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REVIEW

Role of trophic models and indicators in current marine fisheries management

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ABSTRACT: The previous decade has witnessed a flourishing of studies on how fisheries and marine food webs interact, and how trophic models and indicators can be used for assessment and management purposes. Acknowledging the importance of complex interactions among species, fishermen and the environment has led to a shift from single species to an ecosystem-wide approach in the science supporting fisheries management (e.g. Johannesburg Declaration, Magnuson-Stevens Act). Moreover, fisheries managers today acknowledge that fishing activities are linked to a range of societal benefits and services, and their work is necessarily a multi-objective practice (i.e. ecosystem-based management). We argue that the knowledge accumulated thus far points to trophodynamic models and indicators as key tools for such multi-dimensional assessments. Nevertheless, trophodynamic approaches are still underutilised in fisheries management. More specifically, most management decisions continue to rely on single species and sector-specific models. Here we review examples of applications of trophodynamic indicators within fisheries assessments in well-studied ecosystems, and discuss progress made (as well as lack thereof) towards increased integration of these metrics into marine resource management. Having clarified how trophic indicators fit within current policy and management contexts, we propose ways forward to increase their use in view of future management challenges.

KEY WORDS: Trophic dynamics · Ecological indicators · Ecological network analysis · Ecosystem-based management · Integrated ecosystem assessments · Ecopath with Ecosim

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INTRODUCTION

Trophic interactions play an important role in ecosystem functioning and structure, as proposed early on by pioneers such as Elton (1927) and Lindeman (1942) and later supported by a vast amount of research (see e.g. Duffy et al. 2007). In fisheries management, the need to take into consideration multi-species interactions in order to appropriately manage marine resources was explicitly stated in the FAO Code of Conduct for Responsible Fisheries (FAO 1995). However, the importance of trophic dynamics only became broadly acknowledged in the past decade, urging a shift towards ecosystem-based man-

agement (EBM) for marine fisheries worldwide (Pikitch et al. 2004). The work by Pauly et al. (1998) on 'fishing down marine food webs', i.e. sequential depletion of top predators which forces fishermen to progressively target species farther down in the food web, was pivotal in starting a discussion within the scientific community on the need to address long-term changes in ecosystem structure and function linked to fishing activities. This idea subsequently stimulated a rich range of research from critical revisions of underlying data analysis and interpretation (e.g. Caddy et al. 1998, Essington et al. 2006, Branch et al. 2010, Sethi et al. 2010, Pinsky et al. 2011, Hornborg et al. 2013), to explorations of when and how the concept may be usefully applied (e.g. Libralato et al. 2004, Pauly & Watson 2005, Stergiou & Tsikliras 2011, Pauly et al. 2013, Kleisner et al. 2014, Moutopoulos et al. 2014, Shannon et al. 2014a). Extensive and influential research in the fields of ecosystem properties, fisheries production and sustainable exploitation levels has greatly expanded our understanding of the complex dynamics underlying fisheries (e.g. Hunt & McKinnell 2006, Coll et al. 2008, Chassot et al. 2010, Cury et al. 2011, Smith et al. 2011), beyond the initial bioeconomic single-species models that had dominated fisheries research for many decades (Schaefer 1957).

As this ecosystem-wide approach to marine resources management increasingly became a stated intent of various policy strategies (e.g. FAO Code of Conduct for Responsible Fisheries [FAO 1995], Reykjavik 2001 and Johannesburg 2002 declarations, newly reformed Common Fisheries Policy in the EU in 2014, and the reauthorized US Magnuson-Stevens Fishery Conservation and Management Act of 2006), scientists focussed on developing assessment tools and metrics that could account for feedback effects between fishing dynamics and multi-species dynamics (see e.g. Cury et al. 2005, IEEP 2005, Samhouri et al. 2009, Shin & Shannon 2010). Hundreds of trophodynamic indicators have been proposed since, where we refer to trophodynamics as dynamic multi-species interactions, such as predation and competition, that, by influencing productivity and energy transfer, regulate system functioning (Lindeman 1942), and trophodynamic models and indicators as tools that capture such dynamics (e.g. Walters et al. 1997, Cury et al. 2005). At the same time, in order to sort through this growing information, a flurry of research focussed on setting and implementing rules to choose, combine and interpret indicators, including considerations on data availability, responsiveness and clear links to drivers and to management objectives (Rice 2000, IEEP

2005, Link 2005, Rice & Rochet 2005, Rochet et al. 2005, Shin & Shannon 2010). Later efforts expanded to developing frameworks to guide the full process from goal-setting, to reference point choice, to indicator use (Sainsbury et al. 2000, Levin et al. 2009, Samhouri et al. 2010, 2012), to communicating results to stakeholders (Cash et al. 2003, Rice & Rochet 2005, Shannon et al. 2014b). Moreover, trophodynamic models were used to identify best measures for fisheries management (e.g. Fulton et al. 2005), to challenge reference values determined with single-species assessments (Walters et al. 2008) and to identify key ecological thresholds (e.g. Plagányi et al. 2014a).

Based on these developments, one might have expected trophodynamic indicators and models to gain an increasingly central role in fisheries management. So far, they have mainly been used for reporting on the state of the ecosystem, such as the Marine Trophic Index (Pauly & Watson 2005), one of the Convention on Biological Diversity target indicators (Biodiversity Indicators Partnership 2010) or the EU Marine Strategy Framework Directive (MSFD) descriptors for Good Environmental Status (GES) of marine food webs (EU-COM 2008). However, such applications do not have well-defined links to management strategies, let alone specific management actions.

We argue that trophodynamic measures and models are vital in order to support place-based, integrated ecosystem assessments of marine fisheries (Billick & Pierce 2011), particularly in the current policy context where EBM is a growing priority. In the following sections, we provide examples of lessons learned from trophodynamic models of well-studied ecosystems where current policy explicitly mandates EBM. These are intended as illustrative examples rather than as an exhaustive list. We then discuss possible reasons hindering practical applications of trophodynamic metrics for tactical decisions of fisheries management authorities, and provide a perspective on future challenges and opportunities, including the integration with social dimensions, for their widespread use.

LESSONS LEARNED FROM TROPHODYNAMICS IN EBM: ILLUSTRATIVE EXAMPLES

Ecological network analysis and the Baltic fisheries

One way to describe multispecies systems and their energy flow is through ecological network analysis (ENA). ENA was first proposed by Ulanowicz (1986) to study trophic interactions and ecosystem function-

ing while capturing the structural and dynamic complexity of marine ecosystems (Mageau et al. 1995, Ulanowicz 2004, Christensen 2013). The approach was developed further also owing to its application to outputs of linear inverse models (de Laender et al. 2010), and was later incorporated in the Ecopath with Ecosim (EwE) software (Christensen & Walters 2004).

The ecosystem metrics used in ENA are calculated on the basis of flows and biomasses usually estimated from modelled food web dynamics (Heymans et al. 2007). The indicators, based on well-grounded (yet not straightforward) theoretical arguments (Ulanowicz 1986), describe ecosystem properties in terms of resilience, maturity and ecosystem structure, to help interpret the effects of fisheries and other sources of disturbance (Christensen 1995, Shannon et al. 2009, Niquil et al. 2014). For example, in the Baltic Sea, ENA has been used to explore the structure of energy flow pathways mediated by trophic interactions (Wulff et al. 1989, Christensen 2013) and, recently, to investigate changes in the Baltic Sea food web (Tomczak et al. 2009). Fisheries-related indices used include primary production required (PPR; Pauly & Christensen 1995) to sustain catch per primary production unit (PP), or PPR/PP, as a measure of energy efficiency (Tudela et al. 2005), and mean trophic level of catch (mTLc) as a measure of food-web integrity (Pauly et al. 1998). With this approach, temporal changes in the fisheries effect on ecosystems have been shown using the Central Baltic Sea food web model, documenting changes in cod, sprat and herring stocks as well as in the species composition of the zooplankton assemblage (Tomczak et al. 2013).

A climate-driven shift in system primary productivity in the 1990s, well documented by Möllmann et al. (2009), triggered a shift in fishing pressure from Baltic cod, up until then the main commercially exploited species, to sprat, resulting in an increase in total catches. This caused a shift from high mTLc associated with relatively low PPR/PP, to high catches at low mTLc in combination with high PPR/PP, despite an increase in PP (Fig. 1). More in detail, Tomczak et al. (2012) found that as a result of the substantial increase of sprat catches, as well as catches in general, cod diet switched to higher trophic level prey, thus decreasing the contribution of the benthic component to the food web. The redirected energy flows from benthic to pelagic pathways resulted in a less efficient food web with lower redundancy, i.e. a less resilient state. These results show how the interplay of environmental drivers and fisheries caused a redistribution in the food web pathways, with negative consequences on the system's long-term stabil-
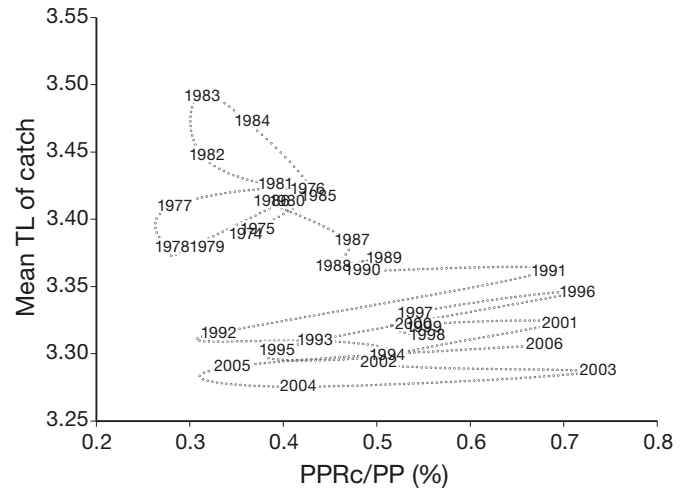

Fig. 1. Changes through time of mean trophic level (TL) of commercial catches versus primary production required to sustain the catch (PPRc/PP) in the central Baltic Sea catch from 1974 to 2009. The regime shift occurred at the end of the 1980s (Möllmann et al. 2009), when fisheries shifted their main target from cod to sprat, and can be seen as a decline in TL of catch, coupled with an increase in PPRc. For data and methods see Tomczak et al. (2013)

ity. Although they helped elucidate mechanisms that link fisheries with concerning changes in the system's resilience, ENA studies are not currently used to inform fisheries management tactical decisions.

Multi-species models are currently used limitedly to determine more accurate natural mortality parameters in a few stock assessment models in the Baltic (Möllmann et al. 2014). Yet, ENA indicators could play an important role in helping frame single-species evaluations within a system-wide and longer-term perspective of viable strategic solutions, so as to prevent or at least dampen future undesired changes. For example, they could be used in an integrated ecosystem assessment framework (Möllmann et al. 2014), and to dynamically update harvest control rules according to the state of the ecosystem. Although required by EAF, this broader perspective tends to be overlooked, possibly because changes in system dynamics and resilience are not easily related to actionable quantitative thresholds. For example, the mechanisms uncovered with ENA suggest that shifts in productivity, in cod diet or in the relative proportion of cod and sprat, can be indicators of changes resulting in higher risk for future ecosystem health, and in turn future fisheries productivity. Such indicators could be used to detect early warning signs that fishing effort should be reduced or shifted to different species and age classes. However, the understanding of such mechanisms is still mostly

qualitative, making it difficult to establish indicator thresholds that would trigger such harvest control rules. Moreover, it is difficult to establish the most appropriate harvest control rules to implement in such cases, although these models point to the key driving mechanisms in the system that should be monitored in order to detect signs of important system shifts. The recent guidance for the review of the MSFD decision for GES descriptor D4 (Food Webs), commissioned to ICES, while recognizing the challenges of producing advice from food web structure, takes the approach that knowledge gaps are not an excuse for inaction (ICES 2015). The recommendation made is that indicators for which the direction of change is not clearly interpreted as an improvement or decline of system health, or for which healthy value ranges are hard to establish, should be regularly measured as surveillance indicators, in order to learn more about how those indicators behave, and so as to be alerted to important system changes (Shephard et al. 2015). Key steps for an integrated ecosystem assessment are the establishment of broad goals, and then deriving from these the more detailed operational objectives guiding their implementation (Möllmann et al. 2014). Establishing such goals in a policy context could begin incorporating long-term ecosystem level concerns in fisheries management plans, and is the intended purpose of efforts underway, such as the Baltic Health Index (www.su. se/ostersjocentrum/english/baltic-eye/research/baltic-health-index).

Trophodynamic indices in the North Pacific: developing links to management strategies

The US North Pacific Exclusive Economic Zone has 2 main management bodies: the Pacific Management Council (overseeing resources off California, Oregon and Washington) and the North Pacific Management Council (overseeing resources off Alaska). These bodies manage highly valuable fisheries, such as Alaska pollock *Gadus chalcogrammus* (formerly *Theragra chalcogramma*) in the Alaska region and Pacific hake *Merluccius productus* in the US west coast region. Research on trophic and multispecies dynamics has been very active in both regions for several decades. At a multi-species level, different approaches to the development of trophodynamic indices have been taken, including mass-balanced energy models (Field & Francis 2006) and network models (Gaichas & Francis 2008). The former have been instrumental in identifying critical energetic pathways of the food web

and evaluating their dynamics in relation to harvest scenarios and climate variability (Aydin et al. 2005). Network models, on the other hand, are used to identify critical nodes and the potential of trophic reverberation through the food web. For example, the network model of the Gulf of Alaska by Gaichas & Francis (2008) identified species with high connectivity that, if impacted, have the potential to induce community-wide effects through changes to lower trophic level species. Such species include the highly abundant and commercially harvested stocks of Alaska pollock, Pacific cod *G. macrocephalus*, Pacific halibut *Hippoglossus stenolepis* and arrowtooth flounder *Atheresthes stomias*. These insights, therefore, point to which elements and dynamics should be monitored, in order to preserve the system's resilience.

In the US west coast fisheries management areas, research on trophic indexes has mostly focused on energy transfer and mass balance models (e.g. Field et al. 2006, Tolimieri et al. 2013). Mean trophic level of species is also computed and reported as part of the newly implemented integrated ecosystem assessment (Levin et al. 2009, www.noaa.gov/iea/CCIEA-Report/index.html), but it does not directly enter into management decisions. More empirical approaches have focussed on indicators related to community functioning and have provided new insights in trophodynamic research. For example, in a recent analysis of the last decade of fisheries-independent data, Tolimieri et al. (2013) found a decline in mean trophic level of the US west coast groundfish. The decline was driven by a reduction in biomass of high trophic level species, rather than an increase in lower trophic level species. Interestingly, these patterns occurred at a time of reduced fishing pressure; they are potentially driven by internal trophic dynamics rather than by direct fisheries removal. These results highlight how interpretation of trophic indicators needs to be context-sensitive. This creates potential communication challenges with managers who may be inclined to mistrust indicators that are not unequivocally linked to fishing pressure. A useful approach could be to incorporate uncertainty estimates that account for model specification and indicator behaviour as part of the potential sources of error (Fulton et al. 2005, Saisana et al. 2005).

Research focussed on key predator–prey interactions has also been very active in the North Pacific. For example, through the analysis of Alaska pollock abundances over different life-history stages, Bailey (2000) noted a shift of recruitment control, from late larval to the juvenile stages. The shift occurred in the mid- to late 1980s and coincided with a major

increase in arrowtooth flounder biomass, spatially overlapping with juvenile stages of Alaska pollock. Ciannelli & Bailey (2005) showed how, in warmer years, species that are generally segregated increase their spatial overlap due to the opening of cross-shelf thermal gateways. In a more recent analysis using historical fishery-independent catch data, Hunsicker et al. (2013) examined the effect of environmental variability on temperature-driven spatial overlap of juvenile Alaska pollock (the prey) and adult arrowtooth flounder (the predator) in the eastern Bering Sea. Results indicated that there is potential for flounder to trophically control Alaska pollock abundance due to increased overlap in the distribution of the 2 species under incumbent increases in water temperature and flounder abundance (Fig. 2). The findings of Hunsicker et al. (2013) could inform long-term management strategies, because they identify potential consequences of joint climate variability and exploitation stress on species' spatial dynamics and ensuing trophic interactions. For example, this information could be used to design an adaptive framework for regulatory measures that reduce juvenile pollock mortality in years when overlap with flounder is expected to be higher.

The establishment of management targets in both US regions remains mostly based on single-species assessment models and empirical analyses of fisheries-independent survey data, but trophodynamic indices enter at various stages of the policy-making process (Hollowed et al. 2011). In the Bering Sea, for example, in addition to setting single-species catch limits, there is an overall cap on annual total catch of all groundfish species that is lower than the sum of target species/species complexes limits, in consideration of the fact that there are food web interactions that are not accounted for by single-species models (DiCosimo et al. 2010, FR 2013). In order to preserve species that may be indirectly affected by fishing, limits are also imposed on the harvesting of species listed as 'forage species', i.e. species that are prey to marine mammals and seabirds (FR 2013). More detailed predator–prey models, focussing on a subset of species, are also used to provide alternative estimates of predation-driven natural mortality (e.g. Jurado-Molina et al. 2005).

Size-based indicators: examples from the EU

Given the broad availability of individual length or weight information on fished species, many studies on ecosystem-level effects of fisheries have focused
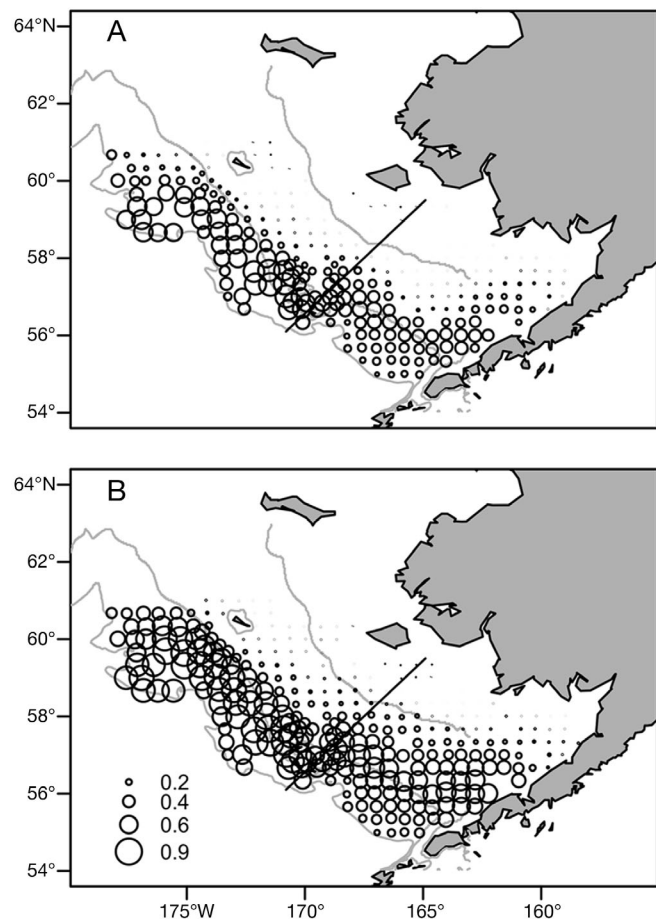

Fig. 2. Mean predicted probability of overlap between adult flounder (predator) and age-1 pollock (prey) during years of (A) low flounder biomass (before 1995) and (B) high flounder biomass (1995 and after) in the eastern Bering Sea. Circle size reflects probability of species overlap. The 50, 100 and 200 m depth contours are shown. The black line represents the northeast–southwest separation of survey strata, reflecting differences typically observed in groundfish assemblages in the northwest and southeast regions. Thermal gateways cause overlap of flounder and age-1 pollock otherwise segregated by a cold water mass. Flounder can prey on age-1 pollock, and this overlap is expected to increase with climate change, with potential consequences for the pollock fishery. Modified from Hunsicker et al. (2013)

on size-based community indicators. These indicators take advantage of the well tested link between changes in community structure and energy transfer with size-selective effects of fisheries (Kerr & Dickie 2001). Different size-based indicators have been proposed and used in the past decade (see e.g. Shin et al. 2005, 2012 and references therein), showing that mean fish size, the slope of the distribution of biomass across body sizes, i.e. biomass size spectra, and the relative proportion of large fish have historically declined with the increase in fishing mortality

in many systems (e.g. Jennings et al. 2002, Daan et al. 2003, Jennings & Blanchard 2004, Greenstreet & Rogers 2006), and similar effects have been found comparing locations with different fishing effort (e.g. Claudet et al. 2006, Greenstreet & Rogers 2006, Sundelöf et al. 2013). Based on this evidence, in the EU, the relative proportion of large fish in demersal communities, i.e. the Large Fish Indicator (LFI), was formally adopted by OSPAR to assess achievement of one of the established ecological quality objectives (Heslenfeld & Enserink 2008), and then as one of the indicators used to monitor progress towards acquiring GES, as prescribed by the recently implemented European MSFD (EU-COM 2008). This represents a well studied indicator for which a target reference point was established, below which there is a requirement to rebuild the community through a reduction in fishing pressure. The LFI has known relationships with overall fishing mortality on the demersal fish community that can be used to implement this effort reduction (Modica et al. 2014). Using this index in isolation, nonetheless, does not allow to disentangle changes due to fishing pressure from changes in productivity, and/or shifts in species com-

position of the community (Fung et al. 2012), and thus cross-validation using multiple approaches is needed (e.g. Daan et al. 2003, Greenstreet & Rogers 2006, Shannon et al. 2014b, Moutopoulos et al. 2014). For example, Fig. 3 illustrates how synthetic trophodynamic indicators such as PPR can be misleading if interpreted in isolation. In fact, a decline in PPR suggests more efficient cycling of resources and a healthier state, but this appears to be mainly driven by an increase in low trophic level catches, due to the decline and subsequent conservative management of cod (as illustrated by the rebuilding trend for the cod population biomass at the tail end of the time-series, while catches are kept low by management measures; Hornborg et al. 2013). For this reason, proposed indicator frameworks generally call for suites of complementary indicators (Rochet et al. 2005, Niemeijer & de Groot 2008). There are several well-tested community level indicators with known properties, for which historical reference points and desirable directions of change can be used as benchmarks (e.g. Large et al. 2013), and indicator suites that, in combination, can be interpreted unambiguously (e.g. Dambacher et al. 2009). But transforming multi-species
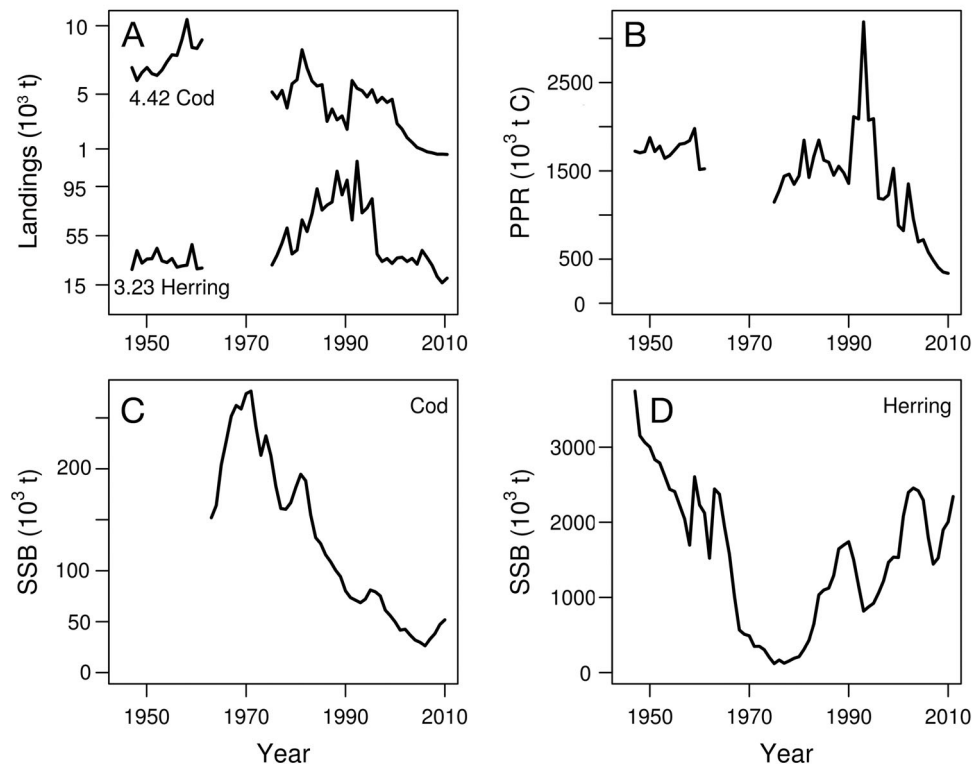

Fig. 3. Changes in catches, primary production required (PPR) and spawning stock biomass (SSB) for cod and herring in the Swedish landings from ICES area IIIA. (A) Changes over time in landings of the top-predator (cod) and forage species (herring), with corresponding trophic level (TL) indicated on the plot. (B) Changes in overall PPR over time. (C) SSB for North Sea cod stock from ICES. (D) SSB for North Sea autumn spawning herring stock from ICES. Modified from Hornborg et al. (2013)

information into recommendations that can be received in a single-stock oriented management context can often be even less straightforward than the case of the LFI. In some cases, the relationship between the indicator and the drivers that affect its behaviour are not fully understood or do not allow setting reference points. These might still be useful as 'surveillance' metrics, giving early warning signals of ecosystem disruption that should trigger in-depth analyses to understand whether management interventions might be required (Shephard et al. 2015).

Trophodynamic insights from the Mediterranean Sea

European Mediterranean countries, although under the same MSFD requirements as northern European countries, are part of an entirely separate basin, shared with North African and Middle-Eastern countries. Several studies have used trophodynamic indicators to gain insight into fishing effects in Mediterranean sub-systems at multiple scales: from the Black Sea (Oguz et al. 2012, Akoglu et al. 2014) to the Greek seas (Moutopoulos et al. 2014), and from the Adriatic Sea (Coll et al. 2010) and the Venice Lagoon (Libralato et al. 2004) to the Gulf of Gabes (Hattab et al. 2013). In contrast with the North Sea, where there is a long tradition of transnational collaboration fostered by the ICES (www.ices.dk), the majority of European Mediterranean studies are small-scale and sub-basin analyses. However, these regions are clearly interconnected both socially and ecologically (Coll & Libralato 2012). The lack of large-scale studies and the shorter tradition for resource assessment (Colloca et al. 2013) is generally justified by the high ecological heterogeneity, or by lack of data, particularly for the southern basin. Nonetheless, this gap is likely to stem from institutional fragmentation as well, in spite of collaborative efforts promoted, for example, by the General Fisheries Council for the Mediterranean (GFCM) (Cochrane & De Young 2008) and the Mediterranean Science Commission (CIESM).

A multivariate analysis of indicators obtained from over 40 models applied to Mediterranean regions (Coll & Libralato 2012) revealed that, in addition to spatial differences explained by ecosystem characteristics (i.e. level of influence of coastal activities), it was possible to identify a large-scale spatial pattern driven by differences in fishing exploitation. In particular, the mTLc was significantly different between areas with different fishing pressure. Notably, as predicted by Pauly et al. (1998), mTLc was higher in lightly to unexploited ecosystems (mTLc = 3.72 ± 0.25 SD) than in highly exploited ones (mTLc = 2.78 ± 0.67; Coll & Libralato 2012). Using trophodynamic metrics to classify landings data at the whole basin scale (Pranovi et al. 2014), moreover, highlighted a general and progressive degradation in some Mediterranean sub-basins, mainly driven by fishing pressure. These results, although obtained using commercial catch data, were consistent with those from scientific trawl survey data (e.g. Ferretti et al. 2013 for the Adriatic Sea). This is relevant because, compared to northern European waters, there have been fewer standardized large-scale surveys in the Mediterranean, with shorter time-series and patchier data quality. These studies prove that lack of use of trophodynamic metrics in the region cannot be justified by data limitations. Other studies have also confirmed that landings data can be informatively used in trophodynamic studies of the region. Such studies compared landings-derived indicators with models (Pranovi et al. 2003, Libralato et al. 2004, Oguz et al. 2012, Akoglu et al. 2014), or landings with trawl survey patterns (Ferretti et al. 2013) and trophodynamic indicator trends derived from landings, versus trawl survey data and models (Coll et al. 2010). Despite the knowledge gained, and successful validation tests, trophodynamic indicators to date have had extremely limited influence on the management of Mediterranean waters.

One reason for the limited management use of trophodynamic indicators is a general difficulty in defining absolute indicator thresholds to be used as references to trigger single species management measures. Nevertheless, a set of these indicators could be used to establish precautionary boundaries that help define a safe space within which single species assessment can operate. As an example, Libralato et al. (2008) proposed an approach for embedding mTLc, catches and PPR in a framework defining reference values and practical management targets (i.e. maximum allowable catches), and these were specifically applied in the Mediterranean Sea.

Another candidate reason for the reluctance to adopt trophodynamic models and indicators as assessment and/or monitoring tools is the perceived low explanatory power of the models used. Given the complexity of Mediterranean ecological systems, with very diverse fishing targets and mixed fishing metiers (e.g. Moutopoulos et al. 2013), it is difficult to reduce the system to few dominant variables, so much so that any simplified model is viewed with skepticism by managers. Nonetheless, some of the

studies cited thus far have shown that indicators can provide reliable information despite socio-ecological intricacies of the system (Moutopoulos et al. 2014, Pranovi et al. 2014). Nevertheless, complex integrated modelling approaches (Moutopoulos et al. 2013), embedding the known complexity of Mediterranean ecosystems and fisheries, and using common fisheries management reference points (i.e. fishing effort, fishing mortality), have not received much attention either. Therefore, lack of access to quality data, lack of scientific understanding, insufficient guidance on potential applications, or oversimplified modelling do not appear to be sufficient reasons to explain why trophodynamic indicators and models are not used in management planning, assessments or monitoring (Coll et al. 2010). It seems that a role is also played by the need for managers to demonstrate results on short-term temporal scales that match political mandates, combined with strong pressures from industry stakeholders serving a short-term-goal driven market.

Overall, Mediterranean countries appear to have made even less progress in using trophodynamic information to inform management than Northern EU coastal countries. On a positive note, the adoption of common guidelines under the umbrella of the EU Common Fisheries Policy, and, importantly, the requirements established by the EU MSFD to use common benchmarks across all European countries, created an incentive to closing this gap in the northern Mediterranean basin. It is worth mentioning, however, that other Mediterranean countries that are not subject to the EU requirements, lacking resources and political will, are likely to keep lagging behind, standing in the way of a basin-wide implementation of EBM (Coll & Libralato 2012).

CONCLUSION AND PERSPECTIVES

Research on changes in food web dynamics has helped elucidate how fishing influences ecosystem functioning and how this can in turn feed back on fishing dynamics, thus helping to evaluate effects of management decisions (Yodzis 1998, Folke et al. 2004, Coll et al. 2006, Walters et al. 2008, Mackinson et al. 2009, Frank et al. 2011, Coll & Libralato 2012, Tomczak et al. 2013). Nonetheless, the information ensuing from these studies has had few direct implications for tactical management decisions as of yet; it is limited to, for example, modifying fishing mortality estimations based on food web interactions in traditional single-species models (Hollowed et al. 2011,

Möllmann et al. 2014), or establishing precautionary fishing quotas that leave a reservoir for other marine predators (e.g. Agnew 1997, FR 2013). This is probably not by chance, as these uses tend to expand on existing management tools and practices.

Instead, the use of information provided by ecosystem- and community-level indicators is often lacking a clear link between trophic indicators that highlight a fisheries-driven threat and specific management measures to address it. Indicator science has consolidated best practices and analytical approaches for designing indicators of changes in ecosystem health in a way that is relevant to managers. Emphasis has been placed on developing measures that are easy to interpret and communicate, robust and with well-understood behaviour (Rice & Rochet 2005). Yet many indicators often fall short for the requirement of being actionable. Even in cases where it may be possible to identify management solutions, the cost of indicator simplicity is the potential that they may be relying on flawed assumptions, or there may be confounding factors driving indicator patterns (Branch et al. 2010, Sethi et al. 2010, Hornborg et al. 2013). For this reason, proposed frameworks recommend using suites of complementary indicators with a clear framework to interpret the results (Rochet et al. 2005, Dambacher et al. 2009, Fay et al. 2013, Fulton et al. 2014). These can provide more robust advice than a single indicator, and can be calculated from easily available and regularly updated empirical data, although they would benefit from a clearer communication of associated uncertainty (or improved accuracy).

Dynamic models, on the other hand, can help account for non-linear interactions, emergent properties, environmental forcing and non-stationary responses. Large food web models, such as EwE (Christensen & Walters 2004), Altantis (Fulton et al. 2004a,b) and Osmose (Shin & Cury 2001), can be integrated with hydrodynamical-biogeochemical models and used as decision support tools by simulating different fishing mortality across species, different nutrient availability and hydrodynamic conditions under varying climate change and management scenarios (e.g. Kaplan et al. 2010). Such tools might be used to identify strategies for long-term sustainability (Plagányi 2007). Spatially explicit models and empirical studies of spatio-temporal dynamics (Hunsicker et al. 2013) can also help parameterize fine-scale trade-off analyses to inform marine spatial management decisions (e.g. White et al. 2012).

Indeed, complex trophodynamic models and simulations still lack the detailed mechanistic understand-

ing of spatio-temporal trophic links needed to make quantitative assessments and forecasts with the level of accuracy required to identify tactical management provisions (Aydin & Mueter 2007). Nevertheless, they have proven valuable towards understanding broad ecosystem dynamics, which is especially relevant in order to prevent/mitigate regime shifts or impacts of near future climate-driven changes (Cheung et al. 2013, Poloczanska et al. 2013), and inform long-term sustainability strategies. For example, ecosystem-level indicators based on ENA and food web analysis could provide early warning signals of risks of fishing heavily certain species and size groups (Cury et al. 2005, IEEP 2005, Moloney et al. 2005), and monitor whether exploitation is occurring within a safe operating space (Libralato et al. 2008). These considerations should be part of an EAF, but are rarely considered in policy contexts (although see e.g. the Puget Sound Integrated Ecosystem Assessment, Tallis et al. 2010). Framing short-term decisions, such as catch quotas, into a long-term sustainability perspective may also be valuable for managing multiple objectives simultaneously. For example, Williams et al. (2011) showed that the amount of Chinook salmon required to rebuild the killer whale population in British Columbia, Canada, would not leave enough salmon for recreational fisheries and First Nations tribes' fishing quotas. These seemingly conflicting objectives may be reconciled by setting a staggered time-line for achieving each management objective: temporarily reducing fishing pressure would allow the killer whale and salmon populations to rebuild, but delay benefits to fishermen to when the salmon population is recovered.

An appropriately defined suite of different indicators and models of varying complexity can give the rapid response and reduced-uncertainty type of insight required by managers, combined with the bigger picture and longer-term vision (Plagányi et al. 2014b) that ensures that potential emerging effects and tipping points (e.g. Plagányi et al. 2014a, Kelly et al. 2015) are not overlooked. Additionally, a combination of several models and indicators provides the opportunity to appropriately weigh risks and benefits associated with different actions, thus improving credibility (Harwood & Stokes 2003, Fulton et al. 2014). Estimation of uncertainty is now routinely performed in a number of single-species assessments used in fisheries management approaches, but dealing with uncertainties is increasingly problematic as model complexity increases (McElhany et al. 2010). For trophodynamic indicators, often confidence intervals are not reported, although some sensitivity ana-

lyses of empirical indicators exist (e.g. Jennings & Dulvy 2005, Travers et al. 2006, Libralato et al. 2008). If one were to employ a combination of models and indicators, as suggested above, it would be best to adopt a framework that leverages diverse techniques, such as Bayesian hierarchical methods and model-averaging, or model ensembles to provide confidence bounds for the combination of estimates used (Hill et al. 2007, Cressie et al. 2009). Importantly, recent approaches developed for decision-making under uncertainty could be adopted to weigh the risks and effectiveness of different interventions (e.g. Smith et al. 2007, Fulton et al. 2014). In addition, greater clarity and transparency in communicating uncertainty to managers could help address mistrust and, by helping distinguish when interpretations of results are considered more or less robust, encourage the perception of trophodynamic estimates as reliable.

In summary, even if current advances do not yet support a fully integrated ecosystem assessment process leading to specific management provisions, there is still ample room for improving upon current management practices with existing techniques and tools for long-term strategic goals. It seems that a role is also played by governance institutional inertia. This might be motivated by the challenge of shifting towards more complex, interdisciplinary, integrated approaches for a governance structure that, for a long time, was designed to function on a single-sector basis. Indeed, the few examples cited of management implementation of trophodynamic studies involve modifications of existing measures that are easier to implement than a radical overhaul (Aswani et al. 2012, Berkes 2012). But it is also likely that the institutional challenge lies in the need for managers to demonstrate results on the short-term temporal scales that match political mandates, combined with strong pressures from industry stake-holders serving a short-term-goal driven market (García & Charles 2007). In this context, the incentive to adopt provisions with effects that can be quickly demonstrated is understandably higher than to embark in more uncertain, complex and costly approaches serving long-term risk avoidance (Sutinen & Soboil 2003, García & Charles 2007, Daily et al. 2009). In face of these hurdles, tangible evidence of ecosystem-level impacts may help create the political context for managers to prioritize ecosystem-wide concerns over fishing sector needs (García & Charles 2007). One example of a public opinion-driven shift in management priorities from short-term gains to conservation goals is the recent EU discard ban that was facilitated by a highly advertised discarding incident (Borges 2015; although

see Sardà et al. 2015). This highlights the importance of demonstrating how reduced system resilience translates into risks for the long-term provision of goods and services that people care about.

Towards integrated socio-ecological assessments

The language of current policy mandates increasingly describes management objectives in terms of ecosystem services, i.e. benefits to people (e.g. FAO 1995, Restrepo et al. 1998, EU-COM 2008) and refers to ecosystem health in a holistic way that incorporates people as an integral part of the system (e.g. see Halpern et al. 2012). The paradigm shift necessary to implement EBM (Berkes 2012) requires understanding what kind of ecosystem can deliver the desired societal goals, such as food security, biodiversity conservation, cultural values, etc., so as to establish appropriate indicators of the state of the system and set reference points. This implies that fisheries management is a multi-objective practice that must evaluate trade-offs across different possibly conflicting ways humans value and use marine resources (Leslie & McLeod 2007, Link 2011, de Jonge et al. 2012, Halpern et al. 2012). In order to assess these trade-offs, management advice must be increasingly interdisciplinary, integrating across multiple sectors, and across time and space (e.g. integrated ecosystem assessments, Levin et al. 2014). This more recent development of the EBM perspective accentuates more than ever the need to understand how trophodynamics affect the way management actions ripple through the ecosystem, and what are the far-reaching effects on the various goods and services desired by people. To do so, a new generation of models is required, expanded to include climatic and physical drivers on one hand, and socio-economic dimensions on the other (Fulton 2010, Rose 2012). Tools to, at least in part, deal with these new levels of complexity in decision-making already exist (e.g. SeaSketch, www.seasketch.org; the Ocean Health Index, www.oceanhealthindex.org; InVest, www.naturalcapital-project.org/InVEST.html), and several institutes and initiatives are actively providing this type of advice (e.g. Centre for Ocean Solutions, www.centerforoceansolutions.org; Future Earth, www.futureearth.info; the Natural Capital Project, www.naturalcapitalproject.org; IPBES, www.ipbes.net/), as well as applied examples (Fulton et al. 2014, Levin et al. 2014).

There are still many areas in need of improvement, but this review suggests that the biggest challenge standing in the way of implementing an ecosystem approach, at least in developed countries, may no longer be the lack of understanding of food web dynamics. Rather, 2 important issues arise from integrated ecosystem assessments. The first is the lack of decision support tools helping with the integration of ecological and social assessments of fisheries (Agboola & Braimoh 2009, Christie 2011, Koehn et al. 2013); the second is the need for a more effective institutional process to translate this interdisciplinary knowledge into action (Mora et al. 2009, Coll & Libralato 2012). Increased integration requires developing interdisciplinary expertise (Hollowed et al. 2011, de Jonge et al. 2012, Longo & Halpern 2015), and new levels of coordination and information exchange across agencies and departments (Arkema et al. 2006, García & Charles 2007, Rosenberg & Sandifer 2012, Dickey-Collas 2014). Changes may also be necessary in traditional monitoring protocols and routine sampling designs, in order to collect the information supporting new kinds of models (e.g. Casini et al. 2011, Möllmann et al. 2014). Participatory and multi-stakeholder decision-making is understandably complex (Dickey-Collas 2014), and may put strain and cause fatigue to institutions that were originally conceived to work autonomously.

In data-poor regions, on the other hand, the main obstacles may still be in gaining a better understanding of the local ecosystem — data availability and technical know-how often remain major stumbling blocks. Single-species assessment methods for data-poor fisheries are developing rapidly (Thorson et al. 2013, Carruthers et al. 2014, Rosenberg et al. 2014), as well as community-level indicators that can be based on catch data (e.g. Libralato et al. 2008, Kleisner & Pauly 2011), and some data-poor multi-species models also exist (Ryther 1969, Jennings et al. 2008, Gascuel & Pauly 2009, Rosenberg et al. 2014). Many of the indicators proposed rely on assumptions derived from better studied systems that may behave differently since there is a geographical bias in the types of systems that are better known (i.e. temperate, upwelling) versus those that are more data poor and less studied (i.e. tropical, oligotrophic; e.g. Vasconcellos & Cochrane 2005, Shin et al. 2012). Nonetheless, examples from data-limited applications exist (Smith et al. 2007), and alternative methods can be adopted, incorporating local expert opinion and ancillary sources of qualitative information (Ban et al. 2009). Ongoing initiatives are focusing on generating data-limited management tools with a multi-objective approach (e.g. RARE's Fish Forever project toolkits, www.fishforever.org), or incorporating ecosystem considerations (e.g. EDF's Framework for Integrated

Stock and Habitat Evaluation [FISHE], http://fishe.edf.org). Ecosystem-based management is still viewed as an over-ambitious goal in systems lacking capacity and/or resources (Patrick & Link 2015). The research reviewed here shows that it is a goal that cannot be ignored, particularly in regions where fisheries are so central to human wellbeing, and that there is research to help implement them. Similarly to data-rich systems, however, socio-economic aspects need to be integrated into the assessments (Koehn et al. 2013), and focussing on procedural elements of management will be an integral part of success in implementing an ecosystem approach (McClanahan et al. 2009). The particular social and governance characteristics of data-limited contexts and developing nations make it more advisable to build EBM on customary management processes rather than attempt a radical revolution of existing institutions (Aswani et al. 2012). Therefore, in envisioning a path towards EBM in data-poor situations, single-species assessment tools should be developed in tandem with approaches capturing ecosystem dynamics while taking into account local socio-economic processes, in order to move towards overall ecosystem health (Tallis et al. 2010, Fulton et al. 2014).

We have come a long way and have made notable progress in our understanding of the role of trophodynamics in regulating fisheries, and in developing tools to use this knowledge to improve fisheries management. Nonetheless, in order to incorporate this knowledge in management processes, there is still ample room for improvement. Indicators and models of varying complexity should be used in combination and uncertainty should be clearly communicated, including assessing risks associated with different interventions. Importantly, these approaches are, at least currently, best suited for long-term strategic planning, which is at odds with more customary short-term objectives. Incentives to favour long-term targets of sustainability and system resilience can come from demonstrating clear links with goods and services people care about, and provide decision support systems that can evaluate the full set of multi-sectorial trade-offs. Thus the concept of EBM is enriched to become truly interdisciplinary, and embrace an idea of ecosystem health that includes evaluating social and economic dimensions (Daily et al. 2009, Halpern et al. 2012).

In view of an increasingly crowded and growingly unpredictable planet, bridging the remaining gaps to a true implementation of an ecosystem approach in developed as well as developing regions appears more urgent than ever.

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