be necessary anyway to interpret ratio metrics. Some of these abundance-related indicators may be given a higher priority if they are also useful for informing an aspect of food web resilience. For example, both the Gini-Simpson diversity indices for small and large fish and the Species Richness Index were thought to be potentially useful for assessing food web resilience.

3.4. Suggested food-web indicators

The following indicators are the refined set of food-web indicators recommended for current use based on the selection criteria (Table 1) and accounting for the wider considerations in the selection process (Table 2a,b):

Guild level biomass (and production)

Guild-level biomasses and production address structural attributes of food webs, and can also serve as proxies for functioning. It was noted that the typical use of this type of indicator has been for fishes, but if feasible this indicator should include multiple guilds across all trophic levels, such as primary producers, zooplankton, benthos, and charismatic megafauna, beyond just fish or upper tropic levels. The guilds should be determined as appropriate for the taxa in a given regional sea.

Primary Production Required to sustain a fishery (PPR)

This addresses the functioning attribute of food webs and is a measure of the ecological footprint of a fishery. However, this metric can (and often does) integrate a wide range of removals from the food web. Derivatives of this food-web indicator could, where feasible, be contrasted to measures of primary production to ensure it is directly appraised against field data. Satellite imagery makes estimates of primary production widely available (given the usual caveats of remotely sensed data), and typical landings and associated data are also widely available, making PPR more integrative and feasible than is often perceived.

Seabird (charismatic megafauna) productivity

The breeding success of seabirds addresses the structural and functional attribute of a food web and can also serve as a proxy for resilience. Although particular to seabirds, especially breeding success/chicks per pair, it was recognized that seabirds may not be prominent or important in all regional seas. Similar productivity indicator could be calculated for marine mammal taxa (i.e. pup production rates).

Zooplankton size biomass index

This indicator addresses both structural and functional attributes of food webs. Although indicators associated with this taxonomic group were often ranked lower, they represent an important part of the food web - the link between primary production at lower trophic level and upper trophic level consumption and growth.

Integrated trophic indicators (mean TL, mean size)

Trophic indicators address both structural and resilience attributes of food webs. It was critical to include an explicitly integrative measure that provided some view of the overall system and did not focus on only certain facets of it. There are many possible indicators in this category from which to choose, such as mean trophic level, mean, or proportion at size of the community (depending upon abundance) and trophic data availability in a given regional sea.

3.5. Indicators for development

Food-web indicators that were recommended for future development were Ecological Network Analysis indicators, the Gini-Simpson dietary diversity index and condition indicators. These indicators lacked the available data to be considered currently useful for management, but all were determined to be representative of multiple aspects of the food-web (integrated food-web perspective). Some indicators that were suggested to be currently operational (marine trophic level indicators, primary producers and zooplankton indicators) were also thought to require more development to fully meet their potential and range as indicators for food-web and other indicator uses.

4. Discussion

The five food-web indicators recommended from this process cover important facets of food webs, particularly addressing structural, functional and resilient features of marine food webs (Jennings and Collingridge, 2015; Polis and Strong, 1996; Thompson et al., 2012). It is likely that multiple indicators are needed to track the multiple features that comprise food webs and delineation of GES (Large et al., 2015a, 2015b; Mallory et al., 2010; Rice and Rochet, 2005a) of which these five candidates are suitable options. All of the five food-web indicators proposed here are generally applicable in terms of capturing the main facets of food-web dynamics (ICES, 2014; Methratta and Link, 2006b; Shannon et al., 2009) and readily link to known behaviors of food webs. Many of these indicators are broad enough in context to be applied across many marine ecosystems (coastal, temperate, arctic, tropical, etc.; Andrews et al., 2013; Coll and Libralato, 2012; Fulton et al., 2005; Hayes et al., 2015; Parsons et al., 2008; Zador et al., 2014).

Yet even the five proposed indicators may not all have widely and consistently monitored data available to sufficiently calculate the metrics. Although important to track lower-trophic level dynamics and linkages to upper-trophic level taxa, the zooplankton indicator may not have widely collected data nor be as easily interpreted, given the high seasonality of these taxa (Pershing et al., 2005; Stige et al., 2014; Vargas et al., 2006). The integrated trophic indicators hold equal promise, but similarly may not always have measures of trophic level or equivalent (TL; Gaichas et al., 2012; Hornborg et al., 2013; Pranovi et al., 2012; Rossberg et al., 2006). Justifiable assumptions regarding TL, using common databases on trophic ecology of taxa (e.g. fishbase; Froese and Pauly, 2013; Froese, 1992), may provide a means to more readily calculate these indicators in the absence of local trophic data. Size-based integrated indicators are an type of indicator that are less demanding on data and has been found to show clearer responses in food webs (Engelhard et al., 2015; Fung et al., 2013; Greenstreet et al., 2011; Shephard et al., 2011) The salient point is that there are well-studied extant indicators able to track and delineate environmental status in marine food webs (Houle et al., 2012). These were explored in the MSFD GES context (ICES, 2015, 2013b, 2008; Shephard et al., 2014), but are generally applicable for marine conservation considerations.

Regardless of the specific indicator set chosen, a replicable, transparent, defendable and clear process for selection is required (Dale and Beyeler, 2001; Link, 2010; Shin et al., 2010a). The process demonstrated here is broadly applicable in a wide array of conservation situations and it is as important as the outcomes. It is essentially a multi-criteria decision analysis (Mendoza and Martins, 2006), whereupon the selection of indicators is agreed-to before use in tracking ecosystem status. The criteria for indicator assessment used here are sufficiently robust to be applied in a range of situations, with one of the five main criteria specifically evaluating how useful a given indicator is to management. These criteria are converging in the marine management context, but can be readily used in other forms of natural resource management (e.g. terrestrial, estuarine). Due to the well-documented quantitative and qualitative evaluation in the selection process, there is a high level of confidence in the choice of the final set of indicators. This process allows for regular updates and inclusion of novel information (Curtin and Prellezo, 2010; Kershner et al., 2011) while maintaining a record of how selections are made. This process is general enough to be used regardless of the type of ecosystem and conservation issue being considered, as long as the criteria are agreed upon *a priori* (Espinosa-Romero et al., 2011; Martin et al., 2009).

Although similar selection processes have a wide history of use in conservation (Mendoza and Martins, 2006), it could be even more widely and rigorously applied.

Based on the evaluation process, the food-web indicators selected in this study can offer some guidance towards possible management actions. Guild-level biomass reflects measures of biodiversity and structural relationships within ecosystems (Garrison and Link, 2000; Rosenfeld, 2002). It can be an integrative indicator to evaluate the status of a particular guild group in relation to another. For instance, lower numbers of forage fish will have direct (and indirect) impacts on larger predators and seabirds (Cury et al., 2011; Garrison and Link, 2000) or could indicate low levels of primary productivity (Jennings and Collingridge, 2015; Polivina et al., 2001). Either way, management responses to maintain forage fish could be identified based on the information that guild-level indicators provide (Heath et al., 2014). Similarly, integrated trophic indicators can address multiple aspects of structure, function and resilience in ecosystems, where lower mean size or trophic level indicate impacts on large, predatory animals (Methratta and Link, 2006b; Pauly and Watson, 2005; Rosenfeld, 2002). Specific fisheries management actions with respect to changes in these indicators over time could include adjusting harvest control rules for particularly overexploited guilds, but could also include concentrating fishing efforts on lower age or size groups (Anderson et al., 2008). Both higher-trophic (seabird and charismatic megafauna productivity) and lower-trophic indicators (PPR and zooplankton index) are reflective of bottom-up processes viewed from opposing ends of the food web (Cury et al., 2011; Einoder, 2009; Hilting et al., 2013). PPR is an integrative indicator that represents the amount of primary productivity to sustain a fishery, and offers a means to compare energy requirements across different fisheries (Chassot et al., 2010; Gascuel et al., 2005). Seabird productivity is an indicator of food availability (forage fish) and can also be sensitive to contaminants and environmental pollutants (Mallory et al., 2010). Direct management actions to influence these indicators could be either top-down control rules aimed at relieving fishing pressure on lower-trophic species or bottom-up policies directed to improve water quality or habitat, which may also include improved management at land-sea interfaces (Furness and Camphuysen, 1997; Kendall et al., 2010; King and Baker, 2010; Mallory et al., 2010; Teichert et al., 2015). Specific management actions will be dependent on regional circumstances and the responses of the indicators to local pressures, but by using common indicators it will be possible to compare ecosystem status between regions and to help management at all levels (from regional to national to international) and to make effective decisions to improve the world's oceans.

This proposed set of candidate indicators is a start towards operationalizing the delineation of marine ecosystem status, but may require a few further steps before becoming fully operational. Food-web indicators may be interesting scientifically and relevant for management, but if they cannot inform management actions directly they certainly have less utility. Establishing decision criteria that trigger management actions for EBM requires an understanding of how pressure variables influence indicators, as well as the level of a particular pressure at which significant changes in ecosystem structure or function appear (Blanchard et al., 2010; Coll et al., 2010; Groffman et al., 2006; Link, 2010, 2002a; Samhouri et al., 2010). Such thresholds have been explored with a wide range of analytical methods, such as cumulative sums (CUSUM; Hinkley, 1970), sequential t-test (STARS; Rodinov, 2004), empirical fluctuation processes (Zeileis and Kleiber, 2005), and significant zero crossings of piecewise regression models (Chaudihuri and Marron, 1999; Samhouri et al., 2012, 2010; Sonderegger et al., 2008; Toms and Lesperance, 2003; Toms and Villard, 2015) or generalized additive models (Large et al., 2013), all to identify the level of pressure that results in a significant indicator response (Andersen et al., 2009). These univariate relationships are useful for establishing decision criteria (Fay et al., 2013; Large et al., 2013;

Samhouri et al., 2010), however, they do not fully account for multiple pressures that likely interact and occur concurrently. An assessment of ecosystem status based on suites of indicators will be more powerful. Using multiple indicators to evaluate ecosystems will help to avoid the possibility of misinterpretation which can occur when indicators are evaluated in isolation (Coll and Libralato, 2012; Longo et al., 2015; Rice and Rochet, 2005b). Multivariate approaches exist to detect thresholds, including translating indicator response into a surface dependent on multiple pressures (i.e., fishing and environmental pressure; Frederiksen et al., 2007; Large et al., 2015a; Scott et al., 2006), multivariate ordination methods (Baker and King, 2010; King and Baker, 2010) and extensions of regression tree and gradient forest analyses (Baker and Hollowed, 2014; Ellis et al., 2008; Large et al., 2015b; Liaw and Wiener, 2002; Pitcher et al., 2012; Prasad et al., 2006). Understanding how multiple pressure variables concurrently influence ecosystem status, as evinced by thresholds in indicators, will help to further operationalize these indicators as reference points for management.

Both the EU MSFD and US IEA efforts have a similar framework that includes indicators as a critical part of the management decision-making process (Andrews et al., 2013; Levin et al., 2009; Rogers et al., 2010; Shephard et al., 2014). Currently, many of the efforts from both the MSFD and IEA frameworks assess ecosystems by using a suite of indicators. Despite the frequent absence of thresholds of indicators to establish reference points, there is still sufficient information and examples of using such indicators to inform marine ecosystem management advice. Even qualitative and directional features of indicators can and have been used operationally (Andrews et al., 2013; Espinosa-Romero et al., 2011; Foley et al., 2015; Greenstreet et al., 2012; Large et al., 2015a, 2015b; Link et al., 2015; Longo et al., 2015; Samhouri et al., 2012, 2010; Zador et al., 2014). Thus, the monitoring, tracking and presentation of the food-web indicators proposed here can help to operationally delineate GES.

When assessing the status of marine ecosystems, it is important to adequately characterize the food web (Branch et al., 2010; Link, 2002b; Thompson et al., 2012). Certainly there are other aspects of marine ecosystem status, a fact which is explicitly acknowledged in the MSFD. Yet, too often the development of marine indicators neglect to consider food webs (Hayes et al., 2015). Understanding food webs in ecosystems is paramount because they are able to unify ecological sub-disciplines (behavior, dispersal, physiology, thermodynamics etc.) and to examine interactions among guilds (Polis and Strong, 1996; Rossberg, 2013; Thompson et al., 2012). Food webs are able to integrate species-based and functional-based approaches to examine biomass distributions and energetic flows within systems. Another key aspect of ecosystems that is encompassed by food webs is resilience. A resilient system reacts only weakly to pressure, but resilience might be lost with increasing pressures, leading to rapid changes to different states or regimes. Such transition is thus the result of an accumulation of the disturbing effects of pressures (Folke et al., 2004; Gunderson, 2000; Sasaki et al., 2015). Additionally, ecosystems may exhibit legacy effects of earlier pressures (Folke, 2006; Hughes et al., 2005). Despite the difficulty in studying food webs in their entirety (including large data requirements and advanced computational abilities), emergent trends have been established in food-web ecology at both the community (Fredriksen, 2003; Neira et al., 2009) and ecosystem level (Link et al., 2015).

An important aim of EBM is to balance between multiple, often conflicting objectives. How management actions take shape depends on all user groups involved, including stakeholders, indigenous communities, fishers, tourists, NGOs, etc. (Branch et al., 2006; Link, 2010; Marasco et al., 2007). The most successful implementation of EBM will one where user groups are equally engaged, can agree on a set objectives, work towards common economic-social-conservation management goals and ultimately overcome inertia in the decision making process (Arkema et al., 2006; deReynier et al., 2010; Espinosa-Romero et al., 2011; Leslie and McLeod, 2007; Link, 2010; Pitcher et al., 2009; Röckmann et al., 2015; Sandström et al., 2015). The set of indicators proposed in this study is an example of how such information can be used to more fully implement EBM by evaluating one facet of marine ecosystem objectives associated with food webs. More so, the process described here is an important means to explore the tradeoffs not only in selecting these indicators but also the underlying objectives and dynamics that each represents.

Ecological indicators for conservation (including food-web indicators) are useful to summarize complex information concerning marine ecosystem status (Cury and Christensen, 2005; Dulvy et al., 2006; Fulton et al., 2005; Hayes et al., 2015; ICES, 2015; Methratta and Link, 2006b). Clearly defined, consistent metrics at the global scale can provide management in multiple countries with the tools to make EBM more operational (Leslie and McLeod, 2007; Lester et al., 2010; Link, 2010; Link et al., 2011; Smith et al., 2007; Thrush and Dayton, 2010). As management efforts continue to implement EBM to meet conservation objectives, having a suite of indicators, a process to select them and ensuring that they map to clear management needs will remain increasingly important.

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Table 1. Criteria and sub-criteria used in the selection process for operational food-web indicators.

Criteria	Sub-criteria (issues)	Rationale
Availability of underlying data	Existing and ongoing data	Indicators are supported by current or planned monitoring programmes that provide the data necessary to derive the indicator. Ideal monitoring programmes should have a time series capable of supporting baselines and reference point setting. Data should be collected on multiple sequential occasions using consistent protocols.
	Relevant spatial coverage	Data should be derived from an appropriate proportion of the regional sea, at appropriate spatial resolution and sampling design, to which the indicator will apply.
	Relevant temporal coverage	Data should be collected at appropriate sampling frequency and for an appropriate extent of time relevant to the time scale of the process or attribute the indicator describes.
Quality of underlying data	Indicators should be technically rigorous	Indicators should ideally be easily and accurately determined using technically feasible and quality assured methods.
	Reflects changes in ecosystem component that are caused by variation in any specified manageable pressures	The indicator reflects change in the state of an ecological component that is caused by specific significant manageable pressures (e.g. fishing mortality, habitat destruction). The indicator should therefore respond sensitively to particular changes in pressure. The response should based on theoretical or empirical knowledge, thus reflecting the effect of change in pressure on the ecosystem component in question; signal to noise ratio should be high. Ideally the pressure-state relationship should be defined under both the disturbance and recovery phases.
	Magnitude, direction and variance of indicator is estimable	The indicator should exhibit a predictable direction, exhibit clear sense of magnitude of any change, and estimates of precision should allow for detection of trends or distinct locales - requiring that some measure of sampling error or variance estimator is available.
Conceptual basis	Scientific credibility	Scientific, peer-reviewed findings should underpin the assertion that the indicator provides a true representation of process, and variation thereof, for the ecosystem attribute being examined.
	Associated with key processes	The link between the indicator and a process that is essential to food web functioning should be clear and established, based on our current understanding of trophic dynamics.
	Unambiguous	The indicator responds unambiguously to a pressure.
Communication	Comprehensible	Indicators should be interpretable in a way that is easily understandable by policy-makers and other non-scientists (e.g. stakeholders) alike, and the consequences of variation in the indicator should be easy to communicate.
Management	Relevant to management	Indicator links directly to mandated management needs, and idealy to management response. The relationship between human activity and resulting pressure on the ecological component is clearly understood.
	Management thresholds targets are estimable	Clear targets that meet appropriate target criteria (absolute values or trend directions) for the indicator can be specified that reflect management objectives, such as achieving GES. Ideally control rules can be developed.
	Cost-effectiveness	Sampling, measuring, processing, analysing indicator data, and reporting assessment outcomes should make effective use of limited financial resources.

Table 2a. Assessment of food-web indicators for Energy flow indicators against the criteria in Table 1. A maximum score for Availability of data= 6, Quality of data = 6, Conceptual = 6, Communication = 2, Management = 6 (maximum total score = 26). Asterisks (*) denote food-web indicators that were selected for current use.

	Food-web indicator	Availability	Quality	Conceptual	Communication	Management	Score	Percent	Other indicator uses
Energy Flow indicators	*Seabird breeding success	6	3	6	2	5	22	85	Biological diversity
	Mean weight at age of predatory fish species from data	4	5	5	2	5	21	81	Fisheries
	Total mortality Z	4	5	4	1	5	19	73	Fisheries
	Productivity of key predators	6	3	4	1	4	18	69	
	*Primary production required to support fisheries	4	3	6	0	5	18	69	Fisheries, Biological diversity
	Productive pelagic habitat index	6	4	4	1	3	18	69	Eutrophication,Fisheries,Biological diversity
	Ecosystem exploitation	5	3	2	1	5	16	62	Fisheries
	Community condition	3	5	3	2	3	16	62	Fisheries
	*Mean trophic level of catch	4	4	2	1	4	15	58	Fisheries
	*Marine trophic index of the community	4	3	4	1	3	15	58	
	*Mean trophic level of the community	4	3	4	1	3	15	58	
	Disturbance index	4	3	4	1	2	14	54	
	Loss in secondary production index	4	3	4	0	3	14	54	Fisheries
	Cumulative distribution of biomass assessment	4	3	4	0	3	14	54	Fisheries
	Trophic balance index	4	2	3	0	4	13	50	
	Mean transfer efficiency for a given trophic level or size	3	2	4	0	1	10	38	
	Finn cycling index	3	1	4	0	1	9	35	Fisheries

Table 2b. Assessment of food-web indicators for Ecosystem resilience and Structural indicators against the criteria in Table 1. A maximum score for Availability of data= 6, Quality of data = 6, Conceptual = 6, Communication = 2, Management = 6 (maximum total score = 26). Asterisks (*) denote food-web indicators that were selected for current use.

	Food-web indicator	Availability	Quality	Conceptual	Communication	Management	Score	Percent	Other indicator uses
Ecosystem resilience indicators	*Mean trophic links per species	3	2	4	1	2	12	46	Biological diversity
	Ecological network analysis derived indicators	4	1	4	1	2	12	46	
	Gini-Simpson dietary diversity index	3	2	4	1	1	11	42	
	Herbivory to detritivory ratio	3	1	4	1	1	10	38	
	Ecological network indices of ecosystem status and change	4	1	4	0	1	10	38	
	System omnivory index	3	1	2	0	1	7	27	
Structural indicators	Guild surplus production models	6	6	6	1	6	25	96	Fisheries
	Large fish indicator	6	6	5	2	6	25	96	Fisheries
	*Total biomass of small fish	6	5	5	2	5	23	88	Fisheries
	Proportion of predatory fish	6	3	5	2	6	22	85	Biological diversity,Fisheries
	*Mean length of surveyed community	6	6	4	2	4	22	85	Biological diversity,Fisheries
	Pelagic to demersal ratio	6	5	4	2	4	21	81	Fisheries,Eutrophication
	*Biomass of trophic guilds	4	3	5	2	6	20	77	Biological diversity,Fisheries
	Lifeform-based indicator for the pelagic habitat	6	5	4	1	4	20	77	Biological diversity,Eutrophication,Sea-floor integrity
	Region-specific indicators of abundance and spatial distribution	6	3	4	1	5	19	73	Biological diversity,Fisheries,
	Scavenger biomass	3	5	5	1	5	19	73	Biological diversity,Sea-floor integrity
	Geometric mean abundance of seabirds	6	3	5	1	4	19	73	Biological diversity
	Size spectra slope	6	4	4	1	4	19	73	Biological diversity,Fisheries,Sea-floor integrity
	Fish biomass to benthos biomass from models	4	3	4	2	4	17	65	Biological diversity,Fisheries,Sea-floor integrity
	*Zooplankton spatial distribution and total biomass	4	4	3	2	4	17	65	Biological diversity,Eutrophication,
	Zooplankton mean size	4	4	3	2	4	17	65	Biological diversity,Eutrophication

Gini-Simpson diversity index	6	2	2	0	4	14	54	Biological diversity
Species richness index	6	2	2	2	2	14	54	Biological diversity

Appendix A:

Indicators that were examined (details, history, rationale).

Functional Indicators linked to Energy Flow

Seabird breeding success

Many species of seabirds feed on lower trophic level forage species such as krill, squid, and pelagic fish. Seabirds summarize changes in these forage species communities that are often linked to patterns of exploitation (Cury and Christensen, 2005; Cury et al., 2011). Seabird breeding success has been consistently monitored across many ecosystems and provides robust estimates of both forage fish abundance and success of charismatic species (Cury et al., 2011). Seabird breeding success can be a useful indicator, however, it may overlap with other measures of forage fish success.

Productivity (production per unit biomass) of key predators

Metrics characterizing productivity of predators at high trophic levels have been identified by Rogers et al. (2010) as an important class of food-web indicators. They argued that "[t]he abundance of species in the food web will generally be determined by the abundance of suitable prey taxa on which they can feed. Some species, or groups of species, may play a significant part in food web dynamics and so their population status will effectively summarize the main predator-prey processes in the part of the food web that they inhabit." Food quantity or quality is known to affect survival and reproduction of many marine species including birds (Wanless et al., 2005), mammals (Soto et al., 2006) and fishes (Litzow et al., 2006). It has been argued (Boyd and Charles, 2006; Cury et al., 2011; Rogers et al., 2010) that required prey abundance to quantitatively and qualitatively sustain viable populations of predators constitutes a threshold value which can serve as a reference point for productivity based indicators. "Productivity (production per unit biomass) of key species or trophic groups" was listed among the Criteria for GES by the EC (EU, 2010). Among others, it has been implemented in form of the HELCOM (2013) core indicators "Pregnancy rates of marine mammals", "White-tailed eagle productivity", "Abundance of sea trout spawners and parr", and "Abundance of salmon spawners and smolt".

Mean weight at age of predatory fish species from data

Fish weight and condition metrics provide information on state (e.g., food limitation) in an ecosystem. The indicator proposed by Shephard et al. (2014) describes the average "weight anomaly" for the pelagic fish community in a given year, which is the deviation around an observed long-term mean. The youngest and oldest age groups of each stock are excluded to avoid sampling bias. Values are then averaged over all ages for each stock to obtain a mean annual anomaly for that stock. Stock anomalies are then averaged by year to obtain a regional mean weight anomaly for the whole pelagic or predatory fish communities, respectively, where indicator values should fluctuate around zero in the long-term. The comparison between species and stocks can give additional information on whether food becomes limiting in general or whether just some species or trophic guilds are impacted.

Changes in this indicator can be caused by changes in food availability as well as an increase or decrease in predator populations. The demand for food can be also influenced by temperature. Therefore, the indicator should only be interpreted in conjunction with additional information (e.g. biomass of forage fish, benthos, sea temperature, predator abundance, etc.). The indicator will respond predominantly to non-anthropogenic impacts and to a lesser degree to indirect anthropogenic impacts through food limitation.

Total Mortality *Z* (Production:Biomass ratio)

Total mortality has a large effect on both year-to-year survival, long term reference points such as F_{MSY}, and resilience. If mean weight of species in the stock and catch remain constant over time, this indicator is conceptually equivalent to production/biomass. Further, the inverse of total mortality is a direct indicator of longevity, an indicator which is often more readily communicated outside the scientific community. It responds to management through direct fishing mortality and the abundance of predatory fish (ICES, 2013b).

Primary Production Required to support fisheries

The energy contained in solar radiation lead to primary production (PP) by phytoplankton and so fuels marine ecosystems. Subsequently, energy is transferred through food webs by predation and lost through metabolic processes. Ecosystem production results from the conversion of organic matter at each trophic level and is dependent on ecological features such as the number of feeding links, the efficiency of energy transfer from one trophic level to the next, and temperature (Chassot et al., 2010). Production available to fisheries depends upon fishing mortality and targeted trophic levels in the food web. Fisheries focusing only on lower trophic levels may be energetically more efficient than those focused on top predators (Gascuel and Pauly, 2009; Pauly and Christensen, 1995).

Primary Production Required (*PPR*) is the primary production and detritus flows from TL 1 that are required to sustain fisheries (expressed as t/km²/year). This allows the evaluation and comparison of fishing activities across ecosystems. The *PPR* is obtained by calculating the flows backwards, expressed in primary production and detritus equivalents, for all pathways from the caught species down to the primary producers and detritus. The *PPR* increases with fishing intensity. *PPR* has been analyzed also in reference to PP, to reflect a percentage of PP used to sustain catches.

Productive pelagic habitat index (chlorophyll fronts)

Productive fronts (chlorophyll-a fronts) are key large-scale features in marine ecosystems since they last long enough to sustain zooplankton production and are considered among the main vectors of ocean productivity along the food chain (Belkin et al., 2009; Druon et al., 2012, 2011; le Fevre, 1986; Olson et al., 1994; Polivina et al., 2001).

The frequency of chlorophyll-a fronts with an intermediate range of chlorophyll-a content identifies the productive features that attract top-predators, i.e. areas of efficient energy transfer between trophic levels outside of low and high chlorophyll levels (from about 0.1 to 3.0 $mg.m^{-3}$). Indeed, high chlorophyll levels potentially correspond to eutrophic areas where the food chain is disrupted and primary production is not available to upper trophic levels. Nutrient availability, hydrological and atmospheric forcing are captured by this indicator, but it captures in particular the variability of ecosystem productivity available to high tropic levels, independently of fishing pressure.

The indicator of pelagic productivity results from the demonstrated links between top-predators and chlorophyll-a fronts observed for fast-moving predators such as Atlantic bluefin tuna (Druon, 2010; Druon et al., 2011), and fin whale (Druon et al., 2012) and demersal nurseries in the Mediterranean Sea. The generic index of productive pelagic habitats yet requires a formal validation at European scale (https://fishreg.jrc.ec.europa.eu/fish-habitat).

Ecosystem exploitation (fisheries)

This estimates the level of exploitation, integrated over all trophic levels, as the total yield divided by total production for all exploited species. Required data: Yield, biomass and production to biomass ratio for each exploited species.

Community Condition

Community condition is a measure of the overall condition (average weight at length) at the functional group level, and the overall community condition. Condition reflects food availability: fish are heavier for their length when food abundance is plentiful and/or competition for food is low and lighter when food abundance is low and/competition for food is high. It is a reflection of energy flow, food availability and resilience.

Mean trophic level of the catch

Mean trophic level of the catch is one of a suite of trophic level indicators that is based on the average biomass weighted trophic level across all species. Initial work considered the mean trophic level of the catch, based on fishery-dependent catch or landing statistics (Pauly et al., 1998). It describes the average trophic level at which species are removed by the fisheries. As more valuable upper-trophic level fish stocks are depleted, fishers may target lower-value, lower-trophic level fish stocks (Pauly et al., 1998). Recent work suggests that this indicator is a better indicator of fishing pattern and pressure than an indicator of ecosystem state (Shannon et al., 2014).

Marine trophic index of the community (MTI)

The marine trophic index (MTI) (Pauly and Watson, 2005) is another trophic level indicator, calculated with a cut-off point of trophic level greater than 3.25. Originally calculated from fisheries landings data, here it is presented as the MTI of the community, based on scientific survey data, and is considered an indicator of food web functioning (Shannon et al., 2014). It has most commonly been applied to fish (and cephalopods), but could be extended to a wider range of taxa. The marine trophic index of the community, like the mean trophic level of the community (see below), provides a measure of ecosystem integrity and resilience. Declining trophic levels may result in shorter food chains, which may leave ecosystems less able to cope with natural or human-induced change.

Mean trophic level of the community

Average trophic level (TL) obtained from fishery-independent surveys is a commonly used metric that can be used to measure status and trends of ecosystem structure and functioning(Shin et al., 2010). Average TL of the community is expected to decrease in response to fishing, as fisheries tend to target species at upper trophic levels (Pauly et al., 1998). Additionally, fishing can also change the structure of marine food webs by reducing the mean TL and might also influence ecosystem functioning by shortening the length of food chains and releasing predation on lower trophic level organisms (Shin et al., 2010).

Disturbance index

The disturbance index (DI) measures the change in trophic (or size) structure of the ecosystem and is calculated as the sum, across all TLs ≥2 (or size classes), of the absolute difference in the relative biomass (B_{TL}/B_{Total}) within each TL for each year, relative to a reference period (Bundy et al. 2005). The reference period can represent a preferred state of the ecosystem, an ideal state, a theoretical state estimated from an ecosystem model or the beginning to the time period for which there is data. The DI has been shown to respond directly to fishing pressure, but may also be affected by other pressures such as environmental change.

The DI was originally proposed as one of 4 indicators comprising a 4D ecosystem exploitation index (Bundy et al., 2005).

Loss in secondary production index (L index)

The decrease in secondary production was proposed as a proxy for quantifying ecosystem effects of fishing on the basis of a theoretical development and application to a large set of data (Libralato et al., 2008). The L index is calculated by integrating the primary production required to sustain the catches (PPR: Pauly and Christensen, 1995) relative to the primary production (PP) in the ecosystem, taking account of transfer efficiencies (TE, i.e., the efficiency in the transfer of energy from a trophic level to another; Lindeman, 1942) and the trophic level of the catches (TLc; Pauly et al., 1998). Theoretically, these inputs can be combined to measure the loss in secondary production due to fishing (L index) and to evaluate ecosystem effects of all fished species (Libralato et al., 2008).

The application of the L index to a set of well-studied models allowed a probability of being sustainably fished (P_{sust}) to be associated with each L index value, and, by fixing desired sustainability levels (e.g., 75% and 95%) it provide the basis for back-estimating the associated Ecosystem-based Maximum Sustainable Catches (EMSC) (Libralato et al., 2008).

Thus L index is formally defined as an index of ecosystem overfishing and allows application of the index using both landings data and ecosystem models. L index can give rough estimates of overfishing status and management advice measures allowing definition of a region of viable solutions (Cury et al., 2005). L index quantification can be adapted to specific spatial scales (regional spatial assessment) and to large pelagic areas exploiting data from satellite for estimating PP, catches and available data on diets (for TL estimates).

Cumulative distribution of biomass assessment

Accumulation of biomass has been documented for many marine food webs, with the intermediate TLs exhibiting the largest increase in the system cumulative biomass (Gascuel and Pauly, 2009; Link et al., 2009). Changes in this accumulation may reflect shifts in the ecosystem structure and function. According to these observations, from a theoretical point of view, a perturbed ecosystem should lower the stored, cumulative biomass and "stretch out" across TLs. To describe and quantify these changes, the biomass distribution across TLs is fitted to a logistic nonlinear regression model to estimate the main curve parameters: steepness (that is the slope of the tangent passing through the inflection point), inflection TL (that is the projection of the inflection point on the x-axis), inflection CumB (that is the projection of the inflection point on the y-axis), and the basal biomass (that is the y-axis intercept of the fitted curve). Tests, carried out by using both surveys and landings data, showed that the method is robust to possible 'sampling errors' (in terms of TL assignment), sensitive to both environmental and anthropogenic drivers, and when applied to fishery dependent data, responsive (Pranovi et al., 2014, 2012).

Trophic balance index (fishing pattern)

This index measures the evenness (pattern) of exploitation across TLs by comparing their exploitation rates, which are estimated as the sum of yield (Y) divided by the sum of production (P) at each TL. The evenness of exploitation is then given by the coefficient of variation of all Y/P. Required data: Yield, biomass and P/B for each species in the yield.

Mean transfer efficiency for a given TL or size

The transfer efficiency (TE_{TL}) is defined as the fraction of production that is passed from one integer trophic level to the next (Lindeman, 1942; Pauly and Christensen, 1995). It is thus quantifiable as the ratio between the production of the trophic level (TL) and the production at the precedent trophic level (TL-1). Several studies have estimated the pattern of TE by different trophic level after Lindeman's work (Burns, 1989; Lindeman, 1942). It has been used as a diagnostic indicator in some cases (e.g. Libralato et al., 2004) but in most instances the ecosystem average is used as an integrated summary statistic.

Finn Cycling Index

The Finn's cycling index (FCI: Finn, 1976) is the proportion of the total sum of flows in the food web that is recycled in the system. It is measured as the proportion of the total flow that is flowing within circular pathways. Recycling is considered to be an indicator of an ecosystem's ability to maintain its structure and integrity through positive feedback and is used as an indicator of stress and maturity (Christensen, 1995; Monaco and Ulankowicz, 1997; Ulanowicz, 1992; Vasconcellos et al., 1997). FCI is an indicator of the recovery time of an ecosystem through development of routes to conserve nutrients. A high FCI would mean the system would recover faster from a perturbation, whereas a system would be expected to take longer to recover (lower FCI) when it is in a more degraded state.

Functional Indicators linked to ecosystem resilience

Mean trophic links per species

The mean number of trophic links per species reflects how connected a food web is and, potentially, how stable a food web may be (Link, 2002; Link, 2005; Methratta and Link, 2006). Changes to this indicator reflect notable differences in the structure and dynamics of a food web. As an understanding of temporal and spatial characteristics of marine trophic interactions it may not be entirely complete. This index should be used only as a tool to invoke further precautionary action (Link, 2005).

Ecological Network Analysis derived indicators (overall mean Transfer Efficiency)

The mean transfer efficiency (TE_m) for the food web is calculated as the geometric mean of transfer efficiencies for each of the integer trophic levels II to IV from models (Christensen and Pauly, 2008; Christensen et al., 2009). It is a variant of the mean transfer efficiency discussed above. There have been attempts to estimate average TE also on the basis of catches over trophic levels on the assumption that fisheries were in balance for some periods (Pauly and Palomares, 2005) – which would provide a fishing pressure indicator. Average transfer efficiency by ecosystem type based on model outputs have shown some variability across ecosystem types (Libralato et al., 2008) and other pressures as shown in Heymans et al. (2012). The indicator has been proposed as a descriptor of ecosystem health in lakes (Xu and Mage, 2001).

Gini-Simpson dietary diversity index

The Gini-Simpson dietary diversity index is defined as the average, over a representative sample of consumer species, of the Gini-Simpson diversity of the contributions of resource species to consumer diets, by volume or biomass (ICES, 2013b; Rossberg, 2013; Rossberg et al., 2011). It can be determined from stomach-content data. The metric attains values between 0 and 1, with 0 implying no diversity and 1 high diverse. In practice it is computed as

$$1-\frac{\Sigma_{ij}[(p]_{ij})^2}{\Sigma_{ij}p_{ij}} \quad p_{ij}\ i\ j \quad D_{diet}=1-\Sigma_{ij}\frac{[p\text{¿¿}ij]^2}{\Sigma_{ij}p_{ij}}\text{¿},$$

where p_{ij} is the proportional contribution of species i to the diet of j, and the sums run over all diet items resolved to species level (Rossberg, 2013; Rossberg et al., 2011). The indicator may be applied to any component of the ecosystem for which diet data is available, but has so far been computed only for fish (Rossberg et al., 2011). A target for the metric near 0.5 has been proposed (Rossberg, 2013; Rossberg et al., 2011) based on theory and observation data. The indicator may respond to pressures (e.g. Rossberg et al., 2011).

Herbivory : detritivory ratio

This indicator, proposed by (Ulanowicz (1992), is the ratio of the values of the detritivory flow (from detritus to level II) divided by the value for the herbivory flow (from primary producers to level II). It is sometimes presented as H/D (or abbreviated HDR). This indicator was inspired by Lindeman (1942) when he referred to the role of saprophageous organisms and heterotrophic bacteria. This ratio has already been tested as a candidate for defining functional indicators of the food web, but results seem to be case sensitive. For example, Ulanowicz (1992) observed a higher H/D ratio in disturbed situations whereas Dame and Christina (2007) observed exactly the opposite trend. Then the disturbed situation showed a shift to a more detritus-based food web.

Ecological Network indices of ecosystem status and change (Ulankowicz)

Redundancy *(R)* (Monaco and Ulankowicz, 1997) indicates the system’s energy in reserve it describes the distribution of energy flow among the ecosystem pathways, and is an indicator of change in the degrees of freedom of the system (Heymans et al., 2007). Based on the description of *R* by (Ulanowicz, 2004), who suggested that, “… it strongly ties to the effective multiplicity of parallel flows by which medium passes between any two arbitrary system components”. Redundancy is linked by Christensen and Pauly (2008) with system stability and proposed by Heymans et al. (2007) as an index of food-web resilience. According to Bondavalli et al. (2000) high redundancy signifies that either the system is maintaining a higher number of parallel trophic channels in order to compensate for the effects of environmental stress, or that it is well along its way to maturity. With regard to overall performance and robustness, ecosystem level indicators based on ecological network analysis and food-web analysis are informative on intermediate and long time-scales (Cury et al., 2005; IEEP, 2005; Moloney et al., 2005). But they are difficult to use in annual updates and operational approach, and may be more difficult for stakeholders to understand (IEEP, 2005). In addition, use of food-web models and the ecological network analysis approach to explore different management scenarios, through simulation of fishery and nutrient management, could deliver integrated views at ecosystem level.

System omnivory index

The system omnivory index (SOI) measures the distribution of feeding interactions among trophic levels of food webs, thus SOI evaluates the complexity and connectivity of food webs. It has been associated to ecosystem ability to recover from perturbations (Christensen, 1995; Libralato, 2008). Given a food web with n elements, the SOI is calculated as the weighted average of the elements' omnivory, the latter calculated as the omnivory index (OI). The OI of each consumer element i with trophic level TL_i is quantified as the variance of the trophic levels of its preys (TL_j) (Williams and Martinez, 2004). The SOI of a given trophic network is quantified as the weighted average of the OI of all consumers of the network, where the weighting factors are taken as the logarithm of each consumer food intake (Q_i) (Christensen and Pauly, 1993). This allows for accounting of the different strengths of consumer interactions and the logarithm is used on the observation that consumption is approximately log–normally distributed within systems (Christensen and Pauly, 1993).

The topological configuration of links and their weights affect SOI, but it is quite robust to the number of nodes in the web (Libralato, 2008). Comparison of stability and complexity indices including SOI for coastal marine food webs highlighted positive correlation between SOI, magnitude of change and recovery time, thus suggesting that SOI is inversely related to stability of marine ecosystems (Perez-Espana and Arreguin-Sanchez, 1999). Moreover, application of SOI and other ecological indicators, on the basis of outputs of protected and fished marine food webs standardized by number of elements, suggests that SOI is sensitive to fishing (Libralato et al., 2010).

Structural Indicators linked to diversity and ‘canary’ species

Guild Surplus Production models

Guild Surplus Production is tracked in the annual Ecosystem Assessment document for the North Pacific Fisheries Management Council (Zador, 2013). Species are grouped into functional guilds based on feeding and life history studies. Survey and catch time series for each species are used to calculate the surplus production for each guild. To use as a catch limit, in addition to a single-species limit for each managed stock, the sum of quotas for each guild cannot exceed the MSY for the guild as defined by a standard surplus production model. Per-species reductions to meet this overall limit are not proscribed by this index; reductions can be made for stakeholder or economic reasons. For Bering Sea (ecosystem-wide) indicator example, see Meuter and Megrey (2006). The indicator uses is based on survey biomass and catch of the species within each guild.

Total biomass of small fish

This indicator uses survey catch biomass of predefined small (pelagic) fish to assess exploitation levels of commercial stocks. The amount of energy transferred from zooplankton to higher trophic levels by pelagic fish is ultimately limited by the biomass of pelagic fish available.

Shephard et al. (2014) therefore suggest that both the biomass of individual stocks should be above precautionary reference points on average and the total stock biomass of all pelagic fish together should be above a joint community reference point. In practice, the community reference point is always reached when all individual stocks are above precautionary reference levels. However, in the case where one or more stocks are substantially below single stock reference points, additional care should be taken in the exploitation of the remaining stocks in the area.

Proportion of Predatory Fish

Predatory fish species are defined as all surveyed fish species that are not largely planktivorous (i.e. phytoplankton and zooplankton feeders should be excluded: Shin et al., 2010). A fish species is classified as predatory if it is piscivorous, or if it feeds on invertebrates that are larger than the macrozooplankton category (0.2 cm). Detritivores should not be classified as predatory fish. This indicator captures changes in the trophic structure and changes in the functional diversity of fish in the ecosystem. It is sensitive to fishing pressure, but since it is a ratio, it will also be subject to changes in non-predatory fish, whose biomass may vary for other reasons (i.e. environmental driver: Bundy et al., 2010).

This indicator is calculated as the biomass of predatory fish surveyed / biomass surveyed, and the data required are trawl survey data and food habits data (if not available locally, from information in the literature or from comparable systems).

Pelagic to demersal ratio

The ratio of pelagic to demersal fish (P:D ratio) obtained from fishery-dependent or -independent surveys is a commonly used metric that describes trophic energy flow and community structure (de Leiva Moreno et al., 2000; Link, 2005; Rochet and Trenkel, 2003). Changes in P:D ratio have been linked to anthropogenic pressures such as fishing and eutrophication. Targeted fishing can result in notable shifts in this indicator, however, changes may be not be entirely clear, as an increase in the P:D ratio could be caused by an increase in pelagic fish or a relative decrease in demersal fish. As an indicator of food web properties, P:D ratio may overlap with other large and/or forage fish indicators, but does capture important trophic relationships.

Biomass of trophic guilds

Biomass of trophic guilds is a measure of ecosystem structure, estimated as the aggregate biomass of each trophic guild. Individually they provide a measure of the change in biomass of trophic guilds. Collectively, they provide a measure of change in overall structure. It can be applied to all marine species if the information is available, based on survey data or model results. Work to date has largely focused on fish trophic guilds (Rochet et al., 2013; Shackell et al., 2012), but could be extended to invertebrates birds, and marine mammals. Measures of functional diversity could also be developed using these data. Data sources can be from research surveys or models.

Lifeform-based indicator for the pelagic habitat

Ecosystem health theory (reviewed by Tett et al., 2013) suggests that ecosystem resilience, and the sustainability of services, depends *inter alia* on the abundance and relationships of non-substitutable `functional groups' or 'lifeforms'. The abundances and trophic structural relationships of phytoplankters, and their protozoan and mesozooplankton consumers, change seasonally. The Plankton Index (Pi) method takes account of such seasonality and requires the plotting of log-transformed lifeform abundances, based on at least monthly samples, in sets of 2-D state spaces (Gowen et al., 2011). These plots (Tett et al., 2008) often suggest a fuzzy doughnut. Using data for a reference period, an envelope can be drawn to include a fixed proportion (usually 90%) of points in this doughnut. Data from other years can be plotted against this envelope; the Pi[j,t] value (for lifeform pair j and year t) is defined as the proportion of new points that fall inside the envelope. For a given value of t, values of Pi for different lifeform pairs can be averaged. A UK project has identified sets of lifeform pairs that may serve for assessment of environmental status. The lifeform pairs relevant to Food Webs are: (i) chlorophyll concentration and mesozooplankton abundance; (ii) phytoplankton >= 20 µm abundance and phytoplankton < 20 µm abundance; (iii) [adult] copepods >= 2 mm abundance and [adult] copepods < 2 mm abundance. Reference conditions for any of the Pi are expected to be dependent on ecohydrodynamic (EHD) conditions (van Leeuwen et al., 2015). The UK is currently seeking EHD-specific references at sites in the Celtic or Greater North Sea MSFD ecoregions that are, according to expert judgement, in GES. Meanwhile, time-series of Pi will be generated from conditions observed during an agreed (but arbitrary) period of 3 years, and the time series will be assessed for (a) significant trends, and (b) significant correlation with relevant pressures.

Region-specific indicators of abundance & spatial distribution,

Indicators can be selected to track the abundance and spatial distribution of major species which represent key community and or/ecosystem properties. Ideally, species representing different communities or habitats (benthos, plankton, fish, top predators) should be selected, in this way covering a large part of the ecosystem. As ecosystems are typically characterized by few strong links and many weak links among species or trophic levels, one (or few) indicator populations can describe broader ecosystem state and/or human perturbation. Criteria in the MSFD for selecting the groups/species that could be included in this category are those with fast turnover rates, groups/species that are targeted by fisheries, the habitat-defining groups/species, those at the top of the food web, and those tightly linked to other trophic levels (Rogers et al., 2010).

Fish biomass : benthos biomass from models

Ratios are used to measure changes in community structure indicating the distribution of energy in the ecosystem. They are a supplement to biomass indicators and have the advantage that they do not reflect general increases or decreases in biomass in all components but only changes in the relative importance between the two groups. Hence, the ratio pelagic biomass : demersal biomass represents the balance between pelagics and demersals whereas the fish/benthos ratio reflects the proportion of the biomass which is diverted to benthos, including detritivores. The indicator captures changes in the trophic structure and changes in the functional diversity of the ecosystem. It is sensitive to fishing pressure, but since it is a ratio, it will also be subject to changes in non-manageable benthos, whose biomass may vary for other reasons (e.g. environmental driver, Bundy et al., 2010). Data sources can be research surveys (mainly nekton) or models (often benthos, since this is often not surveyed on appropriate spatial and temporal scales).

Zooplankton spatial distribution and total biomass

This indicator, which describes the distribution of zooplankton, is still at the developmental stage with methods, threshold and target values to be developed. The reasoning for this indicator is that zooplankton constitutes an important link between primary producers and higher trophic levels in the food web. Zooplankton plays an important role in the energy transfer and nutrient cycling in the food web. Changes of the composition of the zooplankton community are coupled to environmental changes and can respond quickly to ecosystem changes. Zooplankton biomass and abundance can e.g. respond to invasive species and local oil spills.

Scavenger biomass

Fishery discards provide food subsidies that help maintain fish and seabird populations and may allow some of these populations to be more abundant than they would naturally be (e.g. Link and Almeida, 2002; Polis and Strong, 1996). Surveys of non-targeted scavenger biomass or abundance may provide an index of disturbance (Link and Almeida, 2002; Methratta and Link, 2006b). Additionally, some scavenger species might be viewed as “canary” or “iconic” species that can be used as an early warning of disturbance or excessive fishing pressure.

Geometric mean abundance of seabirds

The Geometric Mean Abundance of Seabirds is computed in regular intervals (e.g. yearly) as the geometric mean of the population sizes (e.g. numbers of individuals or breeding pairs) of those seabirds in the assessment region for which population time series are available, normalised such that the indicator value at the beginning of the indicator time series is one. The indicator is designed after the Living Planet Index (LPI: Loh et al., 2005), which now underlies Aichi Target 5 of the Convention for Biological Diversity. Modern indicator protocols take into account that species may enter or leave the set of species for which time series are available, and that population sizes at low abundances become uncertain. Methods to compute indicator confidence intervals have been developed (Buckland et al., 2011; Loh et al., 2005). By its definition, the proportional rate of change of the indicator equals the average population growth rate of all populations contained in the indicator (here seabirds). Under conditions where populations fluctuate and turn over but overall biodiversity does not change, the indicator is expected not to deviate significantly from one. A steady decline of geometric mean abundance signals biodiversity loss. Seabird populations are known to be highly sensitive to food availability (Cury et al., 2011), and their differentiation of foraging niches (Fasola et al., 1989) is evidence of competition for food among them. Competitive exclusion resulting from loss of biodiversity among their marine resources (e.g. forage fish), or even at lower trophic levels (Rossberg, 2013), can be expected to induce the slow decline of seabird diversity to which this indicator is designed to be sensitive. Geometric mean abundance of seabirds is therefore sensitive to a collapse of the pyramidal distribution of species over trophic levels in food webs (de Ruiter et al., 2005).

Gini-Simpson diversity index (species dominance) of large and small fish by biomass

It is incompatible with GES to bring the foodweb into a state where only a few (large) predator or prey species dominate the system when the biomass of predators and prey was distributed more evenly in the system during the reference period. Species richness may be inadequate as an indicator as it often takes a long time to completely lose a species, while management should be informed and act earlier. The Gini-Simpson index (1-D) applied to the predator and/or prey community provides the possibility to detect unwanted changes in diversity. Simpson's Diversity Index is a measure of diversity which takes into account the number of species present, as well as the relative abundance of each species. As species richness and evenness increase, so does diversity (ICES, 2013b).

Species Richness Index

Species richness measures the number of species within a community. A well-structured and functioning ecosystem will generally have many species (Fung et al., 2015); as a side effect of fishing, species richness may decrease (Rice, 2003, 2000). However, as a food web indicator species richness may provide ambiguous information, since multiple community configurations may produce similar values, as shown by Gislason and Rice (1998) and Rice and Gislason (1996). In these studies the index was calculated as the number of species in any year whose numerical abundance or biomass was larger than some percentage of their value in a reference year. The IUCN Red List criterion of 20% was used as the reference value. Required data: species or functional group P/B, and species or functional group biomass/abundance to compare to reference points.

The Large Fish Indicator (LFI)

The Large Fish Indicator (LFI) is defined as the proportion by weight of large fish in the sample of a specified survey (Greenstreet et al., 2011), where large fish are defined as those longer than a threshold length L_{th}, a region-specific value. The threshold value is chosen such as to optimize the responsiveness of the indicator to fishing pressure, as determined from historic data (Shephard et al., 2011). The LFI takes no account of species identity, only of individual sizes. However, it was shown to reflect mostly the proportion (by weight) of large-bodied species in communities (Shephard et al., 2012). Large-bodied species tend to be more vulnerable to fishing, which is why the LFI is sensitive (Engelhard et al., 2015; Greenstreet et al., 2011; ICES, 2011; Shephard et al., 2013) and specific (Houle et al., 2012) to fishing pressure. Furthermore, by expressing the indicator in terms of proportions by weight, and not by numbers, and through judicious choice of the appropriate length threshold to define large fish, the indicator can be desensitized to variation in the abundance of small fish. The influence of environmentally driven recruitment events on indicator values can therefore be minimized (Greenstreet et al., 2011). Food-web models (Fung et al., 2013; Shephard et al., 2013) and data (Fung et al., 2012) suggest that recovery of the indicator from pressures can be slow (decadal scale). The LFI, as an OSPAR EcoQO for the North Sea, is fully operational. It was named as an indicator for food-web GES (EU, 2010), and has been chosen as a common food-web indicator by HELCOM and OSPAR (in some OSPAR Subregions as a priority candidate indicator).

Mean length of surveyed community

Mean length (ML) of all species caught in a survey, whether fishery-independent, fishery-dependent, or based on landings, can be a useful and simple indicator to evaluate the overall effects of fishing on an ecosystem (Dulvy et al., 2004; Nicholson and Jennings, 2004; Rochet and Trenkel, 2003; Shin et al., 2005). ML quantifies relative abundances of large and small individuals and describes the size distribution of a community (Shin et al., 2005). It is relatively responsive to key pressures (Link, 2010; Pauly et al., 1998). ML is considered measurable and generally robust, however, the direction of response may be caused by increasing stocks of large fish or decreasing in stocks of small fish, leading to potential ambiguity. Whilst the metric is sensitive to fishing pressure, it can also be strongly influenced by environmentally driven recruitment events that introduce large numbers of small fish into the community (Badalmenti et al., 2002; Lekve et al., 2002; Wilderbuer et al., 2002).

Size spectrum slope

Various measures of the change in size can be a useful indicator to describe composition of communities (Nicholson and Jennings, 2004). Size spectrum slope measures the relationships between the biomass (y) of individuals within a body size class and body size (x), both normally plotted on logarithmic scales. Frequently a logarithmic transformation is applied to body size, particularly when weight classes are used. When applied to fish communities, the slope of the relationship becomes increasingly negative in response to fishing pressure; fisheries reduce the abundance of large fish, through the direct effect of fishing and, as a consequence of reduced predation pressure from large fish, the abundance of small fish increases (Daan et al., 2005; Gislason and Rice, 1998; Nicholson and Jennings, 2004; Rice and Gislason, 1996). The size spectrum slope is considered measurable and robust. However, the direction of the response may not be entirely clear (Trenkel and Rochet, 2003), as the steepening of the slope could indicate a decrease of large fish or an increase of small fish. The slope is particularly sensitive to changes in the abundance of small fish, which markedly affect the intercept of the regression line, as such the size spectrum slope can be influenced by environmentally driven recruitment events (Badalmenti et al., 2002; Lekve et al., 2002; Wilderbuer et al., 2002).

Zooplankton Size-Biomass index

This is a zooplankton indicator reflecting both mean individual size and total biomass of the zooplankton community. The indicator represents food web capacity to sustain fish feeding conditions and grazing on primary producers. The rationale is that both mean body size in the community and total community biomass are positively related to fish feeding conditions, whereas total biomass alone is just representative of grazing pressure and trophic transfer efficiency (Fuchs and Franks, 2010). The effects of zooplankton community structure on energy transfer and food web resilience have been demonstrated in both freshwater and marine systems (Jeppesen et al., 2011; Kane et al., 2009; Lougheed and Chow-Fraser, 2002). The index is currently considered as a core indicator for the Baltic Sea (HELCOM, 2013, 2012). In semi-enclosed seas, such as the Baltic Sea, with strong salinity and temperature gradients, no single zooplankton group can adequately reflect community properties (Remm 1984), hence the need for this two-dimensional index. The index value decreases with increasing fishing pressure. Protocols for indicator assessment have been developed by HELCOM Zooplankton Expert Network (ZEN) using nine long-term monitoring datasets in the Baltic Sea (HELCOM, 2013, 2012). In all datasets, the indicator was found to predict deviations from GES conditions. Determination of GES boundaries for the indicator is straightforward and based on the regional basin-specific Environmental Quality Ratios for chlorophyll accepted within Water Framework Directive and weight-at-age for zoo-planktivorous fish (HELCOM, 2013, 2012).