1Towards Ecosystem-Based Management: identifying operational food-web

2indicators for marine ecosystems

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14Abstract

15Modern approaches to Ecosystem-Based Management and sustainable use of marine

16resources must account for the myriad impacts (interspecies, human and environmental

17pressures) affecting marine ecosystems. The network of feeding interactions between co-

18existing species and populations (food webs) are an important aspect of all marine

19ecosystems and biodiversity. Here we describe and discuss a quantitative process to evaluate

20the selection of operational food-web indicators for use in evaluating marine ecosystem

21status. This process brought together experts in food-web ecology, marine ecology, and

22resource management, to identify available indicators that can be used to inform marine

23management. Standard evaluation criteria (availability of data, quality of data, conceptual

24basis, communicability, relevancy to management) were implemented to identify and

25evaluate practical food-web indicators ready for operational use and indicators that hold

26reasonable promise for future use in policy and management. It was recognized that structure

27and functioning of food webs were the major attributes for which indicators were required

28and that resilience of food webs was a key aspect of ecosystem behavior and environmental

29status. Over 60 potential food-web indicators were evaluated and the final selection of

30operational food-web indicators includes: the primary production required to sustain a

31fishery, the productivity of seabirds (or similar charismatic megafauna), zooplankton

32indicators based on community biomass, size structure and productivity, integrated trophic

33indicators (including mean trophic level, mean size, etc.), and the biomass of trophic guilds.

34It was emphasised that more efforts should be made to determine suitable reference points in

35terms of threshold identification for achieving Good Environmental Status, as well as a

36greater level of integration in the development of indicators for international use.

1. Introduction

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

62   Many aspects of ecosystem dynamics are reflected in food webs, the networks formed

63by the trophic interactions between species in ecological communities. Historically, food

64web studies developed from simply recording biological data through to a phase where

65patterns in the data were identified and catalogued. Much work has since focused on

66interpreting data and patterns, using either phenomenological or mechanistic models in food

67webs (Rossberg, 2012). Among representations of food webs in the literature are simple

68directed graphs (topological webs), flow diagrams (energy budgets), representations

69aggregated by size or trophic level, and complex dynamic models (Link et al., 2005; Piroddi

70et al., 2015). Depending on the representation, different structural and dynamic properties of

71food webs emerge from the data. The relationships between these emergent patterns are the

72subjects of much ongoing research (de Ruiter et al., 2005; Link et al., 2015; Rossberg, 2013).

73   Ecological indicators are important to EBM because they serve as proxies for several

74ecological processes (e.g. growth dynamics, energy flow) and are representations of

75ecosystem state (e.g. biodiversity, resilience). In particular, food-web indicators have become

76increasingly important as they represent ecosystem services about which policy makers and

77stakeholders are concerned. The global uses of these indicators to better inform management

78of living marine resources has continued to increase over time (Coll et al., 2008; Fay et al.,

792013; Jackson et al., 2001; Large et al., 2013; Large et al., 2015a; Levin et al., 2009). By

80addressing much of the inherent complexity of marine ecosystems, food-web indicators are

81one of the primary interfaces between policy and science. A critical step in the policy process

82is to agree on food-web indicators that are compelling, intuitive, understandable and

83defensible to all stakeholders, but also capture key food-web states and processes that

84underlie critical and complex ecosystem dynamics. Important instances of such indicators are

85those addressing emergent properties of food webs, which can be predicted without

86understanding in detail the intricate processes operating in these complex systems. This

87predictability is reflected in the existence of simplified models or representations of food

88webs addressing specific emergent properties (ICES, 2013a; Rossberg, 2013). Examples are

89representations of food webs as food chains passing energy and biomass from lower to higher

90trophic levels, representations in form of dynamically interacting aggregated groups of

91species, representations as graphs with arrows (feeding interaction) linking nodes (species),

92where a small number of top predators are supported by increasing numbers of species at

93lower-trophic levels (de Ruiter et al., 2005) or complementarily, representation of the

94distribution of community biomass over body sizes (Kerr and Dickie, 2001). It is important to

95take into account these properties in selecting food-web indicators to develop pragmatic

96indicators applicable to describe ecosystems at regional or larger scales.

97   For operational use, primary requirements are that food-web (or for that matter, any)

98indicators be sensitive, have a basis in theory and be measurable (Dale and Beyeler, 2001;

99Kershner et al., 2011; Link, 2010; Rice and Rochet, 2005a). Those indicators that are well

100studied and linked with emergent properties can address cumulative impacts, integrated

101dynamic responses, detect indirect and unintended consequences and can help to evaluate

102trade-offs in managing ecosystems. Globally, a set of best-practices is coalescing around

103indicator selection: a plethora of indicator selection criteria have been developed that identify

104key facets of indicators (Fulton et al., 2005; Garcia et al., 2000; Greenstreet and Rogers,

1052006; Greenstreet et al., 2011; ICES, 2013a, 2013b; Institute for European Environmental

106Policy (IEEP), 2005; Link, 2005; Methratta and Link, 2006a; Piet and Jennings, 2005; Rice

107and Rochet, 2005b; Rochet and Rice, 2005; Shin and Shannon, 2009; Shin et al., 2010a,

1082010b)

109   While there have been some efforts to develop operational ecological indicators to

110evaluate ecosystem status (ICES, 2015), the task of selecting specific food-web indicators has

111been difficult for a number of reasons. Food-web ecology is a relatively new area of research

112(compared to more established community ecology and population ecology) with rapidly

113emerging information and methods (Link et al., 2015; Longo et al., 2015; Thompson et al.,

1142012). In light of new methodologies, historical data is often unsuitable to calculate the

115necessary metrics to use potential food-web indicators for evaluating ecosystem status. Like

116many other types of ecological indicators, selection of a specific set of food-web indicators

117can imply that some aspects of marine food webs are valued more than others. Therefore, a

118well-balanced selection process for indicators is required that encompasses all currently

119known properties of marine food webs with the necessary data to be confidently used by both

120management and stakeholders.

121   This study aims to provide a list of operational food-web indicators that can be used to

122quantify the emergent properties of food webs in marine ecosystems. The context for this

123work was the EU's MSFD need to delineate GES with regards to food webs (Descriptor 4;

124ICES, 2014; Rogers et al., 2010), but was conducted cognizant of broader potential

125applications to assess ocean status. Here, we develop a strategy using the best available

126knowledge from scientific experts and a quantitative methodology for evaluating food-web

127indicators for implementation in EBM. We also discuss the future development of these

128indicators for practical use as reference points in management.

129**2. Methods**

130To address ongoing global requirements (Europe, North America and elsewhere), three

131objectives related to food-web indicators were explored:

132   • To determine a defined process for selecting and developing food-web

133   indicators.

134   • To develop a short list of suggested food-web indicators related to

135   management contexts (EBM) in Europe and globally.

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

137 This approach led to a two-part set of efforts to a) identify and evaluate operational food-web

138 indicators that can currently be used and b) identify food-web indicators that hold promise in

139 the future for management, but that require further development. This guidance would allow

140 for increased clarity in selecting food-web indicators coherently within and across regions

141 and lead to more defined response and pressure targets for control rules in EBM. As part of

142 this broader effort, this project was developed as part of the ICES workshop to develop food-

143 web indicators for operational use in EBM (ICES, 2014). The workshop brought together

144 international experts in food webs, marine ecology and management to identify appropriate

145 food-web indicators for current use.

146 *2.1.   Food-web indicators*

147 An initial set of 40 food-web indicators were selected from a list of over 60 candidate

148 indicators presented by the workshop experts. Presentations covered all marine functional

149 groups and all attributes of food webs that were considered necessary for a comprehensive

150 evaluation. Duplicate and technically inappropriate indicators were eliminated from the pool

151 of candidate indicators. The remaining 40 food-web indicators were grouped depending on

152 three main food-web attributes which they addressed: functional indicators linked to energy

153 flow, functional indicators linked to ecosystem resilience and structural indicators linked to

154 diversity and 'canary' species (for more detailed descriptions see Appendix A).

155 *2.2.   Selection criteria*

156 A list of 5 criteria and 13 sub-criteria (Table 1) was initially synthesized from a set of criteria

157 determined by previous working groups of experts examining ecological indicators (ICES,

158 2015; Kershner et al. 2011). These criteria were adapted to broadly examine the functionality

159 of the food-web indicators that could be operational within the global context (useful for

160 several countries and regions).

161   Each indicator was evaluated against the selection criteria and scored as 0, 1 or 2, where 0

162 = not met, 1 = partly met, and 2 = fully met. A Delphi method was used whereby sets of

163 indicators were scored by small groups based on consensus, following a discussion

164 establishing common understanding of the indicators themselves and how to apply the criteria

165 to the indicators. Each of the 13 sub-criteria was scored equally and no weighting was

166 applied. Scores were presented as percentages of the total score available (maximum score by

167 the number of categories; i.e. 2 x 13 = 26). Indicators were ranked within the agreed

168 attributes of food webs (Functioning – energy flows, Resilience - ability to recover from

169 perturbation, Structure - species organization). Particular issues or concerns with individual

170 scores were highlighted for subsequent discussions. These were then examined so that all

171 scores were adjusted through consensus-based discussions. This process was used to quantify

172 the usefulness of indicators and to aid in the final selection.

173 *2.3.   Wider consideration for selecting food-web indicators*

174 In addition to the specific criteria for each food-web indicator, a broader set of features was

175 considered through consensus of the experts involved when evaluating the final

176 recommended suite of indicators. The indicators were categorized into two groups, one set

177 that may be currently implemented and one that holds promise for future development. Key

178 considerations were:

179 **Relative ranks** within the major food-web indicator attributes informed the choice of

180 indicators, but were not adhered to in a strictly quantitative manner.

181 **Coverage of all attributes** of food webs. To the extent practicable, all three main categories

182 of food-web indicator attributes were represented.

183 **Coverage of all functional groups** found within a food web. Recognizing that much

184 indicator development has occurred for upper trophic level contexts, we ensured that lower

185 trophic level taxa were not omitted, even though as a group they may have scored lower than

186 more commonly or routinely monitored upper trophic levels.

187 **Major indicator classes (structure, function and resilience)** were as well represented as

188 possible to ensure that important facets of food webs were included.

189 **Current operability** was effectively based on an *ad hoc* review (or weighting) of operability

190 issues related to data availability, management relevance and existence of baselines, targets

191 or related reference points, which although were selection criteria, were deemed critical

192 enough to warrant additional consideration.

193 **Links to other indicator uses** were considered to ensure that we emphasized food-web

194 indicators that are unique to describing food webs. Other indicator uses include biodiversity,

195 fisheries, eutrophication and sea floor integrity.

196 **3. Results**

197 Within each attribute, indicators tended to cluster into groups with similar underlying

198 ecological theory. When selecting priority indicators for further development it was therefore

199 considered necessary to review the full list of indicators and ensure that those that clustered

200 together, but with lower scores, were also taken into consideration to maintain a diversity of

201 indicator formulations.

202   The rank scores were obtained from the unweighted sum of all 13 evaluation sub-criteria

203 (Table 2a, b). When the evaluation was re-run separately using only the first six sub-criteria

204 in Table 1 (linked to practical aspects of indicator measurement), and the next seven criteria

205 (linked to aspects of indicator implementation), there was relatively little difference in the

206 final overall outcome. This suggests that the rank scores were robust to variability in criteria

207 selection and were minimally influenced by single criteria evaluations.

208 *3.1. Energy flow indicators*

209 A relatively large number of indicators were identified which had clear links to functional

210 aspects of food webs (Table 2a). Production or biomass ratios for various parts of the food

211 web detect gross structural changes in the energy flow through a food web which may have

212 been caused by, for example, harvesting of key species, or disruption of distributional overlap

213 between predators and prey through climatic factors.

214   Total Mortality Z (Fishing mortality + natural mortality or production to biomass

215 ratio), is commonly used in the ecosystem modelling community (Ecopath with Ecosim:

216 Christensen and Pauly, 2008; Pauly et al., 2000). Despite the relatively high score this was

217 not the most easily interpretable indicator of food web functioning. This was evident in the

218 low score for the communication criteria (Table 2b). Ecosystem exploitation was considered

219 useful to describe the harvesting pattern of exploited ecosystems. It is an indicator of the

220 pressure of the fisheries on the food web.

221   Primary Production Required (PPR) to sustain a fishery has a solid conceptual basis

222 (Pauly and Christensen, 1995). However, the difficulty of explaining the concept to the lay

223 public contributed to a moderate score for this indicator. Moreover, this indicator does

224 require estimates of transfer efficiency (TE), which is generally assumed to be 10-15%

225 between trophic levels. Note that indicators of transfer efficiency themselves were not

226 selected as indicators for use immediately due to lack of data to systematically estimate TE.

227 Monitoring intermediate marine productivity and chlorophyll *a* fronts by satellite using

228remote observation was considered effective to estimate indicators of energy-flow in food

229webs.

230   Four fairly similar indicators based on trophic level were evaluated (the mean trophic

231level of the catch, the mean trophic index of the fish community, the mean trophic level of

232the community and the trophic balance index). Each has a slightly different formulation, but

233all require good quality and regularly updated data on dietary relationships, time series of

234survey catch or landings from broad regional seas to avoid local population or fleet effects,

235and accurate, agreed upon and regularly updated assessments of the trophic levels of the

236ingested food. Similarly the Trophic Balance Index, describing the fishing pattern of local

237métiers, can be useful in the context of assessing food web effects of fisheries harvesting, but

238has limited application for other pressures.

239   Low scores allocated to indicators such as the disturbance index, loss in production

240index, mean transfer efficiency and Finn Cycling Index were due to uncertainty over the

241quality of the technical assessment (data needs and rigor) and the likely ease of

242implementation. However, some of the indicators may warrant further investigation.

243*3.2.   Resilience indicators*

244It was interesting to note that the six indicators that had a link to resilience of the food web

245were generally scored lower than many other indicators (Table 2b). This may be because they

246are more conceptually complex. It was considered that the top three in this category, the

247Mean number of trophic links per species, Ecological Network Analysis derived indicators,

248and the Gini-Simpson dietary diversity index, all held promise as food-web indicators, but the

249group of experts felt that these would not be recommended as suitable for implementation in

250the short-term. The conceptual and technical difficulty of measuring food-web resilience and

251ability to recover from perturbation partly explains the low scores allocated to the assessment

252 criteria in the area of cost-effectiveness of data gathering, although they all have strong

253 support in the literature.

254   The indicators for this attribute that scored poorly (Herbivory: Detritivory Ratio,

255 Ecological Network Indices, System Omnivory Indices) will take more time to develop. The

256 complexity of their formulation also suggests that, even if further developed, they may be

257 difficult to explain in a management context. More importantly, these indicators need regular

258 diet time series data encompassing the entire food web, which have not been made widely

259 available even to support applied multispecies fishery assessments.

260 *3.3.   Structural indicators*

261 Several indicators in this category obtained relatively high scores, suggesting that managers

262 may want to use these indicators to help interpret patterns observed particularly at higher

263 trophic levels. Another important consideration is the role of aggregated sets of structural

264 indicators, such as those related to phytoplankton, zooplankton, forage fish, scavengers and

265 birds, which together have important implications for food-web resilience (e.g. low or high

266 biodiversity) as well as structure of the individual components. Many structural indicators are

267 describing the same ecosystem components in multiple ways (Table 2a, b) and due to the

268 multi-faceted uses of these indicators (in addition to characterizing food webs) the data are

269 likely to be collected and available.

270   Higher-scoring indicators were those which informed trends in absolute biomass,

271 production, or ratios of both, for a number of guild-level ecosystem components, especially

272 higher predators. For those structural indicators that aggregate across multiple components, it

273 was generally thought preferable to have indicators comprising absolute values rather than

274 ratios, as these data would be necessary anyway to interpret ratio metrics. Some of these

275 abundance-related indicators may be given a higher priority if they are also useful for

276 informing an aspect of food web resilience.   For example, both the Gini-Simpson diversity

277 indices for small and large fish and the Species Richness Index were thought to be potentially

278 useful for assessing food web resilience.

279 *3.4.* *Suggested food-web indicators*

280 The following indicators are the refined set of food-web indicators recommended for current

281 use based on the selection criteria (Table 1) and accounting for the wider considerations in

282 the selection process (Table 2a,b):

283 **Guild level biomass (and production)**

284 Guild-level biomasses and production address structural attributes of food webs, and can also

285 serve as proxies for functioning. It was noted that the typical use of this type of indicator has

286 been for fishes, but if feasible this indicator should include multiple guilds across all trophic

287 levels, such as primary producers, zooplankton, benthos, and charismatic megafauna, beyond

288 just fish or upper tropic levels. The guilds should be determined as appropriate for the taxa in

289 a given regional sea.

290 **Primary Production Required to sustain a fishery (PPR)**

291 This addresses the functioning attribute of food webs and is a measure of the ecological

292 footprint of a fishery. However, this metric can (and often does) integrate a wide range of

293 removals from the food web. Derivatives of this food-web indicator could, where feasible, be

294 contrasted to measures of primary production to ensure it is directly appraised against field

295 data. Satellite imagery makes estimates of primary production widely available (given the

296 usual caveats of remotely sensed data), and typical landings and associated data are also

297 widely available, making PPR more integrative and feasible than is often perceived.

298 **Seabird (charismatic megafauna) productivity**

299 The breeding success of seabirds addresses the structural and functional attribute of a food

300 web and can also serve as a proxy for resilience. Although particular to seabirds, especially

301 breeding success/chicks per pair, it was recognized that seabirds may not be prominent or

302 important in all regional seas. Similar productivity indicator could be calculated for marine

303 mammal taxa (i.e. pup production rates).

304 **Zooplankton size biomass index**

305 This indicator addresses both structural and functional attributes of food webs. Although

306 indicators associated with this taxonomic group were often ranked lower, they represent an

307 important part of the food web - the link between primary production at lower trophic level

308 and upper trophic level consumption and growth.

309 **Integrated trophic indicators (mean TL, mean size)**

310 Trophic indicators address both structural and resilience attributes of food webs. It was

311 critical to include an explicitly integrative measure that provided some view of the overall

312 system and did not focus on only certain facets of it. There are many possible indicators in

313 this category from which to choose, such as mean trophic level, mean, or proportion at size of

314 the community (depending upon abundance) and trophic data availability in a given regional

315 sea.

316 *3.5.   Indicators for development*

317 Food-web indicators that were recommended for future development were Ecological

318 Network Analysis indicators, the Gini-Simpson dietary diversity index and condition

319 indicators. These indicators lacked the available data to be considered currently useful for

320 management, but all were determined to be representative of multiple aspects of the food-web

321(integrated food-web perspective). Some indicators that were suggested to be currently

322operational (marine trophic level indicators, primary producers and zooplankton indicators)

323were also thought to require more development to fully meet their potential and range as

324indicators for food-web and other indicator uses.

325**4. Discussion**

326   The five food-web indicators recommended from this process cover important facets of

327food webs, particularly addressing structural, functional and resilient features of marine food

328webs (Jennings and Collingridge, 2015; Polis and Strong, 1996; Thompson et al., 2012). It is

329likely that multiple indicators are needed to track the multiple features that comprise food

330webs and delineation of GES (Large et al., 2015a, 2015b; Mallory et al., 2010; Rice and

331Rochet, 2005a) of which these five candidates are suitable options. All of the five food-web

332indicators proposed here are generally applicable in terms of capturing the main facets of

333food-web dynamics (ICES, 2014; Methratta and Link, 2006b; Shannon et al., 2009) and

334readily link to known behaviors of food webs. Many of these indicators are broad enough in

335context to be applied across many marine ecosystems (coastal, temperate, arctic, tropical,

336etc.; Andrews et al., 2013; Coll and Libralato, 2012; Fulton et al., 2005; Hayes et al., 2015;

337Parsons et al., 2008; Zador et al., 2014).

338   Yet even the five proposed indicators may not all have widely and consistently

339monitored data available to sufficiently calculate the metrics. Although important to track

340lower-trophic level dynamics and linkages to upper-trophic level taxa, the zooplankton

341indicator may not have widely collected data nor be as easily interpreted, given the high

342seasonality of these taxa (Pershing et al., 2005; Stige et al., 2014; Vargas et al., 2006). The

343integrated trophic indicators hold equal promise, but similarly may not always have measures

344of trophic level or equivalent (TL; Gaichas et al., 2012; Hornborg et al., 2013; Pranovi et al.,

3452012; Rossberg et al., 2006). Justifiable assumptions regarding TL, using common databases

346 on trophic ecology of taxa (e.g. fishbase; Froese and Pauly, 2013; Froese, 1992), may provide

347 a means to more readily calculate these indicators in the absence of local trophic data. Size-

348 based integrated indicators are an type of indicator that are less demanding on data and has

349 been found to show clearer responses in food webs (Engelhard et al., 2015; Fung et al., 2013;

350 Greenstreet et al., 2011; Shephard et al., 2011) The salient point is that there are well-studied

351 extant indicators able to track and delineate environmental status in marine food webs (Houle

352 et al., 2012). These were explored in the MSFD GES context (ICES, 2015, 2013b, 2008;

353 Shephard et al., 2014), but are generally applicable for marine conservation considerations.

354   Regardless of the specific indicator set chosen, a replicable, transparent, defendable and

355 clear process for selection is required (Dale and Beyeler, 2001; Link, 2010; Shin et al.,

356 2010a). The process demonstrated here is broadly applicable in a wide array of conservation

357 situations and it is as important as the outcomes. It is essentially a multi-criteria decision

358 analysis (Mendoza and Martins, 2006), whereupon the selection of indicators is agreed-to

359 before use in tracking ecosystem status. The criteria for indicator assessment used here are

360 sufficiently robust to be applied in a range of situations, with one of the five main criteria

361 specifically evaluating how useful a given indicator is to management. These criteria are

362 converging in the marine management context, but can be readily used in other forms of

363 natural resource management (e.g. terrestrial, estuarine). Due to the well-documented

364 quantitative and qualitative evaluation in the selection process, there is a high level of

365 confidence in the choice of the final set of indicators. This process allows for regular updates

366 and inclusion of novel information (Curtin and Prellezo, 2010; Kershner et al., 2011) while

367 maintaining a record of how selections are made. This process is general enough to be used

368 regardless of the type of ecosystem and conservation issue being considered, as long as the

369 criteria are agreed upon *a priori* (Espinosa-Romero et al., 2011; Martin et al., 2009).

370 Although similar selection processes have a wide history of use in conservation (Mendoza

371 and Martins, 2006), it could be even more widely and rigorously applied.

372   Based on the evaluation process, the food-web indicators selected in this study can offer

373 some guidance towards possible management actions. Guild-level biomass reflects measures

374 of biodiversity and structural relationships within ecosystems (Garrison and Link, 2000;

375 Rosenfeld, 2002). It can be an integrative indicator to evaluate the status of a particular guild

376 group in relation to another. For instance, lower numbers of forage fish will have direct (and

377 indirect) impacts on larger predators and seabirds (Cury et al., 2011; Garrison and Link,

378 2000) or could indicate low levels of primary productivity (Jennings and Collingridge, 2015;

379 Polivina et al., 2001). Either way, management responses to maintain forage fish could be

380 identified based on the information that guild-level indicators provide (Heath et al., 2014).

381 Similarly, integrated trophic indicators can address multiple aspects of structure, function and

382 resilience in ecosystems, where lower mean size or trophic level indicate impacts on large,

383 predatory animals (Methratta and Link, 2006b; Pauly and Watson, 2005; Rosenfeld, 2002).

384 Specific fisheries management actions with respect to changes in these indicators over time

385 could include adjusting harvest control rules for particularly overexploited guilds, but could

386 also include concentrating fishing efforts on lower age or size groups (Anderson et al., 2008).

387 Both higher-trophic (seabird and charismatic megafauna productivity) and lower-trophic

388 indicators (PPR and zooplankton index) are reflective of bottom-up processes viewed from

389 opposing ends of the food web (Cury et al., 2011; Einoder, 2009; Hilting et al., 2013). PPR is

390 an integrative indicator that represents the amount of primary productivity to sustain a

391 fishery, and offers a means to compare energy requirements across different fisheries

392 (Chassot et al., 2010; Gascuel et al., 2005). Seabird productivity is an indicator of food

393 availability (forage fish) and can also be sensitive to contaminants and environmental

394 pollutants (Mallory et al., 2010). Direct management actions to influence these indicators

395could be either top-down control rules aimed at relieving fishing pressure on lower-trophic

396species or bottom-up policies directed to improve water quality or habitat, which may also

397include improved management at land-sea interfaces (Furness and Camphuysen, 1997;

398Kendall et al., 2010; King and Baker, 2010; Mallory et al., 2010; Teichert et al., 2015).

399Specific management actions will be dependent on regional circumstances and the responses

400of the indicators to local pressures, but by using common indicators it will be possible to

401compare ecosystem status between regions and to help management at all levels (from

402regional to national to international) and to make effective decisions to improve the world's

403oceans.

404   This proposed set of candidate indicators is a start towards operationalizing the

405delineation of marine ecosystem status, but may require a few further steps before becoming

406fully operational. Food-web indicators may be interesting scientifically and relevant for

407management, but if they cannot inform management actions directly they certainly have less

408utility. Establishing decision criteria that trigger management actions for EBM requires an

409understanding of how pressure variables influence indicators, as well as the level of a

410particular pressure at which significant changes in ecosystem structure or function appear

411(Blanchard et al., 2010; Coll et al., 2010; Groffman et al., 2006; Link, 2010, 2002a; Samhouri

412et al., 2010). Such thresholds have been explored with a wide range of analytical methods,

413such as cumulative sums (CUSUM; Hinkley, 1970), sequential t-test (STARS; Rodinov,

4142004), empirical fluctuation processes (Zeileis and Kleiber, 2005), and significant zero

415crossings of piecewise regression models (Chaudihuri and Marron, 1999; Samhouri et al.,

4162012, 2010; Sonderegger et al., 2008; Toms and Lesperance, 2003; Toms and Villard, 2015)

417or generalized additive models (Large et al., 2013), all to identify the level of pressure that

418results in a significant indicator response (Andersen et al., 2009). These univariate

419relationships are useful for establishing decision criteria (Fay et al., 2013; Large et al., 2013;

420 Samhouri et al., 2010), however, they do not fully account for multiple pressures that likely

421 interact and occur concurrently. An assessment of ecosystem status based on suites of

422 indicators will be more powerful. Using multiple indicators to evaluate ecosystems will help

423 to avoid the possibility of misinterpretation which can occur when indicators are evaluated in

424 isolation (Coll and Libralato, 2012; Longo et al., 2015; Rice and Rochet, 2005b). Multivariate

425 approaches exist to detect thresholds, including translating indicator response into a surface

426 dependent on multiple pressures (i.e., fishing and environmental pressure; Frederiksen et al.,

427 2007; Large et al., 2015a; Scott et al., 2006), multivariate ordination methods (Baker and

428 King, 2010; King and Baker, 2010) and extensions of regression tree and gradient forest

429 analyses (Baker and Hollowed, 2014; Ellis et al., 2008; Large et al., 2015b; Liaw and Wiener,

430 2002; Pitcher et al., 2012; Prasad et al., 2006). Understanding how multiple pressure

431 variables concurrently influence ecosystem status, as evinced by thresholds in indicators, will

432 help to further operationalize these indicators as reference points for management.

433   Both the EU MSFD and US IEA efforts have a similar framework that includes

434 indicators as a critical part of the management decision-making process (Andrews et al.,

435 2013; Levin et al., 2009; Rogers et al., 2010; Shephard et al., 2014). Currently, many of the

436 efforts from both the MSFD and IEA frameworks assess ecosystems by using a suite of

437 indicators. Despite the frequent absence of thresholds of indicators to establish reference

438 points, there is still sufficient information and examples of using such indicators to inform

439 marine ecosystem management advice. Even qualitative and directional features of indicators

440 can and have been used operationally (Andrews et al., 2013; Espinosa-Romero et al., 2011;

441 Foley et al., 2015; Greenstreet et al., 2012; Large et al., 2015a, 2015b; Link et al., 2015;

442 Longo et al., 2015; Samhouri et al., 2012, 2010; Zador et al., 2014). Thus, the monitoring,

443 tracking and presentation of the food-web indicators proposed here can help to operationally

444 delineate GES.

445   When assessing the status of marine ecosystems, it is important to adequately

446 characterize the food web (Branch et al., 2010; Link, 2002b; Thompson et al., 2012).

447 Certainly there are other aspects of marine ecosystem status, a fact which is explicitly

448 acknowledged in the MSFD. Yet, too often the development of marine indicators neglect to

449 consider food webs (Hayes et al., 2015). Understanding food webs in ecosystems is

450 paramount because they are able to unify ecological sub-disciplines (behavior, dispersal,

451 physiology, thermodynamics etc.) and to examine interactions among guilds (Polis and

452 Strong, 1996; Rossberg, 2013; Thompson et al., 2012). Food webs are able to integrate

453 species-based and functional-based approaches to examine biomass distributions and

454 energetic flows within systems. Another key aspect of ecosystems that is encompassed by

455 food webs is resilience. A resilient system reacts only weakly to pressure, but resilience

456 might be lost with increasing pressures, leading to rapid changes to different states or

457 regimes. Such transition is thus the result of an accumulation of the disturbing effects of

458 pressures (Folke et al., 2004; Gunderson, 2000; Sasaki et al., 2015). Additionally,

459 ecosystems may exhibit legacy effects of earlier pressures (Folke, 2006; Hughes et al., 2005).

460 Despite the difficulty in studying food webs in their entirety (including large data

461 requirements and advanced computational abilities), emergent trends have been established in

462 food-web ecology at both the community (Fredriksen, 2003; Neira et al., 2009) and

463 ecosystem level (Link et al., 2015).

464   An important aim of EBM is to balance between multiple, often conflicting

465 objectives. How management actions take shape depends on all user groups involved,

466 including stakeholders, indigenous communities, fishers, tourists, NGOs, etc. (Branch et al.,

467 2006; Link, 2010; Marasco et al., 2007). The most successful implementation of EBM will

468 one where user groups are equally engaged, can agree on a set objectives, work towards

469 common economic-social-conservation management goals and ultimately overcome inertia in

470the decision making process (Arkema et al., 2006; deReynier et al., 2010; Espinosa-Romero

471et al., 2011; Leslie and McLeod, 2007; Link, 2010; Pitcher et al., 2009; Röckmann et al.,

4722015; Sandström et al., 2015). The set of indicators proposed in this study is an example of

473how such information can be used to more fully implement EBM by evaluating one facet of

474marine ecosystem objectives associated with food webs. More so, the process described here

475is an important means to explore the tradeoffs not only in selecting these indicators but also

476the underlying objectives and dynamics that each represents.

477   Ecological indicators for conservation (including food-web indicators) are useful to

478summarize complex information concerning marine ecosystem status (Cury and Christensen,

4792005; Dulvy et al., 2006; Fulton et al., 2005; Hayes et al., 2015; ICES, 2015; Methratta and

480Link, 2006b). Clearly defined, consistent metrics at the global scale can provide management

481in multiple countries with the tools to make EBM more operational (Leslie and McLeod,

4822007; Lester et al., 2010; Link, 2010; Link et al., 2011; Smith et al., 2007; Thrush and

483Dayton, 2010). As management efforts continue to implement EBM to meet conservation

484objectives, having a suite of indicators, a process to select them and ensuring that they map to

485clear management needs will remain increasingly important.

486

493 **References**

494 Andersen, T., Carstensen, J., Hernández-García, E., Duarte, C.M., 2009. Ecological
495   thresholds and regime shifts: approaches to identification. Trends Ecol. Evol. 24, 49–57.
496   doi:10.1016/j.tree.2008.07.014

497 Anderson, C.N.K., Hsieh, C., Sandin, S. a, Hewitt, R., Hollowed, A., Beddington, J., May,
498   R.M., Sugihara, G., 2008. Why fishing magnifies fluctuations in fish abundance. Nature
499   452, 835–9. doi:10.1038/nature06851

500 Andrews, K.S., Ballance, L.T., Barcelo, C., Barlow, J.P., Bellman, M.A., Steven, J., Brodeur,
501   R.D., Brown, C.J., Chivers, S.J., Cope, J.M., Crone, P.R., Beukelaer, S. De, Dereynier,
502   Y., Devogelaere, A., Dunsmore, R., Robert, L., Gertseva, V. V, Good, T.P., Gray, I.A.,
503   Haltuch, M.A., Hamel, O.S., 2013. Integrated Ecosystem Assessment of the California
504   Current.

505 Arkema, K.K., Abramson, S.C., Dewsbury, B.M., 2006. Marine Ecosystem-Based
506   Management: From Characterization to Implementation. Front. Ecol. Environ. 4, 525–
507   532.

508 Badalmenti, G., Anna, G.D., Pinnegar, J.K., Polunin, N.V.C., 2002. Size-related
509   trophodynamic changes in three target fish species recovering from intensive trawlin.
510   Mar. Biol. 141, 561–570.

511 Baker, M.E., King, R.S., 2010. A new method for detecting and interpreting biodiversity and
512   ecological community thresholds. Methods Ecol. Evol. 1, 25–37. doi:10.1111/j.2041-
513   210X.2009.00007.x

514 Baker, M.R., Hollowed, A.B., 2014. Delineating ecological regions in marine systems:
515   Integrating physical structure and community composition to inform spatial
516   management in the eastern Bering Sea. Deep Sea Res. Part II Top. Stud. Oceanogr. 109,
517   215–240. doi:10.1016/j.dsr2.2014.03.001

518 Belkin, I.M., Cornillon, P.C., Sherman, K., 2009. Fronts in large marine ecosystems. Prog.
519   Oceanogr. 81, 233–236.

520 Blanchard, J.L., Coll, M., Trenkel, V.M., Vergnon, R., Yemane, D., Jouffre, D., Link, J.S.,
521   Shin, Y.-J., 2010. Trend analysis of indicators: a comparison of recent changes in the
522   status of marine ecosystems around the world. ICES J. Mar. Sci. 67, 732–744.
523   doi:10.1093/icesjms/fsp282

524 Bondavalli, C., Ulanowicz, R.E., Bondini, A., 2000. Insights into the processing of carbon in
525   the South Florida Cypress Wetlands: A whole-ecosystem approach to using network
526   analysis. J. Biogeogr. 27, 697–710.

527 Boyd, H., Charles, A., 2006. Creating community-based indicators to monitor sustainability
528   of local fisheries. Ocean Coast. Manag. 49, 237–258.
529   doi:10.1016/j.ocecoaman.2006.03.006

530 Branch, T. a, Hilborn, R., Haynie, A.C., Fay, G., Flynn, L., Griffiths, J., Marshall, K.N.,
531   Randall, J.K., Scheuerell, J.M., Ward, E.J., Young, M., 2006. Fleet dynamics and
532   fishermen behavior: lessons for fisheries managers. Can. J. Fish. Aquat. Sci. 63, 1647–
533   1668. doi:10.1139/f06-072

534 Branch, T.A., Watson, R., Fulton, E.A., Jennings, S., McGilliard, C.R., Pablico, G.T., Ricard,
535   D., Tracey, S.R., 2010. The trophic fingerprint of marine fisheries. Nature 468, 431–5.
536   doi:10.1038/nature09528

537 Buckland, S.T., Studeny, A., Magurran, A.C., Illian, J.B., Newson, S.E., 2011. The geometric
538   mean of relative abundance indices: a biodiversity measure with a difference. Ecosphere
539   2, art100.

540 Bundy, A., Fanning, P., Zwanenburg, K.C.T., 2005. Balancing exploitation and conservation
541   of the eastern Scotian Shelf ecosystem: application of a 4D ecosystem exploitation
542   index. ICES J. Mar. Sci. 62, 503–510.

543 Bundy, A., Shannon, L.J., Rochet, M.-J., Neira, S., Shin, Y.-J., Hill, L., Aydin, K., 2010. The
544   Good(ish), the Bad and the Ugly: a tripartite classifcation of ecosystem trends. ICES J.
545   Mar. Sci. 67, 745–768.

546 Burns, T.P., 1989. Lindeman's contradiction and the trophic structure of ecosystems. Ecology
547   70, 1355–1362.

548 Caldwell, L.K., 1970. Ecosystem as a criterion for public land policy. Nat. Resour. J. 10,
549   203–221.

550 Chassot, E., Bonhommeau, S., Dulvy, N.K., Melin, F., Watson, R., Gascuel, D., Le Pape, O.,
551   2010. Global marine primary production constrains fisheries catches. Ecol. Lett. 13,
552   495–500.

553 Chaudihuri, P., Marron, J.S., 1999. SiZer for exploration of structures in curves. J. Am. Stat.
554   Assoc. 94, 807–823.

555 Christensen, N.L., Bartuska, A.M., Brown, J.H., Carpenter, S., Antonio, D., Francis, R.,
556   Franklin, J.F., Macmahon, J.A., Noss, R.F., David, J., Peterson, C.H., Turner, M.G.,
557   Woodmansee, R.G., 1996. The report of the Ecological Society of America Comittee on
558   the Scientific Basis for Ecosystem Management. Ecol. Appl. 6, 665–691.

559 Christensen, V., 1995. Ecosystem maturity - Towards quantification. Ecol. Modell. 77, 3–32.

560 Christensen, V., Pauly, D., 2008. Ecopath with Ecosim: a user's guide.

561 Christensen, V., Pauly, D., 1993. Trophic models of aquatic ecosystems, 26th ed. Worldfish.

562 Christensen, V., Walters, C.J., Ahrens, R., Alder, J., Buszowski, J., Bang, L., Cheung,
563   W.W.L., Dunne, J., Froese, R., Karpouzi, V., Kaschner, K., Kearney, K., Lai, S., Lam,
564   V., Palomares, M.L.D., Peters-mason, A., Piroddi, C., Sarmiento, J.L., Steenbeek, J.,
565   Sumaila, R., Watson, R., Zeller, D., Pauly, D., 2009. Database-driven models of the
566   world's Large Marine Ecosystems. Ecol. Modell. 220, 1984–1996.
567   doi:10.1016/j.ecolmodel.2009.04.041

568 Coll, M., Libralato, S., 2012. Contributions of food web modelling to the ecosystem approach
569   to marine resource management in the Mediterranean Sea. Fish Fish. 13, 60–88.
570   doi:10.1111/j.1467-2979.2011.00420.x

571 Coll, M., Libralato, S., Tudela, S., Palomera, I., Pranovi, F., 2008. Ecosystem overfishing in
572   the ocean. PLoS One 3, e3881. doi:10.1371/journal.pone.0003881

573 Coll, M., Shannon, L.J., Yemane, D., Link, J.S., Ojaveer, H., Neira, S., Jouffre, D., Labrosse,
574   P., Heymans, J.J., Fulton, E.A., Shin, Y.-J., 2010. Ranking the ecological relative status
575   of exploited marine ecosystems. ICES J. Mar. Sci. 67, 769–786.

576 Constable, A.J., 2011. Lessons from CCAMLR on the implementation of the ecosystem
577   approach to managing fisheries. Fish Fish. 12, 138–151. doi:10.1111/j.1467-
578   2979.2011.00410.x

579 Constable, A.J., de la Mare, W.K., Agnew, D.J., Everson, I., Miller, D., 2000. Managing
580   fisheries to conserve the Antarctic marine ecosystem: practical implementation of the

581   Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR).
582   ICES J. Mar. Sci. 57, 778–791. doi:10.1006/jmsc.2000.0725

583 Curran, K., Bundy, A., Craig, M., Hall, T., Lawton, P., Quigley, S., 2012. Recommendations
584   for Science, Management and an Ecosystem Approach to Fisheries and Oceans Canada,
585   Maritimes Region. Can. Sci. Advis. Secr. Res. Doc. 2012/061, 48.

586 Curtin, R., Prellezo, R., 2010. Understanding marine ecosystem based management: A
587   literature review. Mar. Policy 34, 821–830. doi:10.1016/j.marpol.2010.01.003

588 Cury, P., Boyd, I.L., Bonhommeau, S., Anker-Nilssen, T., Crawford, R.J., Furness, R.W.,
589   Mills, J.A., Murphy, E.J., Osterblom, H., Paleczny, M., Piatt, J.F., Roux, J.-P., Shannon,
590   L., Sydenman, W.J., 2011. Global Seabird response to forage fish depletion - one-third
591   for the birds. Science (80-.). 334, 1703–1706.

592 Cury, P., Christensen, V., 2005. Quantitative ecosystem indicators for fisheries management.
593   ICES J. Mar. Sci. 62, 307–310. doi:10.1016/j.icesjms.2005.02.003

594 Cury, P., Shannon, L., Roux, J., Daskalov, G., Jarre, a, Moloney, C., Pauly, D., 2005.
595   Trophodynamic indicators for an ecosystem approach to fisheries. ICES J. Mar. Sci. 62,
596   430–442. doi:10.1016/j.icesjms.2004.12.006

597 Daan, N., Gislason, H., Pope, J.G., Rice, J.C., 2005. Changes in the North Sea fish
598   community: evidence of indirect effects of fishing. ICES J. Mar. Sci. 62, 177–188.

599 Dale, V.H., Beyeler, S.C., 2001. Challenges in the development and use of ecological
600   indicators. Ecol. Indic. 1, 3–10. doi:10.1016/S1470-160X(01)00003-6

601 Dame, J.K., Christina, R.R., 2007. A statistical test of network analysis: can it detect
602   differences in food web properties. Ecosystems 10, 906–923.

603 de Leiva Moreno, J.I., Agostini, V.N., Caddy, J.F., Carocci, F., 2000. Is the pelagic-demersal
604   ratio from fishery landings a useful proxy for nutrient availability? A preliminary data
605   exploration for the semi-enclosed seas around Europe. ICES J. Mar. Sci. 57, 1091–1102.

606 de Ruiter, P.C., Wolters, V., Moore, J., Winemiller, K.O., 2005. Food Web Ecology: Playing
607   Jenga and Beyond. Science (80-.). 309, 68–71.

608 deReynier, Y.L., Levin, P.S., Shoji, N.L., 2010. Bringing stakeholders, scientists, and
609   managers together through an integrated ecosystem assessment process. Mar. Policy 34,
610   534–540. doi:10.1016/j.marpol.2009.10.010

611 Druon, J.N., 2010. Habitat mapping of the Atlantic bluefin tuna derived from satellite data:
612   its potential as a tool for the sustainable management of fisheries. Mar. Policy 34, 293–
613   297.

614 Druon, J.N., Fromentin, J.M., Aulanier, F., Heikkonen, J., 2011. Potential feeding and
615   spawning habitat of Atlantic bluefin tuna in the Mediterranean Sea. Mar. Ecol. Prog.
616   Ser. 439, 223–240.

617 Druon, J.N., Panigada, S., David, L., Gannier, A., Mayol, P., Arcangeli, A., Canadas, A.,
618   Laran, S., Di. Meglio, N., Gauffier, P., 2012. Potential feeding habitat of fin whales in
619   the western Mediterranean Sea: an environmentla niche model. Mar. Ecol. Prog. Ser.
620   464, 289–306.

621 Dulvy, N.K., Jennings, S., Rogers, S.I., Maxwell, D.L., 2006. Threat and decline in fishes: an
622   indicator of marine biodiversity. Can. J. Fish. Aquat. Sci. 63, 1267–1275.
623   doi:10.1139/f06-035

624 Dulvy, N.K., Polunin, N.V.C., Mill, A.C., Graham, N.A.J., 2004. Size structural change in

625 lightly exploited coral reef fish communities: evidence for weak indirect effects. Can. J.
626 Fish. Aquat. Sci. 61, 466–475. doi:10.1139/f03-169

627 Einoder, L.D., 2009. A review of the use of seabirds as indicators in fisheries and ecosystem
628 management. Fish. Res. 95, 6–13. doi:10.1016/j.fishres.2008.09.024

629 Ellis, N., Pantus, F., Welna, A., Butler, A., 2008. Evaluating ecosystem-based management
630 options: Effects of trawling in Torres Strait, Australia. Cont. Shelf Res. 28, 2324–2338.
631 doi:10.1016/j.csr.2008.03.031

632 Engelhard, G.H., Lynam, C.P., García-Carreras, B., Dolder, P.J., Mackinson, S., 2015. Effort
633 reduction and the large fish indicator: spatial trends reveal positive impacts of recent
634 European fleet reduction schemes. Environ. Conserv. 42, 227–236.
635 doi:10.1017/S0376892915000077

636 Espinosa-Romero, M.J., Chan, K.M. a., McDaniels, T., Dalmer, D.M., 2011. Structuring
637 decision-making for ecosystem-based management. Mar. Policy 35, 575–583.
638 doi:10.1016/j.marpol.2011.01.019

639 EU, 2010. Comission decision of 18 December 2009 adopting multiannual community
640 programme for the collection, management and use of data in the fisheries sector for the
641 period 2011-2013 (2010/93/EU).

642 Fasola, M., Bogliani, G., Saino, N., Canova, L., 1989. Foraging, feeding and time-activity
643 niches of eight species of breeding seabirds in the coastal wetlands of the Adriatic Sea.
644 Ital. J. Zool. 56, 61–72.

645 Fay, G., Large, S.I., Link, J.S., Gamble, R.J., 2013. Testing systemic fishing responses with
646 ecosystem indicators. Ecol. Modell. 265, 45–55. doi:10.1016/j.ecolmodel.2013.05.016

647 Finn, J., 1976. Measures of structure and functioning derived from analysis of flows. J.
648 Theor. Biol. 56, 363–380.

649 Foley, M.M., Martone, R.G., Fox, M.D., Kappel, C. V., Mease, L. a., Erickson, A.L.,
650 Halpern, B.S., Selkoe, K. a., Taylor, P., Scarborough, C., 2015. Using Ecological
651 Thresholds to Inform Resource Management: Current Options and Future Possibilities.
652 Front. Mar. Sci. 2, 1–12. doi:10.3389/fmars.2015.00095

653 Folke, C., 2006. Resilience: The emergence of a perspective for social–ecological systems
654 analyses. Glob. Environ. Chang. 16, 253–267. doi:10.1016/j.gloenvcha.2006.04.002

655 Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., Holling, C.S.,
656 2004. Regime Shifts, Resilience, and Biodiversity in Ecosystem Management. Annu.
657 Rev. Ecol. Evol. Syst. 35, 557–581. doi:10.2307/annurev.ecolsys.35.021103.30000021

658 Frederiksen, M., Furness, R., Wanless, S., 2007. Regional variation in the role of bottom-up
659 and top-down processes in controlling sandeel abundance in the North Sea. Mar. Ecol.
660 Prog. Ser. 337, 279–286.

661 Fredriksen, S., 2003. Food web studies in a Norwegian kelp forest based on stable isotope
662 (δ13C and δ15N) analysis. Mar. Ecol. Prog. Ser. 260, 71–81.

663 Froese, R., 1992. Progress Report on FishBase. ICES Counc. Meet. 852, 1–6.

664 Froese, R., Pauly, D., 2013. FishBase [WWW Document]. URL www.fishbase.org

665 Fuchs, H., Franks, P.J.S., 2010. Plankton community properties determined by nutrients and
666 size-selective feeding. Mar. Ecol. Prog. Ser. 413, 1–15.

667 Fulton, E., Smith, A., Punt, A.E., 2005. Which ecological indicators can robustly detect
668 effects of fishing? ICES J. Mar. Sci. 62, 540–551. doi:10.1016/j.icesjms.2004.12.012

669 Fung, T., Farnsworth, K.D., Reid, D.G., Rossberg, A.G., 2015. Impact of biodiversity loss on
670   production in complex marine food webs mitigated by prey-release. Nat. Commun. 6,
671   6657. doi:10.1038/ncomms7657

672 Fung, T., Farnsworth, K.D., Reid, D.G., Rossberg, A.G., 2012. Recent data suggest no further
673   recovery in North Sea Large Fish Indicator. ICES J. Mar. Sci. 69, 235–239.

674 Fung, T., Farnsworth, K.D., Shephard, S., Reid, D.G., Rossberg, A.G., 2013. Why the size
675   structure of marine communities can require decades to recover from fishing. Mar. Ecol.
676   Prog. Ser. 484, 155–171.

677 Furness, R.W., Camphuysen, K.C.J., 1997. Seabirds as monitors of the marine environment
678   726–737.

679 Gaichas, S., Bundy, a, Miller, T., Moksness, E., Stergiou, K., 2012. What drives marine
680   fisheries production? Mar. Ecol. Prog. Ser. 459, 159–163. doi:10.3354/meps09841

681 Garcia, S.M., Kolding, J., Rice, J., Rochet, M., Zhou, S., Arimoto, T., Beyer, J.E., Borges, L.,
682   Bundy, A., Dunn, D., Fulton, E.A., Hall, M., Heino, M., Law, R., Makino, M., 2012.
683   Reconsidering the Consequences of Selective Fisheries. Science (80-.). 335, 1045–
684   1047.

685 Garcia, S.M., Rice, J., Charles, A., 2015. Balanced harvesting in fisheries: a preliminary
686   analysis of management implications. ICES J. Mar. Sci. 2–11.

687 Garcia, S.M., Staples, D.J., Chesson, J., 2000. The FAO guidelines for the development and
688   use of indicators for sustainable development of marine capture fisheries and an
689   Australian example of their application. Ocean Coast. Manag. 43, 537–556.

690 Garrison, L.P., Link, J.S., 2000. Dietary guild structure of the fish community in the
691   Northeast United States continental shelf ecosystem. Mar. Ecol. Prog. Ser. 202, 231–
692   240. doi:10.3354/meps202231

693 Gascuel, D., Bozec, Y., Chassot, E., Colomb, A., Laurans, M., 2005. The trophic spectrum:
694   theory and application as an ecosystem indicator. ICES J. Mar. Sci. 62, 443–452.

695 Gascuel, D., Pauly, D., 2009. EcoTroph: modelling marine ecosystem functioning and impact
696   of fishing. Ecol. Modell. 220, 2885–2898.

697 Gislason, H., Rice, J., 1998. Modelling the response of size and diveristy spectra of fish
698   assemblages to changes in exploitation. ICES J. Mar. Sci. 55, 362–370.

699 Gowen, R., McQuatters-Gallop, A., Tett, P., Best, M., Bresnan, E., Castellani, C., Cook, K.,
700   Sherer, C., McKinney, A., 2011. Plankton indicators. A report of a defra workshop held
701   at AFBI 2nd-3d June 2011. Belfast.

702 Greenstreet, S., Rogers, S., 2006. Indicators of the health of the North Sea fish community:
703   identifying reference levels for an ecosystem approach to management. ICES J. Mar.
704   Sci. 63, 573–593. doi:10.1016/j.icesjms.2005.12.009

705 Greenstreet, S.P.R., Rogers, S.I., Rice, J.C., Piet, G.J., Guirey, E.J., Fraser, H.M., Fryer, R.J.,
706   2011. Development of the EcoQO for the North Sea fish community. ICES J. Mar. Sci.
707   68, 1–11.

708 Greenstreet, S.P.R., Rossberg, A.G., Fox, C.J., Quesne, W.J.F. Le, Blasdale, T., Boulcott, P.,
709   Mitchell, I., Millar, C., Moffat, C.F., 2012. Demersal fish biodiversity: species-level
710   indicators and trends-based targets for the Marine Strategy Framework Directive. ICES
711   J. Mar. Sci. 69, 1789–1801.

712 Groffman, P.M., Baron, J.S., Blett, T., Gold, A.J., Goodman, I., Gunderson, L.H., Levinson,

713   B.M., Palmer, M. a., Paerl, H.W., Peterson, G.D., Poff, N.L., Rejeski, D.W., Reynolds,
714   J.F., Turner, M.G., Weathers, K.C., Wiens, J., 2006. Ecological Thresholds: The Key to
715   Successful Environmental Management or an Important Concept with No Practical
716   Application? Ecosystems 9, 1–13. doi:10.1007/s10021-003-0142-z

717 Gunderson, L.H., 2000. Ecological Resilience - In Theory and Application. Annu. Rev. Ecol.
718   Syst. 31, 425–439.

719 Hayes, K.R., Dambacher, J.M., Hosack, G.R., Bax, N.J., Dunstan, P.K., Fulton, E. a.,
720   Thompson, P. a., Hartog, J.R., Hobday, a. J., Bradford, R., Foster, S.D., Hedge, P.,
721   Smith, D.C., Marshall, C.J., 2015. Identifying indicators and essential variables for
722   marine ecosystems. Ecol. Indic. 57, 409–419. doi:10.1016/j.ecolind.2015.05.006

723 Heath, M.R., Cook, R.M., Cameron, A.I., Morris, D.J., Speirs, D.C., 2014. Cascading
724   ecological effects of eliminating fishery discards. Nat. Commun. 5, 1–8.
725   doi:10.1038/ncomms4893

726 HELCOM, 2013. HELCOM core indicators. Final report of the HELCOM CORESET
727   project. Balt. Sea Environ. Proc. 136, 1–74.

728 HELCOM, 2012. Development of a set of core indicators: interim report of the HELCOM
729   CORESET project. Part A. Description of the selection process. Balt. Sea Environ. Proc.
730   129A.

731 Heymans, J.J., Guenette, S., Christensen, V., 2007. Evaluating network analysis indicators of
732   ecosystem status in the Gulf of Alaska. Ecosystems 10, 488–502.

733 Heymans, S., Coll, M., Libralato, S., Christensen, V., 2012. Ecopath theory, modelling and
734   application to coastal ecosystems, in: McLusky, D., Woanski, E. (Eds.), Tresteise on
735   Estuarine and Coastal Science. Elsevier, pp. 93–111.

736 Hilting, A.K., Currin, C.A., Kosaki, R.K., 2013. Evidence for benthic primary production
737   support of an apex predator – dominated coral reef food web. Mar. Biol. 160, 1681–
738   1695. doi:10.1007/s00227-013-2220-x

739 Hinkley, D. V., 1970. Inference about the change-point in a sequence of a random variable.
740   Biometrika 57, 1–17.

741 Hornborg, S., Belgrano, A., Bartolino, V., Valentinsson, D., Ziegler, F., 2013. Trophic
742   indicators in fisheries: a call for re-evaluation. Biol. Lett. 9, 20121050.
743   doi:10.1098/rsbl.2012.1050

744 Houle, J.E., Farnsworth, K.D., Rossberg, A.G., Reid, D.G., 2012. Assessing the sensitivity
745   and specificity of fish community indicators to management action. Can. J. Fish Aquat.
746   Sci. 69, 1065–1079.

747 Hughes, T.P., Bellwood, D.R., Folke, C., Steneck, R.S., Wilson, J., 2005. New paradigms for
748   supporting the resilience of marine ecosystems. Trends Ecol. Evol. 20, 380–6.
749   doi:10.1016/j.tree.2005.03.022

750 ICES, 2015. Report of the Working Group on Biodiversity Science (WGBIODIV) 9-13
751   February 2015. Copenhagen, Denmark.

752 ICES, 2014. Report of the workshop to develop recommendations for potentially useful food
753   web indicators (WKFooWI). Copenhagen, Denmark.

754 ICES, 2013a. Report on the working group on the ecosystem effects of fishing activities
755   (WGECO), 1-8 May 2013, Copenhagen, Denmark.

756 ICES, 2013b. Report on the working group on multispecies assessment methods (WGSAM).

757 ICES, 2011. Report on the working group on the ecosystem affects of fishing activities
758   (WGECO). Copenhagen, Denmark.

759 ICES, 2008. Report of the working group on ecosystem effects of fishing activities
760   (WGECO) May 6-13 2008, Copenhagen, Denmark.

761 Institute for European Environmental Policy (IEEP), 2005. A review of the indicators for
762   ecosystem structure and functioning. INDECO Development of Indicators of
763   Environmental Performance of Common Fisheries Policy Report.

764 Jackson, J.B.C., Kirby, M.X., Berger, W.H., Bjorndal, K.A., Botsford, L.W., Bourque, B.J.,
765   Bradbury, R.H., Cooke, R., Erlandson, J., Estes, J.A., Hughes, T.P., Kidwell, S., Lange,
766   C.B., Lenihan, H.S., Pandolfi, J.M., Peterson, C.H., Steneck, R.S., Tegner, M.J.,
767   Warner, R.R., 2001. Historical Overfishing and the Recent Collapse of Coastal
768   Ecosystems. Science (80-.). 293, 629–639.

769 Jennings, S., Collingridge, K., 2015. Predicting Consumer Biomass, Size-Structure,
770   Production, Catch Potential, Responses to Fishing and Associated Uncertainties in the
771   World's Marine Ecosystems. PLoS One 10, e0133794.
772   doi:10.1371/journal.pone.0133794

773 Jeppesen, E., Noges, P., Davidson, T., Haberman, J., Noges, T., Blank, K., Lauridsen, R.,
774   Sondergaard, M., Sayer, C., Laugaste, R., Johansson, L.S., Bjerring, R., Amsinck, S.L.,
775   2011. Zooplankton as indicators in lakes: a scientific-based plea for including
776   zooplankton in the ecological quality assessment of lakes according to the European
777   Water Framework Directive (WFD). Hydrobiologia 676, 279–297.

778 Kane, D., Gordon, S.I., Munawar, M., Charlton, M.N., Culver, D.A., 2009. The Planktonic
779   Index of Biotic Integrity (P-IBI): an approach for assessing ecosystem health. Ecol.
780   Indic. 9, 1234–1247.

781 Kendall, C., Young, M.B., Silva, S.R., 2010. Applications of Stable Isotopes for Regional to
782   National-Scale Water Quality and Environmental Monitoring Programs.
783   doi:10.1007/978-90-481-3354-3

784 Kerr, S.R., Dickie, I.M., 2001. Biomass Spectrum. Columbia University Press, New York,
785   USA.

786 Kershner, J., Samhouri, J.F., James, C.A., Levin, P.S., 2011. Selecting indicator portfolios for
787   marine species and food webs: a Puget sound case study. PLoS One 6, e25248.
788   doi:10.1371/journal.pone.0025248

789 King, R.S., Baker, M.E., 2010. Considerations for analyzing ecological community
790   thresholds in response to anthropogenic environmental gradients. J. North Am. Benthol.
791   Soc. 29, 998–1008. doi:10.1899/09-144.1

792 Large, S.I., Fay, G., Friedland, K.D., Link, J.S., 2015a. Critical points in ecosystem responses
793   to fishing and environmental pressures. Mar. Ecol. Prog. Ser. 521, 1–17.
794   doi:10.3354/meps11165

795 Large, S.I., Fay, G., Friedland, K.D., Link, J.S., 2015b. Quantifying patterns of change in
796   marine ecosystem response to multiple pressures. PLoS One 10, e0119922. doi:10.1371/
797   journal.pone.0119922

798 Large, S.I., Fay, G., Friedland, K.D., Link, J.S., 2013. Defining trends and thresholds in
799   responses of ecological indicators to fishing and environmental pressures. ICES J. Mar.
800   Sci. 70, 755–767.

801 le Fevre, J., 1986. Aspects of the biology of frontal systems. Adv. Mar. Biol. 23, 163–299.

802 Lekve, K., Ottersen, G., Stenseth, N.C., Gjosaeter, J., 2002. Length dynamics in juvenile
803   coastal Skagerrak cod: effects of biotic and abiotic factors. Ecology 83, 1676–1688.

804 Leslie, H.M., McLeod, K.L., 2007. Confronting the challenges of implementing marine
805   ecosystem-based management. Front. Ecol. Environ. 5, 540–548. doi:10.1890/060093

806 Lester, S.E., McLeod, K.L., Tallis, H., Ruckelshaus, M., Halpern, B.S., Levin, P.S., Chavez,
807   F.P., Pomeroy, C., McCay, B.J., Costello, C., Gaines, S.D., Mace, A.J., Barth, J.A.,
808   Fluharty, D.L., Parrish, J.K., 2010. Science in support of ecosystem-based management
809   for the US West Coast and beyond. Biol. Conserv. 143, 576–587.
810   doi:10.1016/j.biocon.2009.11.021

811 Levin, P.S., Fogarty, M.J., Murawski, S. a, Fluharty, D., 2009. Integrated ecosystem
812   assessments: developing the scientific basis for ecosystem-based management of the
813   ocean. PLoS Biol. 7, e14. doi:10.1371/journal.pbio.1000014

814 Liaw, A., Wiener, M., 2002. Classification and Regression by randomForest 2, 18–22.

815 Libralato, S., 2008. System Omnivory Index, in: Jorgensen, S.E., Fath, B.D. (Eds.),
816   Ecological Indicators Vol. 4 of Encyclopedia of Ecology. Elsevier, Oxford, pp. 3472–
817   3477.

818 Libralato, S., Coll, M., Tempesta, M., Santojanni, A., Spoto, M., Palomera, I., Arneri, E.,
819   Solidoro, C., 2010. Foodweb traits of protected and exploited areas of the Adriatic Sea.
820   Biol. Conserv. 143, 2182–2194.

821 Libralato, S., Coll, M., Tudela, S., Palomera, I., Pranovi, F., 2008. Novel index for
822   quantification of ecosystem effects of fishing as removal of secondary production. Mar.
823   Ecol. Prog. Ser. 355, 107–129.

824 Libralato, S., Pranovi, F., Raicevich, S., Da Pointe, F., Giovanardi, O., Pastres, R., Torricelli,
825   P., Mainardi, D., 2004. Ecological stages of the Venice Lagoon analysed using landing
826   time series data. J. Mar. Syst. 51, 331–334.

827 Lindeman, R.L., 1942. The trophic-dynamic aspect of ecology. Ecology 23, 339–417.

828 Link, J.S., 2010. Ecosystem-Based Fisheries Management: Confronting Tradeoffs, 1st ed.
829   Cambridge University Press, New York, USA.

830 Link, J.S., 2005. Translating ecosystem indicators into decision criteria. ICES J. Mar. Sci. 62,
831   569–576. doi:10.1016/j.icesjms.2004.12.015

832 Link, J.S., 2002a. What Does Ecosystem-Based Fisheries Management Mean? Fisheries 27,
833   18–21.

834 Link, J.S., 2002b. Does food web theory work for marine ecosystems? Mar. Ecol. Prog. Ser.
835   230, 1–9.

836 Link, J.S., Almeida, F.P., 2002. Opportunistic feeding of longhorn sculpin (*Myoxocephalus*
837   *octodecemspinosus*): are scallo fishery discards an important food subsidy for
838   scavengers on Georges Bank. NOAA Fish. Bull. 100, 381–385.

839 Link, J.S., Bundy, A., Overholtz, W.J., Shackell, N., Manderson, J., Duplisea, D., Hare, J.,
840   Koen-Alonso, M., Friedland, K.D., 2011. Ecosystem-based fisheries management in the
841   Northwest Atlantic. Fish Fish. 12, 152–170. doi:10.1111/j.1467-2979.2011.00411.x

842 Link, J.S., Pranovi, F., Coll, M., Libralato, S., Christensen, V., Legault, C., 2009. Exploring
843   novel metrics of ecosystem overfishing using energy budget model outputs. Fish. Cent.
844   Res. Rep. 17, 153.

845 Link, J.S., Pranovi, F., Libralato, S., Coll, M., Christensen, V., Solidoro, C., Fulton, E. a,

846   2015. Emergent Properties Delineate Marine Ecosystem Perturbation and Recovery.
847   Trends Ecol. Evol. xx, 1–13. doi:10.1016/j.tree.2015.08.011

848 Link, J.S., Stockhausen, W.T., Methratta, E.T., 2005. Food web theory in marine ecosystems,
849   in: Belgrano, A., Scharler, U.M., Dunne, J., Ulanowicz, R.E. (Eds.), Aquatic Food
850   Webs: An Ecosystem Approach. Oxford University Press, Oxford, UK, pp. 98–113.

851 Litzow, M.A., Bailey, K.M., Prahl, F.G., Heintz, R., 2006. Climate regime shifts and
852   reorganization of fish communities: the essential fatty acid limiation hypothesis. Mar.
853   Ecol. Prog. Ser. 345, 1–11.

854 Loh, J., Green, R.E., Ricketts, T., Lamoreux, J., Jenkins, M., Kapos, V., Randers, J., 2005.
855   The Living Planet Index: using species population time-series to track trends in
856   biodiversity. Philos. Trans. R. Soc. Lond. B. Biol. Sci. 360, 289–295.

857 Longo, C., Hornborg, S., Bartolino, V., Tomczak, M., Ciannelli, L., Libralato, S., Belgrano,
858   A., 2015. Role of trophic models and indicators in current marine fisheries management.
859   Mar. Ecol. Prog. Ser. 538, 257–272. doi:10.3354/meps11502

860 Lougheed, V.L., Chow-Fraser, P., 2002. Development and use of a zooplankton index of
861   wetland quality in the Laurentian Great Lakes Basin. Ecol. Appl. 12, 474–486.

862 Mallory, M.L., Robinson, S. a, Hebert, C.E., Forbes, M.R., 2010. Seabirds as indicators of
863   aquatic ecosystem conditions: a case for gathering multiple proxies of seabird health.
864   Mar. Pollut. Bull. 60, 7–12. doi:10.1016/j.marpolbul.2009.08.024

865 Marasco, R.J., Goodman, D., Grimes, C.B., Lawson, P.W., Punt, A.E., Quinn II, T.J., 2007.
866   Ecosystem-based fisheries management: some practical suggestions. Can. J. Fish. Aquat.
867   Sci. 64, 928–939. doi:10.1139/f07-062

868 Martin, J., Runge, M.C., Nichols, J.D., Lubow, B.C., Kendall, W.L., 2009. Structured
869   decision making as a conceptual framework to identify thresholds for conservation and
870   management. Ecol. Appl. 19, 1079–1090.

871 McLeod, K.L., Lubchenco, J., Palumbi, S.R., Rossenberg, A.A., 2005. Scientific concensus
872   statement on Marine Ecoysystem-Based Management. Commun. Parternesh. Sci. Sea.

873 Mendoza, G.A., Martins, H., 2006. Multi-criteria decision analysis in natural resource
874   managment: a critical review of methods and new modelling paradigms. For. Ecol.
875   Manage. 230, 1–22.

876 Methratta, E., Link, J.S., 2006a. Evaluation of quantitative indicators for marine fish
877   communities. Ecol. Indic. 6, 575–588. doi:10.1016/j.ecolind.2005.08.022

878 Methratta, E., Link, J.S., 2006b. Evaluation of quantitative indicators for marine fish
879   communities. Ecol. Indic. 6, 575–588. doi:10.1016/j.ecolind.2005.08.022

880 Meuter, F.J., Megrey, B.A., 2006. Using multispecies surplus production models to estimate
881   ecosystem-level maximum sustainable yeilds. Fish. Res. 81, 189–201.

882 Moloney, C., Jarre, a, Arancibia, H., Bozec, Y., Neira, S., Roux, J., Shannon, L., 2005.
883   Comparing the Benguela and Humboldt marine upwelling ecosystems with indicators
884   derived from inter-calibrated models. ICES J. Mar. Sci. 62, 493–502.
885   doi:10.1016/j.icesjms.2004.11.009

886 Monaco, M.E., Ulankowicz, R.E., 1997. Comparative ecosystem trophic structure of three
887   U.S. mid Atlantic estuaries. Mar. Ecol. Prog. Ser. 161, 239–254.

888 Neira, S., Moloney, C.L., Cury, P., Mullon, C., Christensen, V., 2009. Mechanisms affecting
889   recovery in an upwelling food web: The case of the southern Humboldt. Prog. Oceanogr.

890   83, 404–416.

891 Nicholson, M., Jennings, S., 2004. Testing candidate indicators to support ecosystem-based
892   management: the power of monitoring surveys to detect temporal trends in fish
893   community metrics. ICES J. Mar. Sci. 61, 35–42. doi:10.1016/j.icesjms.2003.09.004

894 Olson, B.D.B., Hitchcock, G.L., Mariano, A.J., Ashjian, C.J., Peng, G., Nero, R.W., Podest,
895   G.P., 1994. Life on the edge: Marine life and fronts. Oceanography 7, 52–60.

896 Ostrom, E., Burger, J., Field, C.B., Norgaard, R.B., Policansky, D., 1999. Revisiting the
897   Commons: Local Lessons , Global Challenges. Science (80-.). 284, 278–283.

898 Parsons, M., Mitchell, I., Butler, A., Ratcliffe, N., Frederiksen, M., Foster, S., Reid, J.B.,
899   2008. Seabirds as indicators of the marine environment. ICES J. Mar. Sci. 65, 1520–
900   1526.

901 Pauly, D., Christensen, V., 1995. Primary production required to sustain global fisheries.
902   Nature 374, 255–257.

903 Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., Jr, F.T., Series, N., Feb, N., 1998.
904   Fishing down marine food webs. Science (80-.). 279, 860–863.

905 Pauly, D., Christensen, V., Walters, C., 2000. Ecopath, Ecosim, and Ecospace as tools for
906   evaluating ecosystem impact of fisheries. ICES J. Mar. Sci. 57, 697–706.
907   doi:10.1006/jmsc.2000.0726

908 Pauly, D., Palomares, M., 2005. Fishing down marine food web: It is far more pervasive than
909   we thought. Bull. Mar. Sci. 76, 197–211.

910 Pauly, D., Watson, R., 2005. Background and interpretation of the "Marine Trophic Index" as
911   a measure of biodiversity. Philos. Trans. R. Soc. Lond. B. Biol. Sci. 360, 415–23.
912   doi:10.1098/rstb.2004.1597

913 Perez-Espana, H., Arreguin-Sanchez, F., 1999. Complexity related to behavior of stability in
914   modelled coastal-zone ecosystems. Aquat. Ecosyst. Heal. Manag. 2, 129–135.

915 Pershing, A.J., Greene, C.H., Jossi, J.W., Brien, L.O., Brodziak, J.K.T., Bailey, B.A., 2005.
916   Interdecadal variability in the Gulf of Maine zooplankton community, with potential
917   impacts on fish recruitment. ICES J. Mar. Sci. 62, 1511–1523.

918 Piet, G., Jennings, S., 2005. Response of potential fish community indicators to fishing. ICES
919   J. Mar. Sci. 62, 214–225. doi:10.1016/j.icesjms.2004.09.007

920 Piroddi, C., Teixeira, H., Lynam, C.P., Smith, C., Alvarez, M.C., Mazik, K., Andonegi, E.,
921   Churilova, T., Tedesco, L., Chifflet, M., Chust, G., Galparsoro, I., Garcia, A.C., Kämäri,
922   M., Kryvenko, O., Lassalle, G., Neville, S., Niquil, N., Papadopoulou, N., Rossberg,
923   A.G., Suslin, V., Uyarra, M.C., 2015. Using ecological models to assess ecosystem
924   status in support of the European Marine Strategy Framework Directive. Ecol. Indic. 58,
925   175–191. doi:10.1016/j.ecolind.2015.05.037

926 Pitcher, C.R., Lawton, P., Ellis, N., Smith, S.J., Incze, L.S., Wei, C.-L., Greenlaw, M.E.,
927   Wolff, N.H., Sameoto, J.A., Snelgrove, P.V.R., Cadotte, M., 2012. Exploring the role of
928   environmental variables in shaping patterns of seabed biodiversity composition in
929   regional-scale ecosystems. J. Appl. Ecol. 49, 670–679. doi:10.1111/j.1365-
930   2664.2012.02148.x

931 Pitcher, T.J., Kalikoski, D., Short, K., Varkey, D., Pramod, G., 2009. An evaluation of
932   progress in implementing ecosystem-based management of fisheries in 33 countries.
933   Mar. Policy 33, 223–232. doi:10.1016/j.marpol.2008.06.002

934 Polis, G.A., Strong, D.R., 1996. Food web complexity and community dynamics. Am. Nat.
935   147, 813–846.

936 Polivina, J.J., Howell, E., Kobayashi, D.R., Seki, M.P., 2001. The transition zone chlorophyll
937   front, a dynamic global feature defining migrationand forage habitat for marine
938   resources. Prog. Oceanogr. 49, 469–483.

939 Pranovi, F., Libralato, S., Zucchetta, M., Link, J.S., 2014. Biomass accumulation across
940   trophic levels: analysis of landings for the Mediterranean Sea. Mar. Ecol. Prog. Ser. 512,
941   201–216. doi:10.3354/meps10881

942 Pranovi, F., Link, J.S., Fu, C., Cook, A.M., Liu, H., Gaichas, S., Friedland, K.D., Rong Utne,
943   K., Benoît, H.P., 2012. Trophic-level determinants of biomass accumulation in marine
944   ecosystems. Mar. Ecol. Prog. Ser. 459, 185–201. doi:10.3354/meps09738

945 Prasad, A.M., Iverson, L.R., Liaw, A., 2006. Newer Classification and Regression Tree
946   Techniques: Bagging and Random Forests for Ecological Prediction. Ecosystems 9,
947   181–199. doi:10.1007/s10021-005-0054-1

948 Pretty, J., 2003. Social Capital and the Collective Management. Science (80-.). 302, 1912–
949   1914.

950 Rice, J., 2003. Environmental health indicators. Ocean Coast. Manag. 46, 235–259.
951   doi:10.1016/S0964-5691(03)00006-1

952 Rice, J., 2000. Evaluating fishery impacts using metrics of community structure. ICES J.
953   Mar. Sci. 57, 682–688. doi:10.1006/jmsc.2000.0735

954 Rice, J., Gislason, H., 1996. Patterns of change in the size spectra of numbers and diversity of
955   the North Sea fish assemblage, aas reflected in surveys and models. ICES J. Mar. Sci.
956   53, 1214–1225.

957 Rice, J., Rochet, M., 2005a. A framework for selecting a suite of indicators for fisheries
958   management. ICES J. Mar. Sci. 62, 516–527. doi:10.1016/j.icesjms.2005.01.003

959 Rice, J., Rochet, M., 2005b. A framework for selecting a suite of indicators for fisheries
960   management. ICES J. Mar. Sci. 62, 516–527. doi:10.1016/j.icesjms.2005.01.003

961 Rochet, M., Rice, J., 2005. Do explicit criteria help in selecting indicators for ecosystem-
962   based fisheries management? ICES J. Mar. Sci. 62, 528–539.
963   doi:10.1016/j.icesjms.2005.01.007

964 Rochet, M.-J., Collie, J.S., Trenkel, V.M., 2013. How do fishing and environmental effects
965   propogate among and within functional groups? Bull. Mar. Sci. 89, 285–315.

966 Rochet, M.-J., Trenkel, V.M., 2003. Which community indicators can measure the impact of
967   fishing? A review and proposals. Can. J. Fish. Aquat. Sci. 60, 86–99. doi:10.1139/f02-
968   164

969 Röckmann, C., van Leeuwen, J., Goldsborough, D., Kraan, M., Piet, G., 2015. The
970   interaction triangle as a tool for understanding stakeholder interactions in marine
971   ecosystem based management. Mar. Policy 52, 155–162.
972   doi:10.1016/j.marpol.2014.10.019

973 Rockstrom, J., Steffen, W., Noone, K., Persson, A., Chapin, F.S.I., Lambin, E.F., Lenton,
974   T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hughes, T.,
975   van der Leeuw, S., Rodhe, H., Sorlin, S., Snyder, P.K., Costanza, R., Svedin, U.,
976   Falkenmark, M., Karlberg, L., Corell, R.W., Falbry, V.J., Hansen, J., Walker, B.,
977   Liverman, D., Richardson, K., Crutzen, P., Foley, J.A., 2009. A safe operating space for
978   humanity. Nature 461, 472–475.

979 Rodinov, S.N., 2004. A sequential algorithm for testing climate regime shifts. Geophys. Res.
980   Lett. 31, L09204.

981 Rogers, S.I., Casini, M., Cur, P., Heat, M., Irigoe, X., Kuos, H., Scheida, M., Sko, H.,
982   Stergio, K., Trenkel, V.M., Wikner, J., Yunev, O., 2010. Marine Strategy Framework
983   Directive Task Group 4 Report Food Webs.

984 Rosenfeld, J.S., 2002. Functional Redundancy in Ecology and Conservation. Oikos 98, 156–
985   162.

986 Rossberg, A.G., 2013. Food webs and biodiversity: foundations, models, data. Wiley,
987   Oxford, UK.

988 Rossberg, A.G., 2012. Food Webs, in: Hastings, A., Gross, L. (Eds.), Encyclopedia of
989   Theoretical Ecology. University of California Press, Berkeley, CA. USA., pp. 1–13.

990 Rossberg, A.G., Farnsworth, K.D., Satoh, K., Pinnegar, J.K., 2011. Universal power-law diet
991   partitioning by marine fish and squid with surprising stability-diversity implications.
992   Proc. Biol. Sci. 278, 1617–25. doi:10.1098/rspb.2010.1483

993 Rossberg, A.G., Yanagi, K., Amemiya, T., Itoh, K., 2006. Estimating trophic link density
994   from quantitative but incomplete diet data. J. Theor. Biol. 243, 261–72.
995   doi:10.1016/j.jtbi.2006.06.019

996 Samhouri, J.F., Lester, S.E., Selig, E.R., Halpern, B.S., Fogarty, M.J., Longo, C., McLeod,
997   K.L., 2012. Sea sick? Setting targets to assess ocean health and ecosystem services.
998   Ecosphere 3, 41.

999 Samhouri, J.F., Levin, P.S., Ainsworth, C.H., 2010. Identifying thresholds for ecosystem-
1000   based management. PLoS One 5, e8907. doi:10.1371/journal.pone.0008907

1001 Sandström, A., Bodin, Ö., Crona, B., 2015. Network Governance from the top – The case of
1002   ecosystem-based coastal and marine management. Mar. Policy 55, 57–63.
1003   doi:10.1016/j.marpol.2015.01.009

1004 Sasaki, T., Furukawa, T., Iwasaki, Y., Seto, M., Mori, A.S., 2015. Perspectives for ecosystem
1005   management based on ecosystem resilience and ecological thresholds against multiple
1006   and stochastic disturbances. Ecol. Indic. 57, 395–408. doi:10.1016/j.ecolind.2015.05.019

1007 Scott, B.E., Sharples, J., Wanless, S., Ross, O., Frederiksen, M., Daunt, F., 2006. The use of
1008   biologically meaningful oceanographic indices to separate the effects of climate and
1009   fisheries on seabird breeding success, in: Boyd, I.I., Wanless, S., Camphuysen, C.J.
1010   (Eds.), Management of Marine Ecosystems. Cambridge University Press, Cambridge,
1011   UK., pp. 46–62.

1012 Shackell, N.L., Bundy, A., Nye, J.A., Link, J.S., 2012. Common large-scale responses to
1013   climate and fishing across Northwest Atlantic ecosystems. ICES J. Mar. Sci. 69, 151–
1014   162.

1015 Shannon, L.J., Coll, M., Bundy, A., Gascuel, D., Heymans, J.J., Kleisner, K., Lynam, C.,
1016   Piroddi, C., Tam, J., Travers-Trolet, M., Shin, Y.-J., 2014. Trophic level-based
1017   indicators to track fishing impacts across marine ecosystems. Mar. Ecol. Prog. Ser. 512,
1018   115–140. doi:10.3354/meps10821

1019 Shannon, L.J., Coll, M., Neira, S., 2009. Exploring the dynamics of ecological indicators
1020   using food web models fitted to time series of abundance and catch data. Ecol. Indic. 9,
1021   1078–1095. doi:10.1016/j.ecolind.2008.12.007

1022 Shephard, S., Fung, T., Houle, J.E., Farnsworth, K.D., Reid, D.G., Rossberg, A.G., 2012.
1023   Size-selective fishing drives species composition in the Celtic Sea. ICES J. Mar. Sci. 69,

1024   223–234. doi:10.1093/icesjms/fsr200

1025 Shephard, S., Fung, T., Rossberg, A.G., Farnsworth, K.D., Reid, D.G., Greenstreet, S.P.R.,
1026   Warnes, S., 2013. Modelling recovery of Celtic Sea demersal fish community size-
1027   structure. Fish. Res. 140, 91–95. doi:10.1016/j.fishres.2012.12.010

1028 Shephard, S., Reid, D.G., Greenstreet, S.P.R., 2011. Interpreting the large fish indicator for
1029   the Celtic Sea. ICES J. Mar. Sci. 68, 1963–1972. doi:10.1093/icesjms/fsr114

1030 Shephard, S., Rindorf, A., Dickey-collas, M., Hintzen, N.T., Farnsworth, K., Reid, D.G.,
1031   2014. Assessing the state of pelagic fish communities within an ecosystem approach and
1032   European Marine Strategy Framework Directive. ICES J. Mar. Sci. 71, 1572–1585.

1033 Shin, Y.-J., Bundy, A., Shannon, L.J., Simier, M., Coll, M., Fulton, E.A., Link, J.S., Jouffre,
1034   D., Ojaveer, H., Mackinson, S., Heymans, J.J., Raid, T., 2010a. Can simple be useful
1035   and reliable? Using ecological indicators to represent and compare the states of marine
1036   ecosystems. ICES J. … 67, 717–731.

1037 Shin, Y.-J., Rochet, M.-J., Jennings, S., Field, J., Gislason, H., 2005. Using size-based
1038   indicators to evaluate the ecosystem effects of fishing. ICES J. Mar. Sci. 62, 384–396.
1039   doi:10.1016/j.icesjms.2005.01.004

1040 Shin, Y.-J., Shannon, L.J., 2009. Using indicators for evaluating, comparing, and
1041   communicating the ecological status of exploited marine ecosystems. 1. The IndiSeas
1042   project. ICES J. Mar. Sci. 67, 686–691. doi:10.1093/icesjms/fsp273

1043 Shin, Y.-J., Shannon, L.J., Bundy, A., Coll, M., Aydin, K., Bez, N., Blanchard, J.L., Borges,
1044   M.D.F., Diallo, I., Diaz, E., Heymans, J.J., Hill, L., Johannesen, E., Jouffre, D., Kifani,
1045   S., Labrosse, P., Link, J.S., Mackinson, S., Masski, H., Mo, C., Neira, S., Ojaveer, H.,
1046   Abdallahi, M., Perry, I., Thiao, D., Yemane, D., Cury, P.M., 2010b. Using indicators for
1047   evaluating, comparing, and communicating the ecological status of exploited marine
1048   ecosystems. 2 . Setting the scene. ICES J. Mar. Sci. 67, 692–716.

1049 Slocombe, D.S., 1998. Lessons from experience with ecosystem-based management. Landsc.
1050   Urban Plan. 40, 31–39. doi:10.1016/S0169-2046(97)00096-0

1051 Slocombe, D.S., 1993. Implementing Ecosystem-based Management. Bioscience 43, 612–
1052   622.

1053 Smith, A.D.M., Fulton, E.A., Hobday, A.J., Smith, D.C., Shoulder, P., 2007. Scientific tools
1054   to support the practical implementation of ecosystem-based fisheries management. ICES
1055   J. Mar. Sci. 64, 633–639.

1056 Sonderegger, D.L., Wang, H., Clements, W.H., Noon, B.R., 2008. Using SiZer to detect
1057   thresholds in ecological data. Front. Ecol. Environ. 7, 190–195.

1058 Soto, K.H., Trites, A.W., Arias-Schreiber, M., 2006. Changes in diet and maternal attendance
1059   of South American Sea lions indicate changes in the marine environment and prey
1060   abundance. Mar. Ecol. Prog. Ser. 312, 277–290.

1061 Stige, L.C., Dalpadado, P., Orlova, E., Boulay, A.-C., Durant, J.M., Ottersen, G., Stenseth,
1062   N.C., 2014. Spatiotemporal statistical analyses reveal predator-driven zooplankton
1063   fluctuations in the Barents Sea. Prog. Oceanogr. 120, 243–253.
1064   doi:10.1016/j.pocean.2013.09.006

1065 Tallis, H., Levin, P.S., Ruckelshaus, M., Lester, S.E., McLeod, K.L., Fluharty, D.L., Halpern,
1066   B.S., 2010. The many faces of ecosystem-based management: Making the process work
1067   today in real places. Mar. Policy 34, 340–348. doi:10.1016/j.marpol.2009.08.003

1068 Teichert, N., Borja, A., Chust, G., Uriarte, A., Lepage, M., 2015. Restoring fish ecological

1069   quality in estuaries: Implication of interactive and cumulative effects among
1070   anthropogenic stressors. Sci. Total Environ. 542, 383–393.
1071   doi:10.1016/j.scitotenv.2015.10.068

1072 Tett, P., Carrera, C., Mills, D.K., van Leeuwen, S., Foden, J., Bresnan, E., Gowen, R., 2008.
1073   Use of a Phytoplankton Community Index to assess the health of coastal waters. ICES J.
1074   Mar. Sci. 65, 1475–1482.

1075 Tett, P., Gowen, R.J., Painting, S.J., Elliott, M., Forster, R., Mills, D.K., Bresnan, E.,
1076   Capuzzo, E., Fernandes, T.F., Foden, J., Geider, R.J., Gilpin, L.C., Huxham, M.,
1077   McQuatters-Gollop, A.L., Malcolm, S.J., Saux-Picart, S., Platt, T., Racault, M.F.,
1078   Sathyendranath, S., van der Molen, J., Wilkinson, M., 2013. Framework for
1079   understanding marine ecosystem health. Mar. Ecol. Prog. Ser. 494, 1–27.
1080   doi:10.3354/meps10539

1081 Thompson, R.M., Brose, U., Dunne, J.A., Hall, R.O., Hladyz, S., Kitching, R.L., Martinez,
1082   N.D., Rantala, H., Romanuk, T.N., Stouffer, D.B., Tylianakis, J.M., 2012. Food webs:
1083   reconciling the structure and function of biodiversity. Trends Ecol. Evol. 27, 689–97.
1084   doi:10.1016/j.tree.2012.08.005

1085 Thrush, S.F., Dayton, P.K., 2010. What can ecology contribute to ecosystem-based
1086   management? Ann. Rev. Mar. Sci. 2, 419–41. doi:10.1146/annurev-marine-120308-
1087   081129

1088 Toms, J.D., Lesperance, M.L., 2003. A tool for identifying ecological thresholds. Ecology 84,
1089   2034–2041.

1090 Toms, J.D., Villard, M., 2015. Threshold detection: matching statistical methodology to
1091   ecological. Avian Conserv. Ecol. 10, 2.

1092 Trenkel, V.M., Rochet, M.-J., 2003. Performance of indicators derived from abundance
1093   estimates for detecting the impact of fishing on a fish community. Can. J. Fish. Aquat.
1094   Sci. 60, 67–85. doi:10.1139/f02-163

1095 Ulanowicz, R.E., 2004. Quantitative methods for ecological network analysis. Comput. Biol.
1096   Chem. 28, 321–228.

1097 Ulanowicz, R.E., 1992. Ecosystem health and trophic flow networks, in: Costanza, R.,
1098   Northon, B.G., Haskell, B.D. (Eds.), Ecosystem Health: New Goals for Environmental
1099   Management. Washington, DC, pp. 190–225.

1100 van Leeuwen, S., Tett, P., Mills, D., van der Molen, J., 2015. Stratified and nonstratified
1101   areas in the North Sea: long-term variability and biological and policy implications. J.
1102   Geophys. Res. 120, 4670–4686.

1103 Vargas, C. a, Escribano, R., Poulet, S., 2006. Phytoplankton food quality determines time
1104   windows for successful zooplankton reproductive pulses. Ecology 87, 2992–2999.

1105 Vasconcellos, M., Mackinson, S., Sloman, S., Pauly, D., 1997. The stability of trophic mass-
1106   balance models of marine ecosystems: a comparative analysis. Ecol. Modell. 100, 125–
1107   134.

1108 Wanless, S., Harris, M.P., Redman, P., Speakman, J.R., 2005. Low energy values of fish as a
1109   probable cause of a major seabird breeding failure in the North Sea. Mar. Ecol. Prog.
1110   Ser. 294, 1–8.

1111 Wilderbuer, T.K., Hollowed, A.B., Ingraham, W.J., Spencer, P.D., Connors, M.E., Bond,
1112   N.A., Walters, G.E., 2002. Flatfish recruitment response to decadal climactic variability
1113   and ocean condiditons in the eastern Bering Sea. Prog. Oceanogr. 55, 235–247.

1114 Williams, R.J., Martinez, N.D., 2004. Limits to trophic levels and omnivory in complex food
1115   webs: Theory and data. Am. Nat. 163.

1116 Xu, W., Mage, J.A., 2001. A review of concepts and criteria for asessing agroecosystem
1117   health in cludding a preliminary case study of southern Ontario. Agric. Ecosyst.
1118   Environ. 83, 215–233.

1119 Zador, S., 2013. Ecosystem considerations 2013. Appendix C in Stock Assessment and
1120   Fishery Evaluation Report of the Groudfish Resources of the Bering Sea/Aleutian
1121   Islands Regions. Anchorage, AK. USA.

1122 Zador, S., Aydin, K., Barbeaux, S., Batten, S., Bengston, J., Bond, N., Cieciel, K., Fergusson,
1123   E., Fitzgerald, S., Fritz, L., Gann, J., Garcia-reyes, M., Greig, A., Hatch, S., Hebert, K.,
1124   Hermann, A., Ladd, C., Lee, J., Litzow, M., Ortiz, I., Overland, J., Piatt, J., Ream, R.,
1125   Renner, H., Ressler, P., Salo, S., Stabeno, P., Sterling, J., Stockhausen, W., Sydeman,
1126   B., Thompson, A., Towell, R., Urban, D., Wand, M., Wertheimer, A., Whitehouse, A.,
1127   Williams, J., Yasumiishi, E., 2014. Eastern Bering Sea 2014 Report Card.

1128 Zeileis, A., Kleiber, C., 2005. Validating multiple structural change models: a case study. J.
1129   Appl. Econom. 20, 685–690.

1130

1132 Table 1. Criteria and sub-criteria used in the selection process for operational food-web
1133 indicators.

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

1134 Table 2a. Assessment of food-web indicators for Energy flow indicators against the criteria in Table 1. A maximum score for
1135 Availability of data= 6, Quality of data = 6, Conceptual = 6, Communication = 2, Management = 6 (maximum total score = 26).
1136 Asterisks (*) denote food-web indicators that were selected for current use.

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

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1141 Table 2b. Assessment of food-web indicators for Ecosystem resilience and Structural indicators against the criteria in Table 1. A
1142 maximum score for Availability of data= 6, Quality of data = 6, Conceptual = 6, Communication = 2, Management = 6 (maximum
1143 total score = 26). Asterisks (*) denote food-web indicators that were selected for current use.

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

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

1144**Appendix A:**

1145Indicators that were examined (details, history, rationale).

1146 **Functional Indicators linked to Energy Flow**

1147 **Seabird breeding success**

1148Many species of seabirds feed on lower trophic level forage species such as krill, squid, and
1149pelagic fish. Seabirds summarize changes in these forage species communities that are often
1150linked to patterns of exploitation (Cury and Christensen, 2005; Cury et al., 2011). Seabird
1151breeding success has been consistently monitored across many ecosystems and provides robust
1152estimates of both forage fish abundance and success of charismatic species (Cury et al., 2011).
1153Seabird breeding success can be a useful indicator, however, it may overlap with other measures
1154of forage fish success.

1155 **Productivity (production per unit biomass) of key predators**

1156Metrics characterizing productivity of predators at high trophic levels have been identified by
1157Rogers et al. (2010) as an important class of food-web indicators. They argued that "[t]he
1158abundance of species in the food web will generally be determined by the abundance of suitable
1159prey taxa on which they can feed. Some species, or groups of species, may play a significant part
1160in food web dynamics and so their population status will effectively summarize the main
1161predator-prey processes in the part of the food web that they inhabit." Food quantity or quality is
1162known to affect survival and reproduction of many marine species including birds (Wanless et
1163al., 2005), mammals (Soto et al., 2006) and fishes (Litzow et al., 2006). It has been argued (Boyd
1164and Charles, 2006; Cury et al., 2011; Rogers et al., 2010) that required prey abundance to
1165quantitatively and qualitatively sustain viable populations of predators constitutes a threshold
1166value which can serve as a reference point for productivity based indicators. "Productivity
1167(production per unit biomass) of key species or trophic groups" was listed among the Criteria for
1168GES by the EC (EU, 2010). Among others, it has been implemented in form of the HELCOM
1169(2013) core indicators "Pregnancy rates of marine mammals", "White-tailed eagle productivity",
1170"Abundance of sea trout spawners and parr", and "Abundance of salmon spawners and smolt".

1171 **Mean weight at age of predatory fish species from data**

1172Fish weight and condition metrics provide information on state (e.g., food limitation) in an
1173ecosystem. The indicator proposed by Shephard et al. (2014) describes the average "weight
1174anomaly" for the pelagic fish community in a given year, which is the deviation around an
1175observed long-term mean. The youngest and oldest age groups of each stock are excluded to
1176avoid sampling bias. Values are then averaged over all ages for each stock to obtain a mean
1177annual anomaly for that stock. Stock anomalies are then averaged by year to obtain a regional
1178mean weight anomaly for the whole pelagic or predatory fish communities, respectively, where
1179indicator values should fluctuate around zero in the long-term. The comparison between species

1180and stocks can give additional information on whether food becomes limiting in general or
1181whether just some species or trophic guilds are impacted.

1182Changes in this indicator can be caused by changes in food availability as well as an increase or
1183decrease in predator populations. The demand for food can be also influenced by temperature.
1184Therefore, the indicator should only be interpreted in conjunction with additional information
1185(e.g. biomass of forage fish, benthos, sea temperature, predator abundance, etc.). The indicator
1186will respond predominantly to non-anthropogenic impacts and to a lesser degree to indirect
1187anthropogenic impacts through food limitation.

1188 Total Mortality Z (Production:Biomass ratio)

1189Total mortality has a large effect on both year-to-year survival, long term reference points such
1190as F_{MSY}, and resilience. If mean weight of species in the stock and catch remain constant over
1191time, this indicator is conceptually equivalent to production/biomass. Further, the inverse of total
1192mortality is a direct indicator of longevity, an indicator which is often more readily
1193communicated outside the scientific community. It responds to management through direct
1194fishing mortality and the abundance of predatory fish (ICES, 2013b).

1195 Primary Production Required to support fisheries

1196The energy contained in solar radiation lead to primary production (PP) by phytoplankton and so
1197fuels marine ecosystems. Subsequently, energy is transferred through food webs by predation
1198and lost through metabolic processes. Ecosystem production results from the conversion of
1199organic matter at each trophic level and is dependent on ecological features such as the number
1200of feeding links, the efficiency of energy transfer from one trophic level to the next, and
1201temperature (Chassot et al., 2010). Production available to fisheries depends upon fishing
1202mortality and targeted trophic levels in the food web. Fisheries focusing only on lower trophic
1203levels may be energetically more efficient than those focused on top predators (Gascuel and
1204Pauly, 2009; Pauly and Christensen, 1995).

1205Primary Production Required (*PPR*) is the primary production and detritus flows from TL 1 that
1206are required to sustain fisheries (expressed as t/km²/year). This allows the evaluation and
1207comparison of fishing activities across ecosystems. The *PPR* is obtained by calculating the flows
1208backwards, expressed in primary production and detritus equivalents, for all pathways from the
1209caught species down to the primary producers and detritus. The *PPR* increases with fishing
1210intensity. *PPR* has been analyzed also in reference to PP, to reflect a percentage of PP used to
1211sustain catches.

1212 Productive pelagic habitat index (chlorophyll fronts)

1213Productive fronts (chlorophyll-a fronts) are key large-scale features in marine ecosystems since
1214they last long enough to sustain zooplankton production and are considered among the main

1215vectors of ocean productivity along the food chain (Belkin et al., 2009; Druon et al., 2012, 2011;
1216le Fevre, 1986; Olson et al., 1994; Polivina et al., 2001).

1217The frequency of chlorophyll-a fronts with an intermediate range of chlorophyll-a content
1218identifies the productive features that attract top-predators, i.e. areas of efficient energy transfer
1219between trophic levels outside of low and high chlorophyll levels (from about 0.1 to 3.0 mg.m^{-3}).
1220Indeed, high chlorophyll levels potentially correspond to eutrophic areas where the food chain is
1221disrupted and primary production is not available to upper trophic levels. Nutrient availability,
1222hydrological and atmospheric forcing are captured by this indicator, but it captures in particular
1223the variability of ecosystem productivity available to high tropic levels, independently of fishing
1224pressure.

1225The indicator of pelagic productivity results from the demonstrated links between top-predators
1226and chlorophyll-a fronts observed for fast-moving predators such as Atlantic bluefin tuna
1227(Druon, 2010; Druon et al., 2011), and fin whale (Druon et al., 2012) and demersal nurseries in
1228the Mediterranean Sea. The generic index of productive pelagic habitats yet requires a formal
1229validation at European scale (https://fishreg.jrc.ec.europa.eu/fish-habitat).

1230   **Ecosystem exploitation (fisheries)**

1231This estimates the level of exploitation, integrated over all trophic levels, as the total yield
1232divided by total production for all exploited species. Required data: Yield, biomass and
1233production to biomass ratio for each exploited species.

1234   **Community Condition**

1235Community condition is a measure of the overall condition (average weight at length) at the
1236functional group level, and the overall community condition. Condition reflects food availability:
1237fish are heavier for their length when food abundance is plentiful and/or competition for food is
1238low and lighter when food abundance is low and/competition for food is high. It is a reflection of
1239energy flow, food availability and resilience.

1240   **Mean trophic level of the catch**

1241Mean trophic level of the catch is one of a suite of trophic level indicators that is based on the
1242average biomass weighted trophic level across all species. Initial work considered the mean
1243trophic level of the catch, based on fishery-dependent catch or landing statistics (Pauly et al.,
12441998). It describes the average trophic level at which species are removed by the fisheries. As
1245more valuable upper-trophic level fish stocks are depleted, fishers may target lower-value, lower-
1246trophic level fish stocks (Pauly et al., 1998). Recent work suggests that this indicator is a better
1247indicator of fishing pattern and pressure than an indicator of ecosystem state (Shannon et al.,
12482014).

Marine trophic index of the community (MTI)

1249

1250 The marine trophic index (MTI) (Pauly and Watson, 2005) is another trophic level indicator,
1251 calculated with a cut-off point of trophic level greater than 3.25. Originally calculated from
1252 fisheries landings data, here it is presented as the MTI of the community, based on scientific
1253 survey data, and is considered an indicator of food web functioning (Shannon et al., 2014). It has
1254 most commonly been applied to fish (and cephalopods), but could be extended to a wider range
1255 of taxa. The marine trophic index of the community, like the mean trophic level of the
1256 community (see below), provides a measure of ecosystem integrity and resilience. Declining
1257 trophic levels may result in shorter food chains, which may leave ecosystems less able to cope
1258 with natural or human-induced change.

Mean trophic level of the community

1259

1260 Average trophic level (TL) obtained from fishery-independent surveys is a commonly used
1261 metric that can be used to measure status and trends of ecosystem structure and functioning(Shin
1262 et al., 2010). Average TL of the community is expected to decrease in response to fishing, as
1263 fisheries tend to target species at upper trophic levels (Pauly et al., 1998). Additionally, fishing
1264 can also change the structure of marine food webs by reducing the mean TL and might also
1265 influence ecosystem functioning by shortening the length of food chains and releasing predation
1266 on lower trophic level organisms (Shin et al., 2010).

Disturbance index

1267

1268 The disturbance index (DI) measures the change in trophic (or size) structure of the ecosystem
1269 and is calculated as the sum, across all TLs ≥2 (or size classes), of the absolute difference in the
1270 relative biomass (B_{TL}/B_{Total}) within each TL for each year, relative to a reference period (Bundy
1271 et al. 2005). The reference period can represent a preferred state of the ecosystem, an ideal state,
1272 a theoretical state estimated from an ecosystem model or the beginning to the time period for
1273 which there is data. The DI has been shown to respond directly to fishing pressure, but may also
1274 be affected by other pressures such as environmental change.

1275 The DI was originally proposed as one of 4 indicators comprising a 4D ecosystem exploitation
1276 index (Bundy et al., 2005).

Loss in secondary production index (L index)

1277

1278 The decrease in secondary production was proposed as a proxy for quantifying ecosystem effects
1279 of fishing on the basis of a theoretical development and application to a large set of data
1280 (Libralato et al., 2008). The L index is calculated by integrating the primary production required
1281 to sustain the catches (PPR: Pauly and Christensen, 1995) relative to the primary production (PP)
1282 in the ecosystem, taking account of transfer efficiencies (TE, i.e., the efficiency in the transfer of
1283 energy from a trophic level to another; Lindeman, 1942) and the trophic level of the catches
1284 (TLc; Pauly et al., 1998). Theoretically, these inputs can be combined to measure the loss in

1285secondary production due to fishing (L index) and to evaluate ecosystem effects of all fished
1286species (Libralato et al., 2008).

1287The application of the L index to a set of well-studied models allowed a probability of being
1288sustainably fished (P_{sust}) to be associated with each L index value, and, by fixing desired
1289sustainability levels (e.g., 75% and 95%) it provide the basis for back-estimating the associated
1290Ecosystem-based Maximum Sustainable Catches (EMSC) (Libralato et al., 2008).

1291Thus L index is formally defined as an index of ecosystem overfishing and allows application of
1292the index using both landings data and ecosystem models. L index can give rough estimates of
1293overfishing status and management advice measures allowing definition of a region of viable
1294solutions (Cury et al., 2005). L index quantification can be adapted to specific spatial scales
1295(regional spatial assessment) and to large pelagic areas exploiting data from satellite for
1296estimating PP, catches and available data on diets (for TL estimates).

1297 Cumulative distribution of biomass assessment

1298Accumulation of biomass has been documented for many marine food webs, with the
1299intermediate TLs exhibiting the largest increase in the system cumulative biomass (Gascuel and
1300Pauly, 2009; Link et al., 2009). Changes in this accumulation may reflect shifts in the ecosystem
1301structure and function. According to these observations, from a theoretical point of view, a
1302perturbed ecosystem should lower the stored, cumulative biomass and "stretch out" across TLs.
1303To describe and quantify these changes, the biomass distribution across TLs is fitted to a logistic
1304nonlinear regression model to estimate the main curve parameters: steepness (that is the slope of
1305the tangent passing through the inflection point), inflection TL (that is the projection of the
1306inflection point on the x-axis), inflection CumB (that is the projection of the inflection point on
1307the y-axis), and the basal biomass (that is the y-axis intercept of the fitted curve). Tests, carried
1308out by using both surveys and landings data, showed that the method is robust to possible
1309'sampling errors' (in terms of TL assignment), sensitive to both environmental and anthropogenic
1310drivers, and when applied to fishery dependent data, responsive (Pranovi et al., 2014, 2012).

1311 Trophic balance index (fishing pattern)

1312This index measures the evenness (pattern) of exploitation across TLs by comparing their
1313exploitation rates, which are estimated as the sum of yield (Y) divided by the sum of production
1314(P) at each TL. The evenness of exploitation is then given by the coefficient of variation of all Y/
1315P. Required data: Yield, biomass and P/B for each species in the yield.

1316 Mean transfer efficiency for a given TL or size

1317The transfer efficiency (TE_{TL}) is defined as the fraction of production that is passed from one
1318integer trophic level to the next (Lindeman, 1942; Pauly and Christensen, 1995). It is thus
1319quantifiable as the ratio between the production of the trophic level (TL) and the production at
1320the precedent trophic level (TL-1). Several studies have estimated the pattern of TE by different

1321trophic level after Lindeman's work (Burns, 1989; Lindeman, 1942). It has been used as a
1322diagnostic indicator in some cases (e.g. Libralato et al., 2004) but in most instances the
1323ecosystem average is used as an integrated summary statistic.

Finn Cycling Index

1325The Finn's cycling index (FCI: Finn, 1976) is the proportion of the total sum of flows in the food
1326web that is recycled in the system. It is measured as the proportion of the total flow that is
1327flowing within circular pathways. Recycling is considered to be an indicator of an ecosystem's
1328ability to maintain its structure and integrity through positive feedback and is used as an
1329indicator of stress and maturity (Christensen, 1995; Monaco and Ulankowicz, 1997; Ulanowicz,
13301992; Vasconcellos et al., 1997). FCI is an indicator of the recovery time of an ecosystem
1331through development of routes to conserve nutrients. A high FCI would mean the system would
1332recover faster from a perturbation, whereas a system would be expected to take longer to recover
1333(lower FCI) when it is in a more degraded state.

Functional Indicators linked to ecosystem resilience

Mean trophic links per species

1336The mean number of trophic links per species reflects how connected a food web is and,
1337potentially, how stable a food web may be (Link, 2002; Link, 2005; Methratta and Link, 2006).
1338Changes to this indicator reflect notable differences in the structure and dynamics of a food web.
1339As an understanding of temporal and spatial characteristics of marine trophic interactions it may
1340not be entirely complete. This index should be used only as a tool to invoke further precautionary
1341action (Link, 2005).

Ecological Network Analysis derived indicators (overall mean Transfer Efficiency)

1344The mean transfer efficiency (TE_m) for the food web is calculated as the geometric mean of
1345transfer efficiencies for each of the integer trophic levels II to IV from models (Christensen and
1346Pauly, 2008; Christensen et al., 2009). It is a variant of the mean transfer efficiency discussed
1347above. There have been attempts to estimate average TE also on the basis of catches over trophic
1348levels on the assumption that fisheries were in balance for some periods (Pauly and Palomares,
13492005) – which would provide a fishing pressure indicator. Average transfer efficiency by
1350ecosystem type based on model outputs have shown some variability across ecosystem types
1351(Libralato et al., 2008) and other pressures as shown in Heymans et al. (2012). The indicator has
1352been proposed as a descriptor of ecosystem health in lakes (Xu and Mage, 2001).

Gini-Simpson dietary diversity index

1354The Gini-Simpson dietary diversity index is defined as the average, over a representative sample
1355of consumer species, of the Gini-Simpson diversity of the contributions of resource species to
1356consumer diets, by volume or biomass (ICES, 2013b; Rossberg, 2013; Rossberg et al., 2011). It

1357 can be determined from stomach-content data. The metric attains values between 0 and 1, with 0
1358 implying no diversity and 1 high diverse. In practice it is computed as

1359
$$1 - \frac{\sum_{ij}\left[(p)_{ij}\right]^2}{\sum_{ij}p_{ij}} \quad p_{ij} \; i \quad j \qquad D_{diet} = 1 - \sum_{ij}\frac{\left[p\dot{\iota}\dot{\iota}ij\right]^2}{\sum_{ij}p_{ij}}\dot{\iota},$$

1360 where p_{ij} is the proportional contribution of species i to the diet of j, and the sums run over all
1361 diet items resolved to species level (Rossberg, 2013; Rossberg et al., 2011). The indicator may
1362 be applied to any component of the ecosystem for which diet data is available, but has so far
1363 been computed only for fish (Rossberg et al., 2011). A target for the metric near 0.5 has been
1364 proposed (Rossberg, 2013; Rossberg et al., 2011) based on theory and observation data. The
1365 indicator may respond to pressures (e.g. Rossberg et al., 2011).

1366 Herbivory : detritivory ratio

1367 This indicator, proposed by (Ulanowicz (1992), is the ratio of the values of the detritivory flow
1368 (from detritus to level II) divided by the value for the herbivory flow (from primary producers to
1369 level II). It is sometimes presented as H/D (or abbreviated HDR). This indicator was inspired by
1370 Lindeman (1942) when he referred to the role of saprophageous organisms and heterotrophic
1371 bacteria. This ratio has already been tested as a candidate for defining functional indicators of the
1372 food web, but results seem to be case sensitive. For example, Ulanowicz (1992) observed a
1373 higher H/D ratio in disturbed situations whereas Dame and Christina (2007) observed exactly the
1374 opposite trend. Then the disturbed situation showed a shift to a more detritus-based food web.

1375 Ecological Network indices of ecosystem status and change (Ulankowicz)

1376 Redundancy *(R)* (Monaco and Ulankowicz, 1997) indicates the system's energy in reserve it
1377 describes the distribution of energy flow among the ecosystem pathways, and is an indicator of
1378 change in the degrees of freedom of the system (Heymans et al., 2007). Based on the description
1379 of *R* by (Ulanowicz, 2004), who suggested that, "… it strongly ties to the effective multiplicity
1380 of parallel flows by which medium passes between any two arbitrary system components".
1381 Redundancy is linked by Christensen and Pauly (2008) with system stability and proposed by
1382 Heymans et al. (2007) as an index of food-web resilience. According to Bondavalli et al. (2000)
1383 high redundancy signifies that either the system is maintaining a higher number of parallel
1384 trophic channels in order to compensate for the effects of environmental stress, or that it is well
1385 along its way to maturity. With regard to overall performance and robustness, ecosystem level
1386 indicators based on ecological network analysis and food-web analysis are informative on
1387 intermediate and long time-scales (Cury et al., 2005; IEEP, 2005; Moloney et al., 2005). But they
1388 are difficult to use in annual updates and operational approach, and may be more difficult for
1389 stakeholders to understand (IEEP, 2005). In addition, use of food-web models and the ecological
1390 network analysis approach to explore different management scenarios, through simulation of
1391 fishery and nutrient management, could deliver integrated views at ecosystem level.

System omnivory index

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

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

Structural Indicators linked to diversity and 'canary' species

Guild Surplus Production models

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

Total biomass of small fish

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

1429Shephard et al. (2014) therefore suggest that both the biomass of individual stocks should be
1430above precautionary reference points on average and the total stock biomass of all pelagic fish
1431together should be above a joint community reference point. In practice, the community
1432reference point is always reached when all individual stocks are above precautionary reference
1433levels. However, in the case where one or more stocks are substantially below single stock
1434reference points, additional care should be taken in the exploitation of the remaining stocks in the
1435area.

1436   Proportion of Predatory Fish

1437Predatory fish species are defined as all surveyed fish species that are not largely planktivorous
1438(i.e. phytoplankton and zooplankton feeders should be excluded: Shin et al., 2010). A fish
1439species is classified as predatory if it is piscivorous, or if it feeds on invertebrates that are larger
1440than the macrozooplankton category (0.2 cm). Detritivores should not be classified as predatory
1441fish. This indicator captures changes in the trophic structure and changes in the functional
1442diversity of fish in the ecosystem. It is sensitive to fishing pressure, but since it is a ratio, it will
1443also be subject to changes in non-predatory fish, whose biomass may vary for other reasons (i.e.
1444environmental driver: Bundy et al., 2010).

1445This indicator is calculated as the biomass of predatory fish surveyed / biomass surveyed, and the
1446data required are trawl survey data and food habits data (if not available locally, from
1447information in the literature or from comparable systems).

1448   Pelagic to demersal ratio

1449The ratio of pelagic to demersal fish (P:D ratio) obtained from fishery-dependent or -independent
1450surveys is a commonly used metric that describes trophic energy flow and community structure
1451(de Leiva Moreno et al., 2000; Link, 2005; Rochet and Trenkel, 2003). Changes in P:D ratio
1452have been linked to anthropogenic pressures such as fishing and eutrophication. Targeted fishing
1453can result in notable shifts in this indicator, however, changes may be not be entirely clear, as an
1454increase in the P:D ratio could be caused by an increase in pelagic fish or a relative decrease in
1455demersal fish. As an indicator of food web properties, P:D ratio may overlap with other large
1456and/or forage fish indicators, but does capture important trophic relationships.

1457   Biomass of trophic guilds

1458Biomass of trophic guilds is a measure of ecosystem structure, estimated as the aggregate
1459biomass of each trophic guild. Individually they provide a measure of the change in biomass of
1460trophic guilds. Collectively, they provide a measure of change in overall structure. It can be
1461applied to all marine species if the information is available, based on survey data or model
1462results. Work to date has largely focused on fish trophic guilds (Rochet et al., 2013; Shackell et
1463al., 2012), but could be extended to invertebrates birds, and marine mammals. Measures of
1464functional diversity could also be developed using these data. Data sources can be from research
1465surveys or models.

Lifeform-based indicator for the pelagic habitat

1467 Ecosystem health theory (reviewed by Tett et al., 2013) suggests that ecosystem resilience, and
1468 the sustainability of services, depends *inter alia* on the abundance and relationships of non-
1469 substitutable `functional groups' or 'lifeforms'. The abundances and trophic structural
1470 relationships of phytoplankters, and their protozoan and mesozooplankton consumers, change
1471 seasonally. The Plankton Index (Pi) method takes account of such seasonality and requires the
1472 plotting of log-transformed lifeform abundances, based on at least monthly samples, in sets of 2-
1473 D state spaces (Gowen et al., 2011). These plots (Tett et al., 2008) often suggest a fuzzy
1474 doughnut. Using data for a reference period, an envelope can be drawn to include a fixed
1475 proportion (usually 90%) of points in this doughnut. Data from other years can be plotted against
1476 this envelope; the Pi[j,t] value (for lifeform pair j and year t) is defined as the proportion of new
1477 points that fall inside the envelope. For a given value of t, values of Pi for different lifeform pairs
1478 can be averaged. A UK project has identified sets of lifeform pairs that may serve for
1479 assessment of environmental status. The lifeform pairs relevant to Food Webs are: (i) chlorophyll
1480 concentration and mesozooplankton abundance; (ii) phytoplankton >= 20 µm abundance and
1481 phytoplankton < 20 µm abundance; (iii) [adult] copepods >= 2 mm abundance and [adult]
1482 copepods < 2 mm abundance. Reference conditions for any of the Pi are expected to be
1483 dependent on ecohydrodynamic (EHD) conditions (van Leeuwen et al., 2015). The UK is
1484 currently seeking EHD-specific references at sites in the Celtic or Greater North Sea MSFD
1485 ecoregions that are, according to expert judgement, in GES. Meanwhile, time-series of Pi will be
1486 generated from conditions observed during an agreed (but arbitrary) period of 3 years, and the
1487 time series will be assessed for (a) significant trends, and (b) significant correlation with relevant
1488 pressures.

Region-specific indicators of abundance & spatial distribution,

1490 Indicators can be selected to track the abundance and spatial distribution of major species which
1491 represent key community and or/ecosystem properties. Ideally, species representing different
1492 communities or habitats (benthos, plankton, fish, top predators) should be selected, in this way
1493 covering a large part of the ecosystem. As ecosystems are typically characterized by few strong
1494 links and many weak links among species or trophic levels, one (or few) indicator populations
1495 can describe broader ecosystem state and/or human perturbation. Criteria in the MSFD for
1496 selecting the groups/species that could be included in this category are those with fast turnover
1497 rates, groups/species that are targeted by fisheries, the habitat-defining groups/species, those at
1498 the top of the food web, and those tightly linked to other trophic levels (Rogers et al., 2010).

Fish biomass : benthos biomass from models

1500 Ratios are used to measure changes in community structure indicating the distribution of energy
1501 in the ecosystem. They are a supplement to biomass indicators and have the advantage that they
1502 do not reflect general increases or decreases in biomass in all components but only changes in
1503 the relative importance between the two groups. Hence, the ratio pelagic biomass : demersal

1504biomass represents the balance between pelagics and demersals whereas the fish/benthos ratio
1505reflects the proportion of the biomass which is diverted to benthos, including detritivores. The
1506indicator captures changes in the trophic structure and changes in the functional diversity of the
1507ecosystem. It is sensitive to fishing pressure, but since it is a ratio, it will also be subject to
1508changes in non-manageable benthos, whose biomass may vary for other reasons (e.g.
1509environmental driver, Bundy et al., 2010). Data sources can be research surveys (mainly nekton)
1510or models (often benthos, since this is often not surveyed on appropriate spatial and temporal
1511scales).

1512 Zooplankton spatial distribution and total biomass

1513This indicator, which describes the distribution of zooplankton, is still at the developmental stage
1514with methods, threshold and target values to be developed. The reasoning for this indicator is
1515that zooplankton constitutes an important link between primary producers and higher trophic
1516levels in the food web. Zooplankton plays an important role in the energy transfer and nutrient
1517cycling in the food web. Changes of the composition of the zooplankton community are coupled
1518to environmental changes and can respond quickly to ecosystem changes. Zooplankton biomass
1519and abundance can e.g. respond to invasive species and local oil spills.

1520 Scavenger biomass

1521Fishery discards provide food subsidies that help maintain fish and seabird populations and may
1522allow some of these populations to be more abundant than they would naturally be (e.g. Link and
1523Almeida, 2002; Polis and Strong, 1996). Surveys of non-targeted scavenger biomass or
1524abundance may provide an index of disturbance (Link and Almeida, 2002; Methratta and Link,
15252006b). Additionally, some scavenger species might be viewed as "canary" or "iconic" species
1526that can be used as an early warning of disturbance or excessive fishing pressure.

1527 Geometric mean abundance of seabirds

1528The Geometric Mean Abundance of Seabirds is computed in regular intervals (e.g. yearly) as the
1529geometric mean of the population sizes (e.g. numbers of individuals or breeding pairs) of those
1530seabirds in the assessment region for which population time series are available, normalised such
1531that the indicator value at the beginning of the indicator time series is one. The indicator is
1532designed after the Living Planet Index (LPI: Loh et al., 2005), which now underlies Aichi Target
15335 of the Convention for Biological Diversity. Modern indicator protocols take into account that
1534species may enter or leave the set of species for which time series are available, and that
1535population sizes at low abundances become uncertain. Methods to compute indicator confidence
1536intervals have been developed (Buckland et al., 2011; Loh et al., 2005). By its definition, the
1537proportional rate of change of the indicator equals the average population growth rate of all
1538populations contained in the indicator (here seabirds). Under conditions where populations
1539fluctuate and turn over but overall biodiversity does not change, the indicator is expected not to
1540deviate significantly from one. A steady decline of geometric mean abundance signals

1541 biodiversity loss. Seabird populations are known to be highly sensitive to food availability (Cury
1542 et al., 2011), and their differentiation of foraging niches (Fasola et al., 1989) is evidence of
1543 competition for food among them. Competitive exclusion resulting from loss of biodiversity
1544 among their marine resources (e.g. forage fish), or even at lower trophic levels (Rossberg, 2013),
1545 can be expected to induce the slow decline of seabird diversity to which this indicator is designed
1546 to be sensitive. Geometric mean abundance of seabirds is therefore sensitive to a collapse of the
1547 pyramidal distribution of species over trophic levels in food webs (de Ruiter et al., 2005).

1548 **Gini-Simpson diversity index (species dominance) of large and small fish by**
1549 **biomass**

1550 It is incompatible with GES to bring the foodweb into a state where only a few (large) predator
1551 or prey species dominate the system when the biomass of predators and prey was distributed
1552 more evenly in the system during the reference period. Species richness may be inadequate as an
1553 indicator as it often takes a long time to completely lose a species, while management should be
1554 informed and act earlier. The Gini-Simpson index (1-D) applied to the predator and/or prey
1555 community provides the possibility to detect unwanted changes in diversity. Simpson's Diversity
1556 Index is a measure of diversity which takes into account the number of species present, as well as
1557 the relative abundance of each species. As species richness and evenness increase, so does
1558 diversity (ICES, 2013b).

1559 **Species Richness Index**

1560 Species richness measures the number of species within a community. A well-structured and
1561 functioning ecosystem will generally have many species (Fung et al., 2015); as a side effect of
1562 fishing, species richness may decrease (Rice, 2003, 2000). However, as a food web indicator
1563 species richness may provide ambiguous information, since multiple community configurations
1564 may produce similar values, as shown by Gislason and Rice (1998) and Rice and Gislason
1565 (1996). In these studies the index was calculated as the number of species in any year whose
1566 numerical abundance or biomass was larger than some percentage of their value in a reference
1567 year. The IUCN Red List criterion of 20% was used as the reference value. Required data:
1568 species or functional group P/B, and species or functional group biomass/abundance to compare
1569 to reference points.

1570 **The Large Fish Indicator (LFI)**

1571 The Large Fish Indicator (LFI) is defined as the proportion by weight of large fish in the sample
1572 of a specified survey (Greenstreet et al., 2011), where large fish are defined as those longer than
1573 a threshold length L_{th}, a region-specific value. The threshold value is chosen such as to optimize
1574 the responsiveness of the indicator to fishing pressure, as determined from historic data
1575 (Shephard et al., 2011). The LFI takes no account of species identity, only of individual sizes.
1576 However, it was shown to reflect mostly the proportion (by weight) of large-bodied species in
1577 communities (Shephard et al., 2012). Large-bodied species tend to be more vulnerable to fishing,

1578which is why the LFI is sensitive (Engelhard et al., 2015; Greenstreet et al., 2011; ICES, 2011;
1579Shephard et al., 2013) and specific (Houle et al., 2012) to fishing pressure. Furthermore, by
1580expressing the indicator in terms of proportions by weight, and not by numbers, and through
1581judicious choice of the appropriate length threshold to define large fish, the indicator can be
1582desensitized to variation in the abundance of small fish. The influence of environmentally driven
1583recruitment events on indicator values can therefore be minimized (Greenstreet et al., 2011).
1584Food-web models (Fung et al., 2013; Shephard et al., 2013) and data (Fung et al., 2012) suggest
1585that recovery of the indicator from pressures can be slow (decadal scale). The LFI, as an OSPAR
1586EcoQO for the North Sea, is fully operational. It was named as an indicator for food-web GES
1587(EU, 2010), and has been chosen as a common food-web indicator by HELCOM and OSPAR (in
1588some OSPAR Subregions as a priority candidate indicator).

1589   **Mean length of surveyed community**

1590Mean length (ML) of all species caught in a survey, whether fishery-independent, fishery-
1591dependent, or based on landings, can be a useful and simple indicator to evaluate the overall
1592effects of fishing on an ecosystem (Dulvy et al., 2004; Nicholson and Jennings, 2004; Rochet
1593and Trenkel, 2003; Shin et al., 2005). ML quantifies relative abundances of large and small
1594individuals and describes the size distribution of a community (Shin et al., 2005). It is relatively
1595responsive to key pressures (Link, 2010; Pauly et al., 1998). ML is considered measurable and
1596generally robust, however, the direction of response may be caused by increasing stocks of large
1597fish or decreasing in stocks of small fish, leading to potential ambiguity. Whilst the metric is
1598sensitive to fishing pressure, it can also be strongly influenced by environmentally driven
1599recruitment events that introduce large numbers of small fish into the community (Badalmenti et
1600al., 2002; Lekve et al., 2002; Wilderbuer et al., 2002).

1601   **Size spectrum slope**

1602Various measures of the change in size can be a useful indicator to describe composition of
1603communities (Nicholson and Jennings, 2004). Size spectrum slope measures the relationships
1604between the biomass (y) of individuals within a body size class and body size (x), both normally
1605plotted on logarithmic scales. Frequently a logarithmic transformation is applied to body size,
1606particularly when weight classes are used. When applied to fish communities, the slope of the
1607relationship becomes increasingly negative in response to fishing pressure; fisheries reduce the
1608abundance of large fish, through the direct effect of fishing and, as a consequence of reduced
1609predation pressure from large fish, the abundance of small fish increases (Daan et al., 2005;
1610Gislason and Rice, 1998; Nicholson and Jennings, 2004; Rice and Gislason, 1996). The size
1611spectrum slope is considered measurable and robust. However, the direction of the response may
1612not be entirely clear (Trenkel and Rochet, 2003), as the steepening of the slope could indicate a
1613decrease of large fish or an increase of small fish. The slope is particularly sensitive to changes
1614in the abundance of small fish, which markedly affect the intercept of the regression line, as such

1615the size spectrum slope can be influenced by environmentally driven recruitment events
1616(Badalmenti et al., 2002; Lekve et al., 2002; Wilderbuer et al., 2002).

1617   **Zooplankton Size-Biomass index**

1618This is a zooplankton indicator reflecting both mean individual size and total biomass of the
1619zooplankton community. The indicator represents food web capacity to sustain fish feeding
1620conditions and grazing on primary producers. The rationale is that both mean body size in the
1621community and total community biomass are positively related to fish feeding conditions,
1622whereas total biomass alone is just representative of grazing pressure and trophic transfer
1623efficiency (Fuchs and Franks, 2010). The effects of zooplankton community structure on energy
1624transfer and food web resilience have been demonstrated in both freshwater and marine systems
1625(Jeppesen et al., 2011; Kane et al., 2009; Lougheed and Chow-Fraser, 2002). The index is
1626currently considered as a core indicator for the Baltic Sea (HELCOM, 2013, 2012). In semi-
1627enclosed seas, such as the Baltic Sea, with strong salinity and temperature gradients, no single
1628zooplankton group can adequately reflect community properties (Remm 1984), hence the need
1629for this two-dimensional index. The index value decreases with increasing fishing pressure.
1630Protocols for indicator assessment have been developed by HELCOM Zooplankton Expert
1631Network (ZEN) using nine long-term monitoring datasets in the Baltic Sea (HELCOM, 2013,
16322012). In all datasets, the indicator was found to predict deviations from GES conditions.
1633Determination of GES boundaries for the indicator is straightforward and based on the regional
1634basin-specific Environmental Quality Ratios for chlorophyll accepted within Water Framework
1635Directive and weight-at-age for zoo-planktivorous fish (HELCOM, 2013, 2012).