Ecological Indicators 55 (2015) 94–106

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

A benchmarking and assessment framework to operationalise ecological indicators based on time series analysis

Wolfgang Nikolaus Probst*, Vanessa Stelzenmüller

Johann Heinrich von Thünen-Institute of Sea Fisheries, Palmaille 9, 22767 Hamburg, Germany

ARTICLE INFO

Article history:
Received 31 October 2014
Received in revised form 23 February 2015
Accepted 27 February 2015

Keywords:
Assessment
Ecological indicator
Breakpoint analysis
Trend analysis
Operationalisation

ABSTRACT

The call for ecosystem considerations in marine management has instigated the use of ecosystem indicators. Many ecosystem indicators have been suggested under new policy frameworks such as the EU Marine Strategy Framework Directive or the Common Fisheries Policy. But many of these indicators are still under development and cannot be considered as yet operational for environmental assessments. A common reason for this lack of operationability is the absence of valid assessment benchmarks. This study introduces a two-stage approach for the benchmarking and assessment of time series (TSBA) against *a priori* chosen rationale of improvement or maintenance of current conditions. TSBA uses breakpoint- and trend-analysis to obtain long-term benchmarks and assess short term progress. Depending on the outcome of both analyses the action requirements for management can be determined. The method is exemplified on a case study on the size-structure of large North Sea gadoid stocks, which are considered as being sensitive to the impacts of fishing. Three out of six stocks reached their assessment benchmarks, while the three other stocks failed. TSBA is generic and can be applied to any indicator used within any marine policy assessment framework. A strength–weaknesses–opportunity–threat analysis (SWOT) investigated the advantages and disadvantages of TSBA in the context of the currently high political demand of operational ecosystem indicators. Contrary to benchmarks derived from ecological concepts or pressure-state relationship TSBA benchmarks are not specifically linked to limits of resilience or sustainability. However, TSBA may be especially useful in situations where assessment benchmarks from other sources will not be readily available or are associated with high uncertainty.

1. Introduction

The implementation of ecosystem considerations into marine management has become a high ranking political objective in recent decades (Curtin and Prellezo, 2010). There still is confusion about whether these ecosystem aspects should be incorporated into sectoral management such as fisheries (Hilborn, 2011; Rice, 2011) or rather should embedded into a new holistic management approach (Arkema et al., 2006; Leslie and McLeod, 2007).

One example of applying an ecosystem approach to sectoral management can be found in marine fisheries. In many regions of the world the implementation of an ecosystem approach to fisheries management (EAF) gained momentum in the first decade of the 21st century (Garcia et al., 2003; Hilborn, 2011; Kempf, 2010; Rice, 2011). Within Europe the EAF is implemented within the Common Fisheries Policy (CFP) and the marine strategy framework directive (MSFD) (EU-COM, 2008b; Jennings and Rice, 2011; Rätz et al., 2010). The implementation of the EAF is associated with the use of ecological indicators for assessing ecological states, the impact of pressures and the achievements of management targets (Jennings, 2005).

To facilitate the implementation of ecosystem approaches to marine management many new indicators have been proposed by the scientific community and regional seas conventions (Cardoso et al., 2010; HELCOM, 2013; Helsenfeld and Enserink, 2008; Jennings and Dulvy, 2005; OSPAR, 2013; Piet and Jennings, 2005). Currently enormous effort is spent in making these propositions operationable. An indicator is considered as operationable, if it has a clear and tangible metric supported by valid data, is sensitive to anthropogenic pressures, is easy to understand and communicate and can be assessed against a meaningful benchmark (Greenstreet et al., 2011; Rice and Rochet, 2005). While some indicators can be considered as operationable, many others are still rather conceptual and lack specific assessment benchmarks (OSPAR, 2013). Operationable indicators in turn should be defined by respective operational management objectives, which have specific, measureable, achievable, realistic and time limited (SMART) targets, such that management measures can be

* Corresponding author. Tel.: +49 40 38905202; fax: +49 40 38905263.
E-mail address: nikolaus.probst@ti.bund.de (W.N. Probst).

http://dx.doi.org/10.1016/j.ecolind.2015.02.035
1470-160X/

fitted and performance can be evaluated (Stelzenmüller et al., 2013).

Indicators can be categorized according to their intended use. The most common indicators categories are pressure- state- and response indicators (Jennings, 2005). Pressure-indicators measure the magnitude of an anthropogenic pressure, state-indicators reflect the status of an ecosystem component and response measure the effectiveness of management. Currently neither the MSFD nor the CFP categorize their indicators into these groups, but both policies put a strong emphasis on pressure and state indicators (EU-COM, 2008a, 2010).

Pressure-state relationships between indicators may be used to derive assessment benchmarks to distinguish desirable from non-desirable statuses of ecosystem components (Greenstreet et al., 2011; Large et al., 2013). Desirable statuses are the ones which are considered as in-line with “healthy and productive ecosystem” (EU-COM, 2008c) and are also referred to as ‘good environmental status’ (GEnS) (Borja et al., 2013). If indicators, reflecting the status of an ecosystem component, reach their GEnS benchmark, no further management action may be required. By identifying optimum or inflection points in pressure-state relationships targets or limits for indicators can be defined (Large et al., 2013; Rice, 2009; Samhouri et al., 2010). In some cases, however, clear relationships between pressures and states are lacking (Probst et al., 2012). This may be because the range of observed pressures and state is not wide enough to obtain a meaningful relationship (Probst et al., 2013b) or because ecological states are influenced by multiple pressures (Gimpel et al., 2013).

Another possibility to obtain assessment benchmarks for ecological indicators is the application of a theoretical concept. A prominent example for a theoretical concept used within a management framework is the maximum sustainable yield (MSY) applied to the exploitation of fished stocks (Jennings et al., 2001; Salomon and Holm-Müller, 2012). The MSY-concept is based on population dynamics and allows to define target values for two widely applied indicators in fisheries management, namely the spawning stock biomass (SSB) and the fishing mortality (F) (Lassen et al., 2014).

In the absence of theoretical concepts and valid pressure-state relationships ecological indicators may be assessed by analysing the time series of the indicator metric (Greenstreet et al., 2011, 2012; Probst et al., 2013a). An interesting approach for the assessment of indicator time series is breakpoint analysis (BPA). BPA can identify different periods of stability within a time series (Bai and Perron, 2003).

This study presents a two-stage approach for the time series-based assessment and benchmarking of ecological indicators (TSBA). TSBA is a generic and flexible assessment approach to all ecological indicators used within an assessment framework of marine policies. Within TSBA BPA is combined with a short-term trend analysis to assess long-term and short-term changes in the time series of an indicator metric. From the combination of both analysis requirements for management action can be determined. This paper explains how the choice of rationales for setting *a priori* assessment benchmarks can be guided and explore how applicable and accurate the applied time series analysis methods are.

TSBA is exemplified by a case study on large North Sea gadoids. In northern temperate waters gadoids are considered to form an important component of healthy fish communities (Daan et al., 1990; Greenstreet et al., 2011; Shephard et al., 2012) and hence the restoration of their stocks is considered a primary objective in fisheries management (Horwood et al., 2006; Köster et al., 2014; Lindegren et al., 2010). In the North Sea large gadoid species have been abundant until the end of the 1970, thereafter facing strong declines in abundance mostly related to overfishing (Fock et al., 2014; Hislop, 1996).

Table 1
Model for the exemplary time series of Fig. 1. Each segment was modelled by random normal distribution and all modelled segments then were combined. Note that the values of the exemplary time series are arbitrary units.

Period	No. of years	Mean	SD
1800–1849	50	100	10.0
1850–1859	10	75	7.5
1860–1869	10	50	5.0
1870–1879	10	30	3.0
1880–1914	35	25	2.5
1915–1919	5	40	4.0
1920–1926	7	30	3.0
1927–1933	7	20	2.0
1934–1939	6	10	1.0
1940–1944	5	15	1.5
1945–1949	5	20	2.0
1950–1959	10	15	1.5
1960–1969	10	12	1.2
1970–1979	10	8	0.8
1980–1989	10	5	0.5
1990–1999	10	3	0.3
2000–2004	5	5	0.5
2005–2010	6	8	0.8

2. Materials and methods

2.1. Design of an exemplary status time series

To test the applicability and accuracy of the applied time series analysis methods, an exemplary time series reflecting a typical trajectory of a status indicator from an exploited fish stock was modelled. The trajectory of the modelled time series was chosen to reflect the historical development of an industrially exploited groundfish stock in North European waters. The time series ranged from 1800 until 2010 and contained different phases of intensification and relaxation of exploitation pressure (Fig. 1). The time series was modelled to account for increased exploitation with the onset and progression of industrialization during the beginning of the 20th century (Bolster et al., 2011; Fock, 2014; Fock et al., 2014; Thurstan et al., 2010), and includes phases of recovery due to World War I and World War II as well as recent improvements in European fisheries management (Cardinale et al., 2013; Fernandes and Cook, 2013). With the onset of industrialization the exploitation of the stock intensified from 1850 onwards, leading to a first decline of the indicator metric. Until World War I (WWI) the exploitation remained stable. During WWI the stock could recovery due to limited fishing activities. Technological development caused further status decline from 1919 onwards, only World War II (WWII) allowed a brief period of recovery until 1990. From then improvements in fisheries management resulted in a third recovery phase.

The exemplary time series could be segmented into the ‘full’ time series (1800–2010), the ‘historic’ time series (1900–2010) and the ‘modern’ time series (1970–2010). These segments reflect different degrees of data availability for fisheries data, in which most modern surveys were instigated in the 1970 and 1980s. The time series was modelled by coercing segments of randomly distributed values (with 10% S.D.) to model the above mentioned phases (Table 1).

2.2. Developing assessment rationales

The definition of management objectives is at the core of environmental management and cannot be addressed by scientific