Ecosystem-level reference points: Moving toward ecosystem-based fisheries management

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SEDAR103-RD-02

February 2026

Received: 19 September 2023 | Revised: 6 February 2024 | Accepted: 8 February 2024

DOI: 10.1002/mcf2.10285

PERSPECTIVE

Ecosystem-level reference points: Moving toward ecosystem-based fisheries management

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Abstract

Objective: To support the movement in marine fisheries management toward ecosystem-based fisheries management by exploring ecosystem-level reference points (ELRPs) as an option for managing fisheries at the ecosystem level. An ELRP is an ecosystem harvest level or indicator with one or more associated benchmarks or thresholds (i.e., targets, limits) to identify, monitor, or maintain desirable ecosystem conditions and functions.

Methods: This paper explores the development and implementation of ELRPs in fisheries management to support ecosystem and fisheries sustainability, help identify when ecosystem changes that impact fisheries resources occur, and foster discussions of trade-offs in management decisions.

Result: We organize existing and potential ELRPs into five categories (statistical analysis of nonlinear dynamics and tipping points, ecosystem productivity, ecosystem trophic information, biodiversity, and human dimensions), provide an overview of analytical methods that can estimate ELRP benchmarks, provide examples of where ELRP benchmarks are being used today, and evaluate pros and cons of the different ELRP categories. We also attempt to identify potential next steps for fisheries scientists and managers to further the science, development, and application of ELRPs.

Conclusion: Ecosystem-level reference points can be used as a proactive accountability mechanism to achieve ecosystem objectives and maintain the ecosystem in a preferred operating space or as an early warning that ecosystem-level changes (e.g., tipping points) could be imminent if current biological and ecological trends in the system continue.

KEYWORDS

ecosystem approach to fisheries management, ecosystem-based fisheries management, ecosystem cap, ecosystem indicators, productivity cap, ELRP, ecosystem-level reference point

INTRODUCTION

> **Impact statement**
>
> Ecosystem-level reference points (ELRPs) are a fundamental concept in ecosystem-based fisheries management (EBFM) that are not currently being used to their full potential. Given the challenges associated with managing fisheries in a changing climate, this paper highlights ELRPs as a tool to advance the field of EBFM in the United States and globally.

Traditional marine fisheries management tends to be single-species focused, where a fish species or stock is assessed without explicit integration of climate and ecosystem influences (Skern-Mauritzen et al. 2016; Marshall et al. 2019). Managers then identify stock-specific biomass and fishing mortality reference points and set fishing regulations (e.g., harvest limits and quotas and other controls like bag and size limits) to achieve predetermined objectives for the fishery (Methot et al. 2014). This management approach can result in unexpected impacts because there are many factors other than fishing that affect fish populations, including interactions with other species, environmental conditions, habitat, and other anthropogenic pressures. Over the past several decades, fisheries management agencies the world over have recognized the need to move toward more holistic approaches to better understand the impacts of these and other ecosystem interactions on the species being managed (e.g., Pikitch et al. 2004; Link 2010; Gaichas et al. 2018; Fulton et al. 2019; Townsend et al. 2019), an approach commonly referred to as ecosystem-based fisheries management.

In the United States, the National Marine Fisheries Service—the agency responsible for the management of marine resources in federal waters—released an ecosystem-based fisheries management Policy and Road Map in 2016 to clarify its commitment to ecosystem-based fisheries management for U.S. fisheries and establish a framework to guide and accelerate its implementation (National Marine Fisheries Service 2016). The policy defines ecosystem-based fisheries management as

> a systematic approach to fisheries management in a geographically specified area that contributes to the resilience and sustainability of the ecosystem; recognizes the physical, biological, economic, and social interactions among the affected fishery-related components of the ecosystem, including humans; and seeks to optimize benefits among a diverse set of societal goals.

One of the policy's six core guiding principles is to “incorporate ecosystem considerations into management advice” and includes the development and monitoring of ecosystem-level reference points (ELRPs).

Biological reference points, defined here as target levels of an indicator that trigger management actions (Hodgdon et al. 2022), provide a basis or standard for evaluation and have become the cornerstone of global fisheries management (Caddy and Mahon 1995). The definition of thresholds is “the magnitude or intensity that must be exceeded for a certain reaction, phenomenon, result, or condition to occur or be manifested” (Oxford English Dictionary). Thus thresholds can be based on an ecological condition (e.g., an identified ecological tipping point) or a management benchmark (e.g., maintain biomass above 0.5 of the biomass at maximum sustainable yield). Here we apply threshold to situations with ecologically estimated change point, such as tipping points, and use the term “benchmark” to signify management thresholds that are not based on quantified change points. In the traditional single-species approach, biological reference points identify fishing mortality and biomass targets to achieve, as well as limits or thresholds to avoid, based on the biology of the species, including a combination of stock dynamics such as growth, recruitment, natural mortality, and fishing mortality (Gabriel and Mace 1999). Biological reference points are designed to achieve stated sustainability objectives for the fishery, such as maximum sustainable yield, or to maintain biomass above a certain level. Fisheries management programs typically use biological reference points to establish benchmarks that require management action when a reference point is crossed. For example, when a biological reference point is exceeded, management actions, in the form of harvest restrictions, could be triggered to reduce mortality from fishing to more sustainable levels, allow the stock to rebuild, or both. However, biological reference points often do not account for stock-level impacts on the ecosystem (and vice versa), which has led to the development of higher-level (i.e., extended beyond single species, such as the consideration of optimum yield; Patrick and Link 2015) reference points to achieve a range of management objectives (Table 1) and support the need to consider ELRPs. Here we propose a formal definition of an ELRP as an ecosystem harvest level or indicator with one or more associated benchmarks (i.e., targets, limits) or thresholds that are used to identify, monitor, or maintain desirable ecosystem conditions and function. Similar to how biological reference points are used

TABLE 1 Example reference points as defined in the fisheries literature.

Reference point name from literature	Characteristics	Citations
Biological reference point	Biologically based benchmarks that are used to inform managers about a stock's status relative to different fishery management objectives	Collie and Gislason (2001); Sainsbury (2008); Punt et al. (2016); Guo et al. (2019)
Multispecies biological reference point	Biologically based benchmarks that account for two or more species interactions and reflect potentially conflicting fishery management objectives	Moffitt et al. (2016); Link (2010); Gislason (1999)
Ecosystem-based biological reference point; ecological reference point	Ecosystem-level benchmarks that account for multispecies interactions, fishery operations, environmental factors, and/or other ecological variables at the ecosystem level that are important to sustainable fisheries management	Smith et al. (2015); Dolan et al. (2016); Holsman et al. (2016); Forrest and Walters (2009); Forrest et al. (2015); Tyrrell et al. (2011); Moffitt et al. (2016); Payne et al. (2017); Guo et al. (2019); Chagaris et al. (2020); Southeast Data, Assessment, and Review (2020)
Ecosystem-based reference point	Ecosystem-level benchmarks that account for broader dynamics in the ecosystem and reflect management objectives for a variety of ocean uses and services, including sustainable fisheries	Atlantic States Marine Fisheries Commission (2017); Southeast Data, Assessment, and Review (2020); Buchheister et al. (2017); Anstead et al. (2021)
Ecosystem-level decision criteria	Benchmarks based on ecosystem-level indicators that can, but do not have to, be associated with ecosystem tipping points or regime shifts	Samhouri et al. (2009); Tam et al. (2017)

for management, when ecosystem harvest levels or conditions cross the ELRP, management action should be triggered. The distinction is that the reference point is for the entire system of harvested taxa rather than for one stock. Management actions could include reducing ecosystem-wide fishing mortality, increasing protections for lower productivity stocks, adding a closed area, or implementing new monitoring programs (e.g., surveys) of important ecosystem processes (Stier et al. 2022). Ideally, ecosystem-level management responses that are akin to those associated with single-species overfishing (i.e., exceeding target rate of removal) or overfished (i.e., causing target biomass or other levels to become too low) scenarios are preferred.

Existing and potential ELRPs can be simple or complex, ranging from a basic summation of individual species reference points in an ecosystem to more complex analysis of ecosystem transformations (i.e., ecosystem changes that impact more than just a few species). Some ELRPs use widely available data and simple analyses that make them implementable in data-rich and data-poor systems. Often ELRPs include both direct and indirect indicators of change in an ecosystem (Pranovi et al. 2019). To date, most research has been devoted to the identification of appropriate ecosystem indicators for marine systems (e.g., Shin et al. 2012; Boldt et al. 2014), including the physical, biological, and human dimensions. However, only a subset of these indicators can be considered as ELRPs, as most lack identified thresholds or benchmarks. Here we discuss adding ELRPs in conjunction with single-species or multispecies reference points. In theory, ELRPs could replace single-species fisheries reference points; however, additional research and evaluation would be needed to ensure that lower-productivity stocks are not overexploited so as not to erode the condition of any individual component of harvested stocks in an ecosystem.

There are benefits associated with implementing ELRPs. Marine ecosystems are experiencing changes due to natural variability (e.g., decadal climate oscillations) and anthropogenic activities (e.g., pollution, habitat degradation, and fishing). Ecosystem-level reference points can help to improve our understanding of changes that are happening in the ecosystem, including biological marine communities and fisheries. The ELRPs can be used to monitor how an ecosystem functions at a higher level without needing to understand individual ecological processes (Link et al. 2015; Link 2018; Schlenger et al. 2018; Pranovi et al. 2019). Ecosystem-level reference points can also be used to provide early warning of ecosystem changes (Link 2018, 2021), allowing for management interventions to slow or stop the change or prepare users to adapt to the change. Importantly, ELRPs can be used to bring about discussions and decisions related to trade-offs across connected ecosystem components. Ecosystem management recognizes that actions on one species affect other predator and prey populations as well as the physical, social, and economic conditions throughout the system under consideration (National Marine Fisheries Service 2016). The ELRPs can help managers balance competing ecosystem objectives in management decisions by improving the ability to identify and understand potential adverse impacts on other parts of the system and ensure the continuation of well-managed fish stocks.

This paper explores the development and implementation of ELRPs in fisheries management to support ecosystem sustainability, to help identify when changes that impact fisheries resources occur, and to foster discussions of trade-offs in management decisions. We provide an overview of analytical methods that can estimate appropriate thresholds and examples of where they are being used today. We also evaluate pros and cons associated with ELRP categories, highlight potential implementation challenges, and offer guidance for managers to consider the use of ELRPs in their regions. Finally, we attempt to identify next steps for fisheries scientists and managers to further the science, development, and application of ELRPs.

CATEGORIES OF ECOSYSTEM-LEVEL REFERENCE POINTS

Here we divide potential and existing ELRPs into five general categories: (1) statistical analysis of nonlinear dynamics and tipping points, (2) ecosystem productivity, (3) ecosystem trophic information, (4) biodiversity, and (5) human dimensions. Potential and existing ELRPs (Table 2) require a range of data (e.g., catch, survey, model simulations), contain inherent assumptions (e.g., constant trophic level across all ontogenetic stages), and cover various facets of marine ecosystems and coupled socioecological systems. Potential ELRPs are the ecosystem indicators listed in the table that do not have associated benchmarks required to be considered for ELRPs at this time. Existing ELRPs have suggested thresholds or benchmarks (see ecosystem overfishing and cumulative biomass and productivity curves below). Here we briefly introduce each ELRP category and describe a few (but not all) existing and potential ELRPs in each category.

The first category of ELRPs focuses on the nonlinear dynamics associated with tipping points. However, not all ecosystem changes are nonlinear or involve tipping points. Even without the presence of tipping points, ecological thresholds or management benchmarks would still be beneficial but could be based on avoiding specific ecosystem conditions or based on an identified percent change or standard deviation of change from historical levels. The rest of the categories of ELRPs could be created around nonlinear ecosystem changes and the concept of tipping points or could be created for more linear ecosystem changes.

Ecosystem-level reference points based on nonlinear dynamics and tipping points

One approach to setting ELRPs is based on nonlinear dynamics and tipping points, concepts first introduced for ecological systems by Hollings in 1973. Nonlinear environment–stressor relationships have one or more curves or points of rapid change (Selkoe et al. 2015). A tipping point, or nonlinear threshold, is the point where the rapid change from one set of ecological conditions to another occurs (Figure 1B; Selkoe et al. 2015; Samhouri et al. 2017). When the entire ecosystem changes after crossing a tipping point, it is often referred to as a "regime shift." Several studies have shown that nonlinear dynamics (Figures 1B,C, 2) are common in ecological systems, with one study finding that 52% of more than 700 driver–response relationships evaluated were nonlinear (Hunsicker et al. 2016). Nonlinear dynamics and the existence of tipping points increase the risk and the cost of nonaction in the system. Instead of incremental increases in a pressure (e.g., fishing harvest) resulting in an incremental decrease in an ecosystem component (e.g., fishing biomass), the increase in pressure can result in an abrupt shift in the ecosystem (e.g., a collapse in the fishery with significant impacts on other ecosystem conditions) (Hunsicker et al. 2016). Additionally, it is often difficult to restore an ecosystem back to its original state after a threshold has been crossed. This is especially true if the system experiences hysteresis, where an abrupt change in state occurs and two different stable states occur under the same environmental conditions (Figure 1C). In such situations, the pathway leading to the shift to the alternate stable state is not the same as the pathway for restoring the system.

These nonlinear threshold responses or tipping points can form the basis for management targets and decision making aimed at maintaining the ecosystem in the desired state. For example, Large et al. (2015) noted a change in ecosystem properties when fishing mortality was above ~400,000 metric tons. Therefore, managers could set a precautionary ELRP at fishing mortality levels below this threshold. To avoid crossing tipping points, management often focuses on controlling the most important driver of the change. This type of management can work well if the driver is a human activity that can be controlled. However, if the driver is related to climate changes, management may not be able to control the driver and instead may

TABLE 2 Existing and potential ELRPs within the five general ELRP categories: (1) statistical analyses of nonlinear dynamics and tipping points, (2) ecosystem productivity, (3) ecosystem trophic information, (4) biodiversity, and (5) human dimensions. We note where existing and potential ELRPs have identified global thresholds.

Indicator	Description	Global thresholds?	References
ELRPs based on statistical analysis of nonlinear dynamics and tipping points			
Critical slowing down, critical speeding up, flickering	Critical slowing down (or speeding up) occurs when the temporal recovery from a disturbance decreases (or increases). Flickering occurs when the ecosystem alternates between two states over a short period of time.	No	Scheffer et al. (2009); Litzow et al. (2013); Titus and Watson (2020)
Changes in autocorrelation and variance	Responses of ecosystem components to drivers become more variable as the tipping point is approached.	No	Scheffer et al. (2009); Litzow et al. (2013)
System-specific tipping point thresholds	Thresholds identified for specific ecosystems that are necessary to maintain ecosystem in desired state.	No	Large et al. (2013, 2015); Foley et al. (2015); Hunsicker et al. (2016); Martone et al. (2017)
ELRPs based on ecosystem productivity			
Ecosystem overfishing indices (EOF)	Ratio of catch to chlorophyll *a* or to area.	Yes	Link and Watson (2019)
Ecosystem cap	Limit on the total biomass that can be removed from an ecosystem by fishery landings and discards per year.	No	Link (2018)
Primary production required (PPR) to sustain catch	How much primary production is needed to sustain current catch.	Not applicable	Pauly and Christensen (1995)
Percentage of the primary production required to sustain catch (%PPR) and ecosystem-based maximum sustainable catch (EMSCp)	%PPR is used to calculate EMSCp, which is the maximum catch that allows the ecosystem to achieve given probability (*p*) of being sustainable.	Yes (by ecosystem type)	Tudela et al. (2005); Coll et al. (2008)
ELRPs based on ecosystem trophic information			
Cumulative biomass curves	Determines inflection points for cumulative biomass to trophic level and for cumulative production to cumulative biomass. Cumulative biomass below 33% threshold indicates the ecosystem is degraded.	Yes	Pranovi and Link (2009); Pranovi et al. (2014); Link et al. (2015); Libralato et al. (2019)
Trophic spectra	Distribution of abundance, biomass, or catch across trophic levels in an ecosystem. More informative than mean trophic index.	No	Gascuel et al. (2005)
High trophic level indicator (HTL) and apex predator indicator (API)	HTI is the percentage of consumers at TL ≥4; API is the percentage of predators (excluding planktivores) at TL ≥4.	Yes	Bourdaud et al. (2016)
Loss production index (L-Index)	The L-Index is a proxy to quantify the effects of fishing.	Yes (consistent data units)	Libralato et al. (2008); Coll et al. (2008)
Mean trophic index (MTI), fishing in balance (FiB) index, and fisheries sustainability index (FSI)	MTI, FiB index, and FSI combine to track changes in fishing pressure across trophic levels and geographic areas. They are meant to be analyzed together to track fishing pressure (at higher or lower trophic levels and into new geographic areas).	No	Kleisner and Pauly (2011)

(Continues)

TABLE 2 (Continued)

Indicator	Description	Global thresholds?	References
Time cumulated indicator (TCI) and efficiency cumulated indicator (ECI)	TCI measures the residence time of biomass within the food web; ECI quantifies the fraction of secondary production reaching the top of the trophic chain.	No	Maureaud et al. (2017)
ELRPs based on ecosystem trophic information and biodiversity			
Ecosystem exploitation index (EEI)	EEI integrates four indicators: trophic balance index (evenness of exploitation across trophic levels), exploitation index (level of exploitation), species richness index (number of species above some identified percentage), and disturbance index (change in trophic structure).	No	Bundy et al. (2005)
ELRPs based on biodiversity			
Functional diversity, functional redundancy	Measures of the diversity or redundancy of specific roles or traits within an ecosystem.	No	Cadotte (2011); Cadotte et al. (2011); Leuzinger and Rewald (2021)
Nondeclining exploited species (NDES)	NDES tracks the percentage of exploited species with level or increasing biomass.	No	Kleisner et al. (2015)
ELRPs based on human dimensions			
Economic indicators	Potential indicators include diversity of catch, reliance on fishing, etc.	No	Kasperski and Holland (2013); Jepson and Colburn (2013); Cline et al. (2017)
Well-being indicators	Potential indicators include resource access, self-determination, community values, traditional products, sense of place, and livelihoods.	No	Dolan and Metcalfe (2012); Breslow et al. (2017); Nelson et al. (2022)

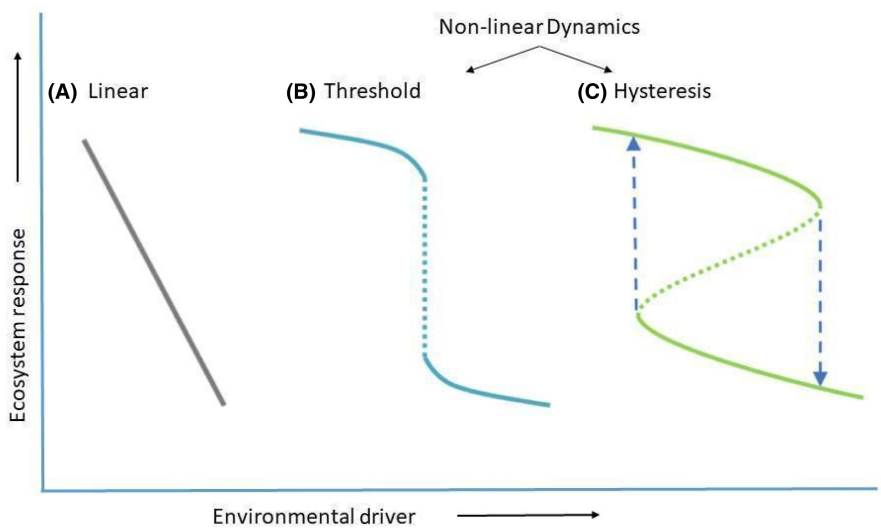

FIGURE 1 Schematic of the range of possible environmental driver–response relationships, from (A) linear and (B) nonlinear threshold to (C) nonlinear hysteresis. Modified with permission from Dudgeon et al. (2010).

focus on maintaining ecosystem resilience to increase system robustness or focus on adaptation to the impending change. Here we discuss two approaches for using tipping points and nonlinear dynamics to inform development of ELRPs and management thresholds: (1) monitor for statistical early warning signs of an impending tipping point when the exact location of a tipping point is unknown and (2) estimate or infer ecosystem-specific tipping point levels.

Statistical early warning indicators of an approaching tipping point

When an ecosystem is suspected of having a tipping point but the location of the tipping point is unknown, managers might be able to use statistical early warning signs to estimate the proximity to a tipping point and associated risk. Statistical early warning signs come from studies that show how complex systems can exhibit generalizable, consistent, and repeatable statistical behavior preceding a tipping point (Biggs et al. 2009; Litzow et al. 2013; Dakos et al. 2015; Foley et al. 2015). These signs include critical slowing down, flickering, and increase in autocorrelation and variance (Scheffer et al. 2009; Titus and Watson 2020). Critical slowing down (or speeding up) occurs when the time to recover from a disturbance increases (decreases) due to loss of resilience. Flickering occurs when the structure and function of the ecosystem alternates between two states over a short period of time. Other potential statistical early warning signs include changes in spatial or temporal autocorrelation (change across the ecosystem tends to become correlated in space and time) and variance (responses of ecosystem components to drivers become more or less variable as the tipping point is approached). We posit that these "early warning sign" indicators could be ELRPs if thresholds or benchmarks of change are identified. For example, an ELRP could be a given increase (e.g., 1 standard deviation) in spatial variance of high economic target fish stocks over time.

Litzow et al. (2013) found that spatial variability in catch increased 1–4 years prior to the collapse of crustacean fisheries in Alaska. The researchers conclude that if managers had been monitoring for changes in spatial variability, they could have been alerted to the impending crash 1–4 years prior to its occurrence and possibly had time to reduce fishing pressure. The use of spatial indicators is beneficial, as they require a shorter time series of data prior to the tipping point than temporal indicators to detect trends (Litzow et al. 2013). Litzow et al. (2013) suggest that these indicators work best when evaluating trends across multiple populations to increase power to detect significant changes in spatial variability of populations. A caution about statistical early warning signs is that they have not yet been shown to be consistent in complex systems (Biggs et al. 2009; Dakos et al. 2015) and may not occur in situations where there are not tipping points (Litzow and Hunsicker 2016). Additionally, the scale of the indicator (i.e., observation intervals) must match the dynamics of the biological response (i.e., life span of responsive organisms) for a signal to be detected (Bestelmeyer et al. 2011) and signals may be weak and perform poorly due to high levels of noise often found in real-world systems (Perretti and Munch 2012). Several studies also caution that they may not provide enough "warning" for managers to be able to take action to prevent the shift from occurring (see Biggs et al. 2009; Dakos et al. 2015). Thus, more research is needed before "early warning signs" can be considered as fully operational ELRPs, but they hold promise currently as monitoring indicators.

Statistical analysis of ecosystem-specific dynamics or tipping points

For some ecosystems, it may be useful to identify and implement ELRPs specific to that ecosystem, especially if a threshold is identified for a known or suspected tipping point. For example, off the northeast United States, Large et al. (2015) identified an exploitation threshold (~400,000 metric tons), where exploitation above this level resulted in a shift from pelagic to demersal ecosystem structure and an increase in gelatinous ctenophore abundance. Ecosystem-specific tipping points or thresholds can be estimated or inferred through (1) statistical analysis across time to evaluate nonlinear dynamics and threshold responses in historical time series data at one or a few locations, (2) statistical analysis at one point in time across survey data from a large number of sites of a specific ecosystem type that span a gradient of pressure (e.g., fishing intensity) and ecosystem health, or (3) theory-based mechanistic model simulations to estimate indicator levels and associated risk of crossing a tipping point. Once thresholds or tipping points are estimated, ELRPs can be identified at levels that aim to prevent crossing these thresholds, with the appropriate buffer being situational.

Recent advances in statistical software and analysis have made it easier to explore and detect nonlinear responses in ecological systems and identify tipping points (Large et al. 2013, 2015; Foley et al. 2015; Hunsicker et al. 2016; Martone et al. 2017). Using a generalized additive model, nonlinear dynamics can be identified by fitting driver–response relationships and using either the effective degrees of freedom or smoothness functions (Hunsicker et al. 2016). Once a nonlinear relationship has been identified, sequential *t*-test analyses of regime shifts or other forms of change-point analysis can be used to determine the critical point at which the system changes state (Large et al. 2013; Foley et al. 2015; Hunsicker et al. 2016; Martone et al. 2017). This approach can be extended into the multivariate space through the use of dynamic factor analysis (Large et al. 2015). Other multivariate analysis, such as principal component analysis, nonmetric multidimensional scaling, and redundancy analysis, can be used to identify ecological communities or regimes that are significantly different from each other (Martone et al. 2017) and experiencing dynamic conditions. A challenge with these approaches is that they are retrospective analyses; they provide information about where a change had occurred in the past and at what level for the associated driver. However, results of these studies may help managers determine nonlinearity and the location of tipping points in similar systems that have not already experienced a regime shift or provide indications of levels that if approaching again warrant further attention (e.g., the ~400,000-metric-ton level of landings from Large et al. 2015).

For ecosystems that lack a long data time series of available data, tipping points can be determined by analyzing short-term survey data across multiple spatial locations that span a gradient of pressure (e.g., fishing intensity) and ecosystem health. Karr et al. (2015) used this approach to determine levels of fish biomass associated with transitions from coral-dominated to macroalgal-dominated ecosystems. Estimated threshold levels may be robust across spatial scales; however, some studies found that thresholds may vary through space depending on environmental conditions (Marzloff et al. 2016).

If information is available to quantify key ecological processes, then case-specific model simulations of an ecosystem can be used to estimate ELRPs and the associated risk of crossing a tipping point. For example, Marzloff et al. (2016) used Monte Carlo model simulations in combination with an ecosystem model to identify long-term target levels of southern rock lobster *Jasus edwardsii* biomass on Tasmanian reefs in Australia associated with a low risk (less than 5% probability) of shifting to a long-spined sea urchin *Centrostephanus rodgersii* created barren ecosystem (because lobsters eat urchins, and urchins eat kelp; thus, lobster abundance ultimately impacts kelp abundance). Selkoe et al. (2015) and Stier et al. (2022) recommend that managers set precautionary buffers around the estimated tipping points depending on the scientific uncertainty surrounding the location of the tipping, risk tolerances, or cost–benefit analysis (Figure 2). This is a very similar concept to how the USA currently sets annual catch limits for many of its fisheries using Council tier systems (Methot et al. 2014) and thus could be easily understood and implemented.

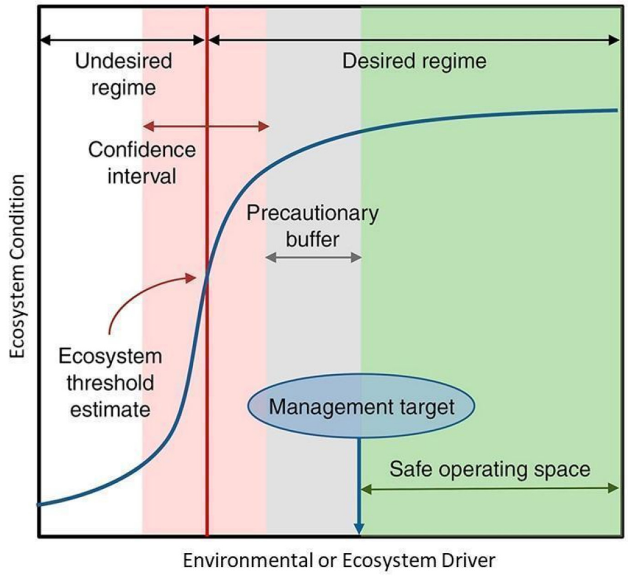

FIGURE 2 A conceptual model for addressing tipping points using management targets and buffers to maintain an ecosystem within a desired state. Modified from Selkoe et al. (2015).

Ecosystem-level reference points based on ecosystem productivity

Fish biomass is directly dependent on the primary productivity of the system, leading to there being a maximum amount of fish biomass that can be sustainably removed from an ecosystem over time (Fogarty et al. 2016). This limit on total removals from an ecosystem due to fishing can be used as an ELRP to help ensure ecosystem sustainability. Removals above that level can result in ecosystem overfishing. When ecosystem overfishing was first described over 20 years ago, the question was asked, “Can fisheries research provide a quantitative basis for defining ecosystem overfishing, and what monitoring information would be necessary to support ecosystem-based management?” (Murawski 2000). Below we describe recent research on universal indicators for ecosystem overfishing and case-specific implementation of ecosystem caps or limits on fisheries removals. Benchmarks associated with ecosystem overfishing do not have to be associated with known or suspected tipping points, but they often are. Other potential ELRPs related to ecosystem productivity (refer to Table 2 for more detail) include the primary production required to sustain catch (Tudela et al. 2005), ecosystem-based maximum sustainable catch (Coll et al. 2008), the loss production index (Libralato et al. 2008), and the fishing in balance index (Kleisner and Pauly 2011).

Ecosystem overfishing reference points

Three examples of ecosystem overfishing indicators with proposed universal benchmarks include the Fogarty, Friedland, and Ryther indices (Link and Watson 2019). The Fogarty and Friedland indices rely on fishery catch and satellite-derived primary production data. The Fogarty index is the ratio of total catch to primary productivity, and the Friedland index is the ratio of total catch to chlorophyll *a*. The Ryther index is simply the total catch per unit area. For these ELRPs, crossing the threshold does not necessarily mean ecosystem overfishing is occurring. Ecosystem overfishing is considered to occur for a system where the sum of all catches is flat or declining, catch per unit effort is declining, and total landings exceed limits of ecosystem production (Link and Watson 2019). Crossing an ecosystem overfishing indicator threshold would suggest that fishing levels across the ecosystem are too high and thus need to be reduced or assessed for other indicators of ecosystem decline.

More recently, Link (2021) suggests that the Ryther index shows that sustained ecosystem overfishing was occurring over the whole of the Northeast U.S. Continental Shelf Large Marine Ecosystem, beginning in the 1950s. Comparing this to catch estimates for individual northeast U.S. fishery species, Link (2021) suggests that large changes in the system (driven by species such as menhaden *Brevoortia* spp., herring [family Clupeidae], cod [family Gadidae], scallops [family Pectinidae], and lobster) could have been anticipated 2–3 years or more in advance, based on ecosystem overfishing shown in the Ryther index as compared with change based on analysis of landings of multiple individual stocks. Similarly, the Ryther index showed signs of ecosystem overfishing in the Gulf of Mexico 3–5 years prior to peak catches, followed by steep declines in the mid-1980s (Link 2021).

To date, ecosystem overfishing ELRPs are not yet being used operationally in management. Ecosystem overfishing indicators are being explored for management applications in parts of Africa (Link et al. 2020) and have been included in the U.S. Northeast State of the Ecosystem reports (Northeast Fisheries Science Center 2021a, 2021b) presented to the New England and Mid-Atlantic Fishery Management Councils (e.g., Figure 3). It is important to note that ecosystem overfishing indicators reported in the State of the Ecosystem reports were calculated for smaller distinct geographic areas (Georges Bank and Gulf of Maine). Additionally, the indicators in the State of the Ecosystem reports were modified to use commercial landings while the thresholds proposed by Link and Watson (2019) used total catch, which reduced the value of the index from the global estimates for the entire Northeast U.S. Continental Shelf Large Marine Ecosystem.

Currently there is no specific management action identified for crossing ecosystem overfishing thresholds, but there are some obvious recommendations that arise if ecosystem overfishing thresholds are exceeded, such as decreasing total fishing mortality. Thus, ecosystem overfishing indices are still being currently used only as indicators and not full ELRPs. They provide context to regional Fishery Management Councils on the overall state of the ecosystem and suite of fisheries.

Ecosystem-specific caps

An ecosystem cap is a limit on the total biomass that is removed from a specific ecosystem by commercial and recreational fisheries (i.e., harvest and dead releases). It may be based on established understanding of biophysical limits within the ecosystem (e.g., ecosystem overfishing benchmarks, suspected tipping point thresholds associated with biomass removal [as described in the

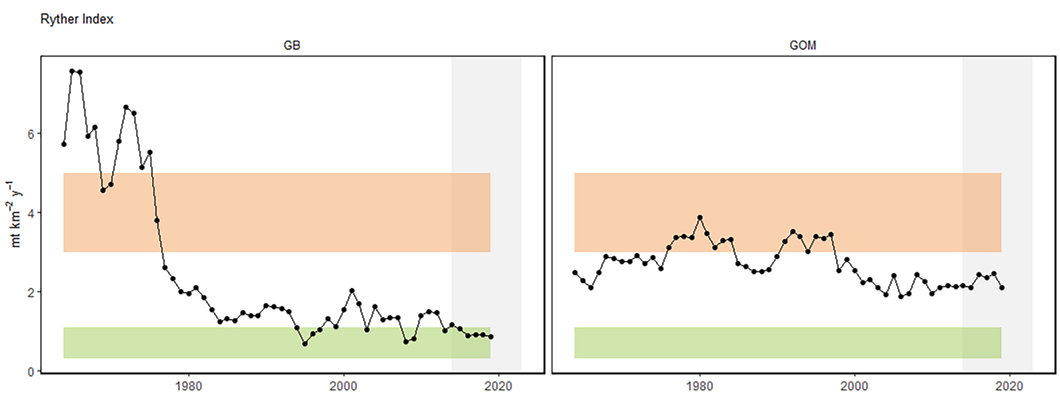

FIGURE 3 An example of the Ryther index (total landings per unit area) for Georges Bank (GB) and Gulf of Maine (GOM) reported in the 2021 New England State of the Ecosystem report. Green shading indicates the target annual range (0.3–1.1 metric tons [mt]/km^2), while orange shading indicates a limit of 3 mt/km^2 above which tipping points could occur when ecosystems are fished. Modified from the Northeast Fisheries Science Center (2021b).

previous section], aggregate production model outputs), or it may be set entirely based on sociopolitical considerations. Ecosystem caps can ensure that total removals from an ecosystem are sustainable and provide a forum for discussions of trade-offs among fisheries if the cap is less than the sum of the single-species catch limits (Link 2018; Holsman et al. 2020). Single-species management with individual maximum sustainable yield estimates can be misleading, as it is typically not possible to fish at maximum sustainable yield for all species simultaneously due to predator–prey dynamics, competition, and other multispecies interactions (Larkin 1977; Gaichas et al. 2012; Patrick and Link 2015; Craig and Link 2023). Hierarchy theory highlights how a system is more stable than its parts (in this case, total catch is more stable than catch of individual species) and managing at the system level can result in a more sustainable system overall (Simon 1962; Link 2018). As noted earlier, an ecosystem cap can be implemented instead of or in addition to single-species or multispecies catch limits. Using a cap instead of single-species catch limits introduces complex challenges and requires additional evaluation to ensure that lower-productivity stocks are not overexploited. An ecosystem cap can also be sensitive to shifting baselines. Fulton et al. (2022) show that a cap based on a system dominated by fast-growing species will be higher than a system based on slower-growing, long-lived species. Adjusting the cap toward high-productivity stocks can limit the ability of the ecosystem to recover to a state with more long-lived species. To date, in the United States, the cap has only been implemented in addition to single-species management (see North Pacific example below).

There are multiple methods for estimating ecosystem cap ELRPs. Below we briefly outline three types of analyses that can provide an estimate for an ecosystem cap: a cap based on (1) primary production, (2) surplus production, and (3) the sum of single-species' maximum sustainable yields, usually reduced by some buffer amount.

The first type of analysis involves using estimates of primary production to determine how much biomass can be sustainably removed from the ecosystem. This is the methodology used for the ecosystem overfishing ELRPs described above. For example, the cap could be derived directly from estimates of net primary production or chlorophyll *a* (Link and Watson 2019) or by limiting fishing mortality to the ratio of new production to total production (Fogarty et al. 2016). An advantage of these methods is that they are often based on readily available information (e.g., satellite estimates of primary productivity) and thus can be implemented in data poor systems. More complex calculations include the propagation of energy up the food chain, which is dependent on estimates of transfer efficiencies (i.e., how much energy is transferred between trophic levels through predation) and the trophic level of catch (Stock et al. 2017; Link and Watson 2019). Accurately predicting the relationship between primary production and fish catches can be complicated by the type of system (e.g., tropical versus temperate) and local conditions (Stock et al. 2017). Stock et al. (2017) also predict that climate change will alter the existing relationships and further reduce the accuracy of sustainable catch estimates extrapolated from estimates of primary productivity.

The second method involves calculating surplus production on an ecosystem scale. Calculations of surplus production are a foundation of fisheries management and are used throughout the world to estimate maximum sustainable yield of individual stocks. The same surplus production models that can be used to estimate single-species maximum sustainable yield can also be used to estimate the aggregate catch across an ecosystem, or ecosystem maximum sustainable yield (Mueter and Megrey 2006; Gaichas et al. 2012; Link et al. 2012). Advantages of this method include limited data requirements (i.e., a time series of catch and biomass; Mueter and Megrey 2006), the fact that the estimate accounts for multispecies interactions without needing to fully understand or ecologically describe the interactions (Lucey et al. 2012), and the familiarity of the resultant reference points to managers (Lucey et al. 2012). A disadvantage is that the calculations do not account for variability in growth, natural mortality, age structure, and recruitment. In general, these types of ecosystem-level analyses do not account for different life histories or productivity between species (Gaichas et al. 2012) or potential fishery interactions (Lucey et al. 2012); however, the former can be partially addressed by calculating surplus production for groups of species with similar life history traits (e.g., demersal versus pelagic) rather than for the entire ecosystem (Fulton et al. 2022).

Finally, ecosystem cap ELRPs can be estimated as a reduction from the sum of single-species maximum sustainable yield. Single-species maximum sustainable yield rarely accounts for predator–prey dynamics and is often estimated in systems with limited predation (large long-lived species at lower abundances), which can lead to overestimates of sustainable catch (Gaichas et al. 2012; Fulton et al. 2022). Recent research has shown that fishing mortality rates on predator and prey are often connected; the fishing mortality rate on Striped Bass *Morone saxatilis* (predator) affects the appropriate fishing mortality rate on Atlantic Menhaden *Brevoortia tyrannus* (prey) and vice versa (Anstead et al. 2021). A maximum of 75–90% of the sum of single-species maximum sustainable yield is often an appropriate target for biomass removals (Constable 2001; Gaichas et al. 2012).

Ecosystem caps have been implemented or are being considered in the USA and around the world. High-level policy documents for the United States and the Food and Agriculture Organization of the United Nations include ecosystem caps within their plans (National Marine Fisheries Service 2016; Koen-Alonso et al. 2019). In addition, at a smaller scale, the New England Fishery Management Council discussed the use of an ecosystem cap in the draft Example Fishery Ecosystem Plan for Georges Bank (New England Fishery Management Council 2019), but a cap has not yet been implemented. The North Pacific Fishery Management Council uses ecosystem caps to manage fisheries in the Gulf of Alaska and the Bering Sea (Mueter and Megrey 2006; Holsman et al. 2020); however, only the Bering Sea ecosystem cap is limiting, and thus, it is the only one discussed here. The cap for the Bering Sea was first estimated in 1984 as a sum of the existing allowable biological catches and codified into law in 2005. The 2 million metric ton cap is limiting in some years and has resulted in catch limits that are much lower than if the cap were not present. For example, the sum of all species allowable catches has totaled more than 2.8 million metric tons in some years (Witherell 1995; Witherell et al. 2000). When the sum of allowable catch across species exceeds the cap, managers reduce allowable catch on individual species to balance multiple management objectives, including maximizing yield, reducing the risk of exceeding catch limits on species that could close a fishery for the season, distributional objectives and mandates, and other ecosystem considerations (Holsman et al. 2020). In analyses of appropriate caps for these ecosystems, Mueter and Megrey (2006) used surplus production models to calculate that the current cap for the Bering Sea reflects a less than 20% probability of exceeding ecosystem-level maximum sustainable yield. Recent research estimating climate effects in the Bering Sea have found that the cap increased resilience for some but not all stocks (Holsman et al. 2020). Species that were often fished below their single-species catch limit due to the cap (e.g., Walleye *Sander vitreus*, Pollock *Pollachius virens*) were found to be more resilient to climate change than species that were almost always fished at their single-species catch limit (e.g., Pacific Cod *Gadus macrocephalus*).[1] This research also highlights how high stock biomass can buffer against environmental uncertainty (e.g., Gaichas et al. 2012; Holsman et al. 2020).

[1]If the ecosystem cap is lower than the sum of the single-species catch limits, managers may be challenged by stakeholders to consider complex biological and socioeconomic trade-offs with lowering the catch limit in one or more fisheries in order to maintain the combined-fishery catch under the ecosystem cap.

Ecosystem-level reference points based on ecosystem trophic information

The distribution of biomass (or catch) across trophic levels can be useful for assessing the impacts of fishing and the health of an ecosystem (Bundy et al. 2005; Gascuel et al. 2005; Kleisner and Pauly 2011; Link et al. 2015). Link et al. (2015) found that perturbed ecosystems tend to have biomass concentrated in lower trophic levels (see cumulative biomass curves below). Pauley et al. (1998) referenced this as fishing down the food web, which has been expanded to include fishing through the food web (Essington et al. 2006) for ecosystems with fisheries targeting middle trophic level species. Regardless, it is important to track how fishing and other stressors are impacting the composition of biomass within the ecosystem. Multiple indicators have been proposed, some with associated ELRP thresholds or benchmarks (see Table 2).

For example, the relative loss production index (L-index) was designed as a proxy to quantify the effects of fishing on an ecosystem that can be based on ecosystem models or landings data (Libralato et al. 2008). The L-index estimates loss in secondary production due to fishing relative to a theoretical situation with no loss due to fishing. Variations of the L-index account for ecosystem properties and fishing activity. When coupled with the probability of being sustainably fished, the L-index for an ecosystem can be compared to a reference level probability to determine the current fishing status. By incorporating the amount and ecological role of species caught (e.g., primary production requirements or trophic level) with ecosystem function (e.g., base primary production or trophic energy transfer), the L-index accounts for both herbivore and carnivore contributions to production (Coll et al. 2008; Libralato et al. 2008). Some drawbacks to the L-index include the assumption of a steady-state system, its high sensitivity to trophic level estimates, and that reference values for the probability of sustainable fishing developed by Libralato et al. (2008) should only be applied to other systems if data with the same units are used (Libralato et al. 2008).

Cumulative biomass curves were developed to determine if there are common and emergent patterns in marine ecosystems based on fundamental ecosystem processes (Pranovi and Link 2009; Pranovi et al. 2014; Link et al. 2015). The theoretical basis, cumulative trophic theory (Link et al. 2015), for such patterns relies on trophic energy transfer and the relationship between trophic level and cumulative biomass and cumulative biomass and cumulative production. The relationships result in sigmoidal and asymptotic logarithmic patterns, respectively (Figure 4, after Link et al. 2015). The sigmoidal curve of

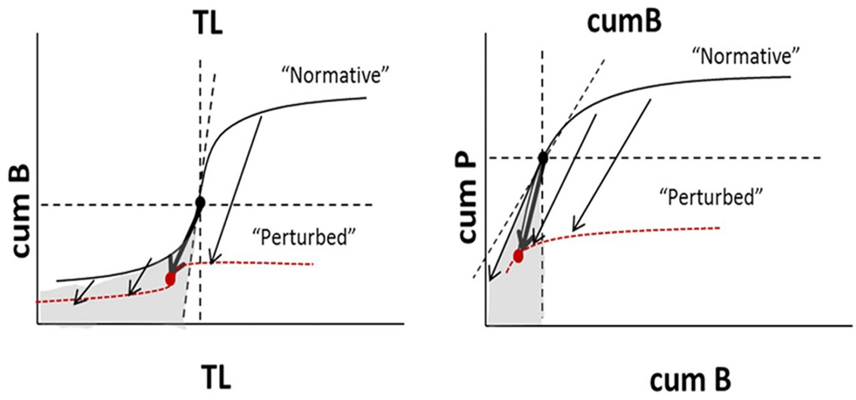

FIGURE 4 The shift (indicated by arrows) for cumulative biomass (cum B) over trophic level (TL) (left panel) and cumulative production (cum P) over cum B (right panel) from a normative system (solid black curve) to a perturbed system (red dashed curve). Axes crossing at right angles indicate inflection point, and the angled dashed line indicates slope. Gray shaded areas represent a zone of perturbation below some ecosystem threshold. Modified from Link (2021).

cumulative biomass to trophic level results from the characteristically rhomboid shape of marine food pyramids resulting from the build-up of standing biomass centered on trophic level 2 (Link et al. 2015; Libralato et al. 2019).

Negative perturbations to the ecosystem, such as ecosystem overfishing, result in flattening of the curves relative to a normal stable state (i.e., removal of biomass at higher trophic levels reduces the magnitude of the asymptote; see Figure 4), while ecosystem recovery exhibits steepening of the curves relative to the perturbed state (Link et al. 2015; Pranovi et al. 2019). Theory suggests that perturbed ecosystems should have reduced cumulative production and biomass accumulation (e.g., stored energy spread across trophic levels) as biomass is shifted toward lower trophic levels.

Testing the theory with commercial marine fisheries capture or landings data as a proxy for biomass found that most marine systems exhibited the expected patterns (Libralato et al. 2019; Pranovi et al. 2019; Link 2021). Further, derived curve parameters (e.g., values of x and y at inflection points and slope; see Figure 4) for a single system tracked over time show patterns of perturbation and recovery. Data from systems with known past perturbation (e.g., fishing pressure, invasive species) follow the expected theoretical patterns in curve parameters (i.e., changes in the inflection point intercepts and slope). Looking across multiple ecosystems suggests that a universal threshold (i.e., ELRP) may be present (Link et al. 2015; Pranovi et al. 2019). Libralato et al. (2019) identified inflection point thresholds for cumulative biomass at ~33%, trophic level at ~3.4, and slope at the inflection point at ~0.5 (e.g., any ecosystem with cumulative biomass below 33% is severely degraded). These global thresholds could be adjusted for specific marine ecosystems based on policy or management risk tolerances; however, the implication is that any detectable shifts from these thresholds represent a major impact on cumulative production and biomass in the ecosystem.

More extensive examination of cumulative biomass to trophic level relationships globally indicates the presence of a sigmoidal curve regardless of latitude, marine ecosystem type, environmental conditions, and stressor level and that marine ecosystems tend to exist in one of three states: continuous perturbation, continuous recovery, or alternating between perturbation and recovery (Pranovi et al. 2019). Thus, the parameters of the biomass curve could be used as ELRPs, either based on global thresholds discussed above, ecosystem-specific thresholds, or a specified amount of change from historical levels (such as a change in inflection point of a given percentage or standard deviation as per the statistical discussion above).

Ecosystem-level reference points based on biodiversity

It is common practice in ecology to track and monitor indicators related to biodiversity, including those related to diversity and richness. Loss of diversity or richness through time is often considered an indicator of decreased ecosystem health. Diversity indicators could become ELRPs if benchmarks are identified. As with other indicators, these benchmarks could be an identified numerical threshold or a benchmark of change (i.e., percent or standard deviation change). Below, we elaborate on the concept of functional diversity, a potential ELRP that may be easier for fishermen to understand and embrace as an initial foray into ELRPs.

From an ecosystem perspective, it is important to understand the role of species within an ecosystem. Loss of species fulfilling an important role (akin to the keystone and foundational species ideas identified by Connell 1961 and Dayton 1972) can have a disproportionate impact on a marine ecosystem. Therefore, we believe there is potential to identify ELRPs based on functional diversity and redundancy.

Ecosystems that have high functional diversity or redundancy are often healthier and more stable than ecosystems with a low functional diversity (Tilman 2001; Hooper et al. 2005; Oliver et al. 2015). Functional diversity measures the diversity or redundancy of specific roles or traits within an ecosystem. If there are a number of species fulfilling a specific role in the ecosystem, then reduction or loss of that species may have minimal effect on the functioning of an ecosystem. Conversely, if one or a limited number of species fill a specific role in an ecosystem, then reduction or loss of that species may have an overly influential effect on the ecosystem.

Leuzinger and Rewald (2021) suggest that maintenance of ecological function is more important than protecting historical biodiversity. They argue that management needs to pay attention to how an ecosystem functions rather than which specific species are present (i.e., are all roles fulfilled and is there redundancy for those roles?). This may become more important as climate change impacts communities via changes in stock distribution and productivity of historically important species.

For example, the functional diversity of herbivore species has been shown to be important to ecosystem status and health. When populations of sea urchin (Euechinoidea) are released from predation, their abundance can denude kelp forests, creating barren areas devoid of kelp and all associated organisms (Filbee-Dexter and Scheibling 2014; Marzloff et al. 2016). In a similar example from the Caribbean, decreased abundance of herbivorous fish species led to high abundance of and dependence on a single herbivore species of sea urchin *Diadema* sp. When disease negatively impacted a large proportion of the sea urchins, algal species were able to outcompete or replace important habitat-forming corals (Lessios et al. 1984; Hughes 1994; Cramer et al. 2017).

Functional diversity is typically quantified via one of two approaches: a quantification of the distribution of traits or guilds within a community or a multivariate measurement of the relative magnitude of species similarities and differences (Cadotte 2011; Cadotte et al. 2011). An alternative approach could be a metric of phylogenetic diversity. Recent research found that in ecosystems where measures of functional diversity have not yet been developed, measurements of phylogenetic diversity (Cadotte et al. 2011) or niche overlap indices from ecosystem models could be used instead. An ELRP could be developed to monitor redundancy at key trophic levels, in functional groupings, or based on phylogenetic diversity. The ELRP benchmarks could be established to maintain redundancy (e.g., number of forage species) or phylogenetic diversity (e.g., a limit set at a percent change in phylogenetic diversity) such that crossing the benchmark results in a deeper assessment of the food web and ecosystem functioning or more precautionary management where redundancy has been lost. In extreme situations where an ecosystem becomes reliant on only one or two species for an important functional role, management may need to implement more precautionary management of those species but may also want to encourage establishment of other species that serve the same function.

Ecosystem-level reference points based on human dimensions

Ecosystem-based fisheries management acknowledges that ecosystems are social–ecological systems. Most of the National Oceanic and Atmospheric Administration (NOAA)'s regional ecosystem status reports include indicators related to human dimensions, such as economic and demographic information. However, expanding beyond these indicators is recommended. Below we briefly introduce social–ecological resilience and human well-being as possible ELRPs that would need additional research before eventual implementation. National Oceanic and Atmospheric Administration's Integrated Ecosystem Assessment approach is increasingly used to support developing socially relevant indicators that measure human well-being (Szymkowiak and Kasperski 2021; Williams et al. 2021) and build resilience in ecosystems and communities (Spooner et al. 2021).

Ecological and social resilience are interconnected, as they are both dependent on the same underlying ecosystem (Folke et al. 2010; Ojea et al. 2017). Social–ecological resilience relates to a system maintaining basic structure, function, self-organization, and adaptation to stress and change in the face of disturbance (Intergovernmental Panel on Climate Change 2007; Davidson et al. 2013; Ojea et al. 2017). Jepson and Colburn (2013) identify various indicators related to community vulnerability and resilience, including measures of population composition, housing characteristics, labor force, and poverty. Examples of social–ecological resilience are varied but many conclude that diversity of fishery catches decreases risk and increases long-term fishermen or fishing community resilience to change (e.g., Kasperski and Holland 2013; Cline et al. 2017). At community scales, aspects of community sensitivity (e.g., reliance on single versus multiple species) and adaptive capacity (e.g., access to alternative fisheries) influence resilience to climate-induced stressors (Fisher et al. 2021). For example, a retrospective study of Alaskan fishing community catch and revenue data over 34 years found that fishing communities with diverse harvest opportunities (i.e., the ability to easily change harvest to different species) had negligible to positive impacts on revenue compared with other fishing communities following a regime shift (Cline et al. 2017). Therefore, an ELRP could be identified based on a proximal level of a benchmark for the diversity of community catch, where the ELRP could be a specific number or amount of change from a historical average. A potential management response to exceeding an ELRP benchmark could be implementation of an incentive program to stimulate community catch diversity.

Economic and fishery catch metrics, including diversity of catch, are important socioeconomic information to track; however, these measures can omit social and cultural aspects of well-being (e.g., community values, traditional products, sense of place, livelihoods) that are important to social–ecological systems (Diener and Suh 1997; Armitage et al. 2012; Nelson et al. 2022). Social well-being can be defined as "a state of being with others and the natural environment that arises where human needs are met, where individuals and groups can act meaningfully to pursue their goals, and where they are satisfied with their way of life" (Armitage et al. 2012). Social and cultural aspects of well-being include varied concepts such as resource access, self-determination, community values, traditional products, sense of place, and livelihoods (Dolan and Metcalfe 2012; Breslow et al. 2017; Nelson et al. 2022).

Identifying and incorporating well-being indicators and associated benchmarks (or ELRPs) into fisheries management can be especially challenging. Many aspects of well-being are hard to measure and often require qualitative assessments via surveys of communities (Yang et al. 2015; Moore et al. 2020). In some circumstances, indicators that focus on the measurable qualities of the social–ecological system can omit or even undermine other important, hard-to-measure aspects of well-being (Breslow et al. 2017; Leong et al. 2019). For example, indicators associated with the economics of a valuable resource (e.g., commercial fishing revenue and exvessel value) can undermine the well-being of some individuals or communities by excluding their concerns (Breslow et al. 2017). Identifying diverse perspectives of stakeholders can help clarify disagreements and reveal common goals or management actions that could achieve thriving, sustainable fisheries and fishing communities (Nelson et al. 2022). Perceptions of well-being vary depending on the spatial and temporal focus of an individual; some individuals tend to focus more on the near term, while others focus on longer-term solutions, which can create adaptation actions that are uncoordinated or maladaptive. For example, near-term solutions (often referred to as coping solutions) can provide immediate relief from a challenge but at the cost of future adaptability (Nelson et al. 2022). Breslow et al. (2017) suggest including a tiered set of indicators to track well-being at various scales. For example, Szymkowiak and Kasperski (2021) worked with the community to identify appropriate indicators related to human dimensions and well-being (e.g., population decline, cost of living, and commercial fisheries harvest retained for subsistence). More work is needed to identify appropriate ELRPs for social indicators. A good place to start could be via identification of unacceptable conditions and the associated "acceptable boundaries" for well-being (Armitage et al. 2012) or defining "safe and just space for humanity" (Dearing et al. 2014), noting that these would need to be identified locally due to cultural differences of what is acceptable. The boundaries could then be translated to potential ELRPs.

Ecosystem-level reference points pros and cons

The ELRP categories that we identify have pros and cons relevant for management (Table 3). As the oceans continue to warm and experience other changes (ocean acidification, lower oxygen levels, pollution, etc.), it becomes even more important to monitor and manage ELRPs. Here we describe some of the caveats of each ELRP category.

Early warning signs of an ecosystem approaching a tipping point provide a powerful rationale for management action. Current research has identified potential generalizable, consistent, and repeatable early warning signs of approaching tipping points; however, these have not yet been shown to be consistent in complex systems. Therefore, ELRPs may need to be based on estimated tipping points specific to that ecosystem that account for regional conditions and stressors. Current analyses for identifying ecosystem-specific tipping points use complex multivariate statistics that can be difficult to understand or interpret, making it hard to identify thresholds or benchmarks and appropriate management responses and to communicate recommendations to managers.

One of the more straightforward options is ELRPs based on the productivity of the ecosystem via an ecosystem cap or ensuring ecosystem overfishing does not occur. The concept of an ecosystem cap is similar to single-species catch limits and thus should be relatively easy to understand and communicate to stakeholders and managers alike. Given that it is ecologically impossible to catch all species simultaneously at their maximum sustainable yield (e.g., Larkin 1977; Gaichas et al. 2012), implementing an ecosystem cap could foster more realistic expectations for fishery removals. Another benefit of a cap is that it creates a reason or venue for discussing trade-offs between fisheries. In addition, an appropriate cap (ELRP) should also ensure ecosystem overfishing does not occur and could ultimately lead to

TABLE 3 Pros and cons of ELRPs.

Indicator type	Pros	Cons
1. Statistical analyses of nonlinear dynamics and tipping points	Important for managing in a changing climate Early warnings of tipping points would provide a powerful rationale for management actions Can use a range of common indicators	Generalizable, consistent, and repeatable thresholds based on early warning signs have not yet been shown to work in complex marine systems System-specific thresholds from multivariate statistics and appropriate management responses can be difficult to identify, interpret, generalize, and communicate Need to watch for confounding factors (e.g., catch indicators tracking target species, not ecosystem status)
2. Ecosystem productivity	Relatively easy to calculate and explain More relatable to single-species reference points Fosters discussion of trade-offs More stable than single-species reference points Applicable in data limited situations Important for managing in a changing climate	Can mean lower catch limits relative to single-species management Can oversimplify systems (stable catch cap can obscure important food web changes) Raises importance of low-productivity species
3. Ecosystem trophic information	Applicable in data limited situations Theoretically could track one indicator Could provide early warning of ecosystem-level change Important for managing in a changing climate	Some ELRPs may be hard to explain Could oversimplify certain complex interactions within and across systems Need to watch for confounding factors (e.g., catch indicators tracking target species, not ecosystem status)
4. Biodiversity	Important for ecosystem resilience, especially in the face of a changing climate Can capture important dynamics of ecosystem functional roles	Need to understand ecosystem roles of species Harder to track Same indicator value can be obtained via multiple species configurations
6. Human dimensions	Acknowledges human aspect of ecosystems Could track economic or well-being aspects of industries	Indicators and thresholds for ELRPs not available at this time Harder to track

higher stock biomasses overall and higher catch per unit effort. Higher stock biomasses, in turn, support ecosystem stability (e.g., Holsman et al. 2020). Identification of an appropriate and acceptable ecosystem cap can be challenging but is quite tenable. Estimates based on a reduction from the sum of single-species maximum sustainable yield are simple but not likely to be accepted by stakeholders or managers. Ideally, estimates would be based on the primary production or modeled fisheries production of the system and vary depending on annual conditions, including the trophic level of catch. The current thresholds identified for the ecosystem overfishing indicators (Link and Watson 2019) are based on global analyses and could be modified based on ecosystem characteristics, such as primary production or trophic level of catch. Since an ecosystem cap will not track all components of an ecosystem, it could miss important food web changes. For example, a change at the base of a food web (e.g., a shift from a larger to a smaller copepod species) may not necessarily be apparent when looking at single-species catch limits or estimates of primary production. However, this could have large impacts on the ecosystem (Friedland et al. 2013) and the appropriate magnitude of fishery removals. Other, more nuanced ELRPs (e.g., ELRPs based on more detailed trophic information) may be worth exploring to address these concerns, although the challenge of identifying associated thresholds or benchmarks remains.

One of the biggest challenges to an ecosystem cap will be buy-in or acceptance by fishermen. Given that an ecosystem cap often results in a decrease in allowable catch for some species, fishermen and managers may be reluctant to embrace the idea due to its economic implications (especially in the short term). Many fishermen feel that U.S. fisheries management already implements overly precautionary management. In reality, this may not be any more precautionary than the current single-species framework, as single-species catch limits are often approximately equal to ecosystem-level catch estimates set at approximately 75% of the sum of single-species maximum sustainable yields (Worm et al. 2009; Patrick and Link 2015).

Ecosystem-level reference points based on trophic information (including ELRPs associated with biomass curves and ecosystem overfishing) offer some potential advantages for management, including the simplicity of tracking a single or a few indicators, feasibility in data-limited systems, and the potential to provide early or real-time warnings of ecosystem change. In addition, ELRPs associated with biomass production curves can be tracked through time to monitor resilience, identify when changes occur, track the direction of ecosystem change (i.e., further decline versus recovery), and potentially provide an early warning of ecosystem deterioration (Link et al. 2015; Pranovi et al. 2019). When the identified thresholds or benchmarks are linked to explanatory stressors, prompt attention can be applied to relevant pressures on the system and changes relative to the benchmark can be readily communicated to managers and stakeholders. Drawbacks to using cumulative biomass curves include that they are not easily understood by stakeholders, indicators can be highly variable over time, they may miss nuances within the ecosystem, and they do not necessarily provide information on causal relationships to manage.

Two of the less-developed ELRP categories are focused on biodiversity and human dimensions. Both of these categories track important aspects of the ecosystem that may be missed in the other categories. For example, it is important to explain, track, and respond to any loss of redundancy in important ecosystem functions, especially when the redundancy drops to one or two species for a function. Similarly, it is important to account for the human component, as well-being is important on its own but more so if it impacts industry willingness to adhere to regulations. More research on both of these categories is needed before ELRPs could be developed and implemented.

DISCUSSION

It is important to understand and track changes occurring at the ecosystem level to provide important context for the status of harvested species and to identify when the ecosystem may be approaching a change or tipping point (Selkoe et al. 2015; Link and Watson 2019). Thus, we encourage fishery managers to consider how ELRPs may be appropriate for their managed systems. Some ELRPs use widely available data and simple analyses that make them implementable in both data-rich and data-poor situations (see Table 2), and there are products currently available to support regional development and implementation of ELRPs. The salient points being that ample data and analyses to estimate ELRPs exists, there are extant venues to explore them, and thus, it seems wise to begin to do so.

Existing products could be on-ramps for ELRPs. The NOAA regional ecosystem status reports (also called State of the Ecosystem reports in some areas; Karnauskas et al. 2017; Craig et al. 2021; Northeast Fisheries Science Center 2021a, 2021b; Gove et al. 2022; Siddon 2022; Ortiz and Zador 2022; Ferriss and Zador 2022; Harvey et al. 2023) synthesize a suite of indicators to provide an overview of the status, trends, and possible future conditions of ecosystem components (NOAA 2022). Similarly the U.S. National Marine Sanctuaries Program has sanctuary condition reports that include portfolios of indicators to track status of ecosystem services within sanctuaries (Brown et al. 2019). Internationally, State of the Climate reports in Australia (Commonwealth Scientific and Industrial Research Organisation 2022) and Ecosystem Overviews in Europe (International Council for the Exploration of the Sea 2022) are similar products. We propose these reports are an opportune venue for presenting ELRPs, in addition to the standard types of indicators that have been tracked so far. For example, the Northeast Integrated Ecosystem Assessment team has recently integrated the Fogarty and Ryther indices (Link and Watson 2019) into the New England and Mid-Atlantic State of the Ecosystem reports (Northeast Fisheries Science Center 2021a, 2021b). Such ELRPs are even being considered in relatively data-poor situations (Link et al. 2022; Ye and Link 2023) that are also beginning to produce ecosystem reports. Additionally, the compilation of these ecosystem status reports affords an opportunity to glean data from which ELRPs can be calculated.

As noted throughout this paper, there are many potential benefits of ELRPs. For example, ELRPs may provide identification of aggregate dynamics within an ecosystem that may be overlooked if reference points are implemented on a species-by-species basis (e.g., emergent ecosystem properties, ecosystem-wide issues that impact multiple species, stocks, and fisheries) and could be used to identify when structural or systemic issues that impact ecosystems occur (e.g., Large et al. 2015; Link and Watson 2019; Fulton et al. 2022). Ecosystem-level reference points could also be an early warning of changes that would not be apparent in single-species reference points for many years, allowing for management prevention of changes to come or social preparation and adaptation (Link 2021). The ELRPs that track higher-level ecosystem functions can provide information on ecosystem conditions without needing to understand individual ecological processes (Link et al. 2015; Link 2018; Schlenger et al. 2018; Pranovi et al. 2019). Ecosystem-level reference points can help managers balance competing ecosystem objectives in management decisions by improving the ability to identify and understand potential adverse impacts on other parts of the system (Fulton et al. 2022). There are also many ELRP implementation challenges, including the identification of appropriate benchmarks and the resources to monitor ELRPs across directed and bycatch fisheries. Identification of appropriate benchmarks and subsequent management response can be difficult (but see Samhouri et al. 2017). It may be necessary to introduce ELRPs in stages. For example, managers could start by tracking ecosystem indicators, then introduce one or two ELRPs with somewhat benign management implications or changes before adding more complex ELRPs with more impactful management changes. Management responses to reaching ELRPs need not be overly prescriptive and could include actions such as increasing monitoring (survey) efforts, considering changes to catch buffers, providing necessary flexibility for adaptation, or increasing resilience of fishing communities. Where thresholds associated with ecosystem changes are difficult to estimate, benchmarks based on percent or standard deviation of change could be attempted, with an understanding that the benchmarks will be updated when more information is available. Fulton et al. (2022) show how multispecies estimates of surplus production can change based on ecosystem state. They note that setting catch caps based on a system with a lot of short-lived species will preclude recovery of that system to an ecosystem with more long-lived species. They therefore suggest that caps need to be tied to management goals and objectives. The challenges to implementing ELRPs are not insurmountable if the objectives clearly warrant their use. Though challenges remain, opportunities to explore ELRPs should be considered more widely than they have been given some of the systemic pressures currently facing marine ecosystems.

Next steps for science and management

Despite the need for continued research on ELRPs, we believe there is sufficient information for management to begin implementing ELRPs today. The ELRPs support ecosystem-based fisheries management objectives by providing a more holistic management approach that takes into account the interactions between species and the environment. Ecosystem indicators are a way to identify and provide early warning of emergent ecosystem properties that impact marine species and habitat (Link 2018). Ecosystem-level reference points can also help predict upcoming changes, including marine tipping points, and help management either proactively avoid tipping points or support social preparation and adaptation.

We acknowledge that more research (and scientific advancement in some cases) is needed in areas for each ELRP category we identify. For example, there is limited research on the existence of generalizable, consistent, and repeatable early warning signs that indicate an ecosystem is approaching a tipping point or the temporal persistence of tipping points and ecosystem-specific thresholds in the face of climate change. Focusing on economically important species and systems may help garner support for such research. Regarding ELRPs based on ecosystem overfishing and ecosystem caps, more research is needed to identify appropriate ecosystem-specific thresholds for these indicators, including improved understanding of how the trophic level of catch influences the magnitude of the ELRP, but these are relatively further along and have been implemented operationally in some instances.

There is also limited research on the use of ecosystem overfishing thresholds as warnings of change. There are many aspects of ELRPs based on trophic information that still need to be explored, including the impact of data type (e.g., survey versus catch data) on resulting threshold estimates and the type of corrections that may be needed to account for shifts in target species and trophic levels. For example, at higher trophic levels, it is possible that more straightforward indicators could fulfill the same need (e.g., median trophic level of catch may effectively equate to the trophic level inflection point on the S-curve). In general, research on the potential management ramifications of not tracking and responding to changes in ecosystem indicators is needed, including for less-developed ELRPs discussed here. One could envision a set of management strategy evaluations (e.g., Kaplan et al. 2021; Walters III et al. 2023) to test these ELRPs. Additionally, with climate change impacting marine resources in a number of different ways that are largely out of our control (Bryndum-Buchholz et al. 2021), it is critical that ELRPs are periodically reevaluated and updated, as needed, to reflect current management objectives and the best scientific information available, as well as simply monitored over time.

As the oceans continue to warm and experience other environmental changes (ocean acidification, lower oxygen levels, etc.), awareness of, and planning for, potential marine tipping points becomes ever more important. This is especially true because it is difficult to reverse a tipping point once it has occurred, and the loss of ecosystem services can be devastating to fishing communities and stakeholders. For ecosystems with known tipping points, discussion of the tipping point (including consequences of passing the tipping point) and appropriate management options to maintain current system configuration are imperative (Kelly et al. 2015; Selkoe et al. 2015). Even if an ecosystem does not have a known tipping point, implementing an ELRP, such as one of the ecosystem overfishing examples, would introduce stakeholders to the concept of an ELRP and could open the door to other ELRPs in the future as the science advances.

ACKNOWLEDGMENTS

We thank J. Samhouri, N. Coleman, E. Schnettler, and M. Ruccio for their comments on earlier versions of this paper. The scientific results and conclusions, as well as any views or opinions expressed herein, are those of the authors and do not necessarily reflect those of NOAA or the U.S. Department of Commerce.

CONFLICTS OF INTEREST STATEMENT

The authors have no conflict of interest regarding the publication of this paper.

DATA AVAILABILITY STATEMENT

Data sharing is not applicable to this article as no new data were created or analyzed in this study.

ETHICS STATEMENT

All ethical guidelines were followed and no animals were handled in the development of this study.

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