ICES Journal of
Marine Science

ICES CIEM International Council for the Exploration of the Sea Conseil International pour l'Exploration de la Mer

ICES Journal of Marine Science (2015), 72(3), 1105–1115. doi:10.1093/icesjms/fsu245

Contribution to the Themed Section: 'Risk Assessment'

Original Article

An exposure-effect approach for evaluating ecosystem-wide risks from human activities

Antony M. Knights[1†*], Gerjan J. Piet[2†], Ruud H. Jongbloed[2†], Jacqueline E. Tamis[2†], Lydia White[3], Ekin Akoglu[4], Laura Boicenco[5], Tanya Churilova[6], Olga Kryvenko[6], Vivi Fleming-Lehtinen[7], Juha-Markku Leppanen[7], Bella S. Galil[8], Freya Goodsir[9], Menachem Goren[10], Piotr Margonski[11], Snejana Moncheva[12], Temel Oguz[13], K. Nadia Papadopoulou[14], Outi Setälä[7], Chris J. Smith[14], Kremena Stefanova[6], Florin Timofte[5], and Leonie A. Robinson[3†]

[1]*Marine Biology and Ecology Research Centre, School of Marine Science and Engineering, Plymouth University, Drake Circus, Plymouth PL4 8AA, UK*
[2]*Institute for Marine Resources and Ecosystem Studies (IMARES), Haringkade 1, IJmuiden 1976 CP, The Netherlands*
[3]*School of Environmental Sciences, University of Liverpool, Nicholson Building, Liverpool L69 3GP, UK*
[4]*Instituto Nazionale di Ocenaographie e di Geofisica Sperimentale (OGS), Borgo Grotta Gigante 42/C, Sgonico, Italy*
[5]*National Institute for Marine Research and Development "Grigore Antipa", Constanta 900581, Romania*
[6]*A.O. Kovalevskiy Institute of Biology and Southern Seas, National Academy of Sciences of Ukraine 2, Nakhimov Av., Sevastopol, Crimea 99011, Ukraine*
[7]*Marine Research Centre, Finnish Environment Institute (SYKE), PO Box 140, Helsinki FI-00251, Finland*
[8]*National Institute of Oceanography, Israel Oceanographic and Limnological Research (NIO-IOLR), Tel Shikmona, Haifa 21080, Israel*
[9]*Cefas, Pakefield Road, Lowestoft, Suffolk NR33 0HT, UK*
[10]*Department of Zoology, Tel Aviv University, Tel Aviv 69778, Israel*
[11]*Department of Fisheries Oceanography and Marine Ecology, National Marine Fisheries Research Institute, ul. Kollataja 1, Gdynia 81-332, Poland*
[12]*Institute of Oceanology, BAS 9000 Varna, PO Box 152, Bulgaria*
[13]*Institute of Marine Sciences, Middle East Technical University, PO Box 28, Erdemli 33731, Turkey*
[14]*Hellenic Centre for Marine Research, Institute of Marine Biological Resources and Inland Waters, PO Box 2214, Heraklion 71003, Crete*

**Corresponding author: tel: +44 1752 587889; fax: +441752 586101; e-mail: antony.knights@plymouth.ac.uk*

Knights, A. M., Piet, G. J., Jongbloed, R. H., Tamis, J. E., White, L., Akoglu, E., Boicenco, L., Churilova, T., Kryvenko, O., Fleming-Lehtinen, V., LeppanenJuha-Markku, Galil, B. S., Goodsir, F., Goren, M., Margonski, P., Moncheva, S., Oguz, T., Papadopoulou, K. N., Setälä, O., Smith, C. J., Stefanova, K., Timofte, F., and Robinson, L. A. An exposure-effect approach for evaluating ecosystem-wide risks from human activities. – ICES Journal of Marine Science, 72: 1105–1115.

Received 19 August 2014; revised 8 December 2014; accepted 9 December 2014.

Ecosystem-based management (EBM) is promoted as the solution for sustainable use. An ecosystem-wide assessment methodology is therefore required. In this paper, we present an approach to assess the risk to ecosystem components from human activities common to marine and coastal ecosystems. We build on: (i) a linkage framework that describes how human activities can impact the ecosystem through pressures, and (ii) a qualitative expert judgement assessment of impact chains describing the exposure and sensitivity of ecological components to those activities. Using case study examples applied at European regional sea scale, we evaluate the risk of an adverse ecological impact from current human activities to a suite of ecological components and, once impacted, the time required for recovery to pre-impact conditions should those activities subside. Grouping impact chains by sectors, pressure type, or ecological components enabled impact risks and recovery times to be identified, supporting resource managers in their efforts to prioritize threats for management, identify most at-risk components, and generate time frames for ecosystem recovery.

Keywords: ecosystem-based management, exposure-effect, human activities, impact, marine, risk framework.

[†] These authors wish to be considered as joint first authors.

Introduction

Current rates of resource exploitation are unsustainable and the ecosystem approach has been widely promoted as the framework to achieve sustainable use (Airoldi and Beck, 2007; EC, 2008; Halpern *et al.*, 2008). By definition, an ecosystem is a diverse range of physical and biological components which function as a unit (*sensu* Tansley, 1935), and therefore, an ecosystem approach should ideally consider the complete range of interactions that human activities have with the ecosystem and its components. However, the number of sectors that exploit the ecosystem and its components is often great, resulting in many different pressures and a complex network of interactions (Knights *et al.*, 2013). Identification and prioritization of interactions for management can therefore be difficult (Bottrill *et al.*, 2008), presenting a major challenge to transforming the ecosystem approach from a concept into an operational framework (Leslie and McLeod, 2007).

The onus has been placed on the scientific community to identify the pathways through which activities cause harm (Leslie and McLeod, 2007; Fletcher *et al.*, 2010). The relationships between human activities and ecological components have commonly been described using linkage-based frameworks. These adopt the causal-chain concept to infer pressure–state relationships (Rounsevell *et al.*, 2010) and have been applied widely in both marine and terrestrial environments (e.g. Elliott, 2002; La Jeunesse *et al.*, 2003; Odermatt, 2004; Scheren *et al.*, 2004; Holman *et al.*, 2005). The simplicity of these frameworks is advantageous as key relationships can be captured and displayed in a relatively simple way (Rounsevell *et al.*, 2010). However, viewing linkages in isolation rather than accounting for the interplay across sectors, activities, pressures, or components may be overly simplistic (Tallis *et al.*, 2010) and can lead to ineffective management (Khalilian *et al.*, 2010). A flexible, problem-solving approach is therefore required that can link the relationship between the human activities and the environment while supporting the decision-making needs of environmental managers.

Risk assessment can provide a solution (Hope, 2006). Risk assessment in general describes the likelihood and consequences of an event. In an ecosystem-based management (EBM) context, risk can be defined as the degree to which human activities interfere with the achievement of management objectives related to particular ecological components (Samhouri and Levin, 2012). It is increasingly seen as a way to integrate science, policy, and management and has been widely used to address a range of environmental issues (e.g. Francis, 1992; Fletcher, 2005; Smith *et al.*, 2007; Hobday *et al.*, 2011; Samhouri and Levin, 2012). There are several risk assessment approaches available using quantitative data (e.g. Francis, 1992; Samhouri and Levin, 2012), which is best suited for strategic of tactical decision-making, or qualitative data (e.g. Fletcher, 2005; Fletcher *et al.*, 2010; Breen *et al.*, 2012), which instead support broad assessments best interpreted and applied as a screening tool. Many ecological risk assessments (Fletcher, 2005; Campbell and Gallagher, 2007; e.g. Astles *et al.*, 2006) are based on a likelihood-consequence approach for estimating the risk of a rare or unpredictable event (Williams *et al.*, 2011). But when an assessment to screen for ongoing, current pressure is needed, then an exposure-effect analysis is more suitable (Smith *et al.*, 2007). Several studies have used the exposure-effect concept to assess risk to habitats and species from ongoing human activities (e.g. Bax and Williams, 2001; Stobutzki *et al.*, 2001) using qualitative descriptors such as habitat resistance (to physical modification) and resilience (the time taken for the habitat to recover to pre-impact condition) to assess habitat vulnerability (Bax and Williams, 2001). Assessments have tended to focus on a single activity or target species (e.g. fishing, Bax and Williams, 2001; Fletcher, 2005; Hobday *et al.*, 2011; Zhang *et al.*, 2011) but have recently been broadened to include a greater number of activities and non-target species and applied at larger management scales (Samhouri and Levin, 2012).

Here, we illustrate how the exposure-effect approach can be used to assess the risk to ecosystems from human activities at considerably larger spatial scales than those previously described. Although the definition of "regional" can be broadly interpreted (e.g. Samhouri and Levin, 2012, used regional to describe the Puget Sound, USA); here, we apply the regional definition given in the Marine Strategy Framework Directive (MSFD) (EC, 2008); a recent Europe-wide environmental policy mechanism. Therein, regional seas are defined as the northeast Atlantic, the Baltic Sea, the Black Sea, and the Mediterranean Sea (Figure 1). We build on (i) a linkage framework made up of potential pressure mechanisms describing how different sectors can impact ecological components of the ecosystem (Knights *et al.*, 2013), and (ii) a pressure-based expert judgement assessment of the exposure and sensitivity of ecosystems to sector activities and their pressures (Robinson *et al.*, 2013) to show the potential risks to ecological components from a holistic range of sectors in each region and which are integral features of marine ecosystems worldwide. This is the first of a series of steps required when implementing EBM (Knights *et al.*, 2014a).

Methods

An assessment of the risk to Europe's regional sea ecosystems from human activities must consider a range of sectors, pressures, and ecological components beyond those included in previous studies (e.g. Bax and Williams, 2001; Samhouri and Levin, 2012). We included (i) up to 17 sectors (the number of sectors included in a regional assessment was dependent on whether it is currently operational in the region), (ii) 23 pressure types, and (iii) 5 broad ecological components (Supplementary Table A1). Two of the ecological components (fish and predominant habitats) were further disaggregated into "sub-components" to give greater resolution and differentiation of the impact of sectors on those components (these sectors were identified as primary drivers of impact in each regional sea; Knights *et al.*, 2013), resulting in a total of 11 ecological components (Supplementary Table A1). Here, we provide an illustration of the approach rather than undertaking an exhaustive assessment and the list of components could be expanded to the end-user's needs, although the components we have included are the main representatives outlined in the EU MSFD (EC, 2008). Furthermore, we only consider direct effects of sector–pressures on ecological components, but we recognize that indirect effects can play an important role in the functioning of an ecosystem (Dunne *et al.*, 2002).

Linkage mapping and pressure (threat) assessment

A first step in developing the assessment framework was the creation of a sector–pressure–ecological component linkage matrix. Each cell in the matrix describes the potential for impact on an ecological component from a sector, wherein a pressure is the mechanism through which an impact occurs. We refer to this linear interaction between a sector, pressure, and ecological component as an "impact chain" herein. Impact chains were defined following an extensive

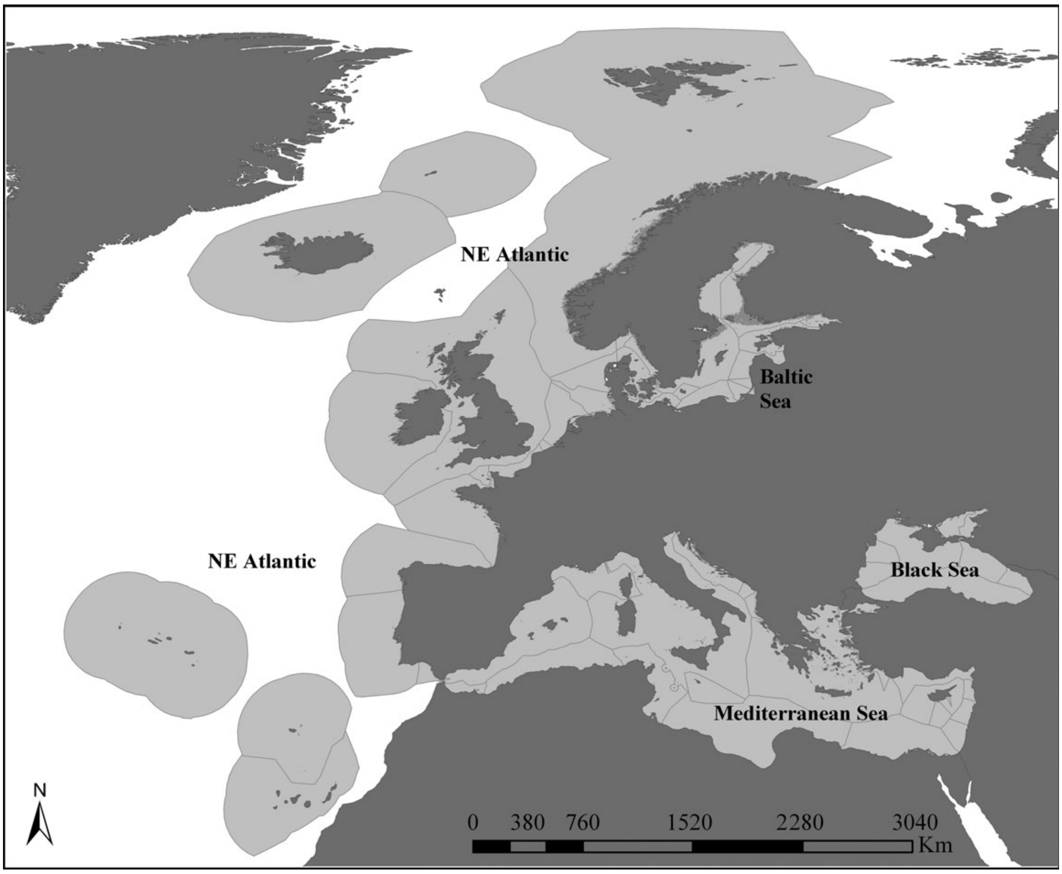

Figure 1. Regional Sea areas of Europe as defined by the MSFD (light grey areas indicate the spatial coverage of the directive). Impact chains were assessed at the scale of the region for the NE Atlantic, Baltic Sea, Black Sea, and Mediterranean Sea. Exclusive economic zone (EEZ) borders are shown.

review of the peer-reviewed scientific literature and published reports (see Knights *et al.*, 2013, for full details of the linkage matrix) resulting in a pre-pressure assessment matrix of 4320 potential impact chains. Accurate calculation of threat and risk is reliant upon the inclusion of all possible impact chains and every effort was made to include all relevant chains (see Knights *et al.*, 2013, for full details), although some more minor linkages may be missing as a result of uncertainty (Walker *et al.*, 2003).

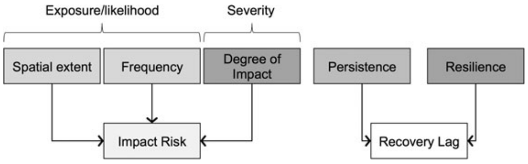

Figure 2. Exposure-effect assessment criteria used in the calculation of risk and RL. Criteria definitions are given in Robinson *et al.* (2013). Definitions: IR is a measure of the likelihood of an adverse ecological impact occurring following a sector – pressure introduction. The greater the IR, the greater the likelihood and severity of an impact. An adverse impact is defined as a negative effect on the state of the ecosystem component, but the state or reduction in state as a result of the impact are not defined. RL is a measure of management potential given the persistence of a pressure and resilience of the impacted ecological component. RL is defined as the time (years) it takes for an ecological component to return to pre-impacted condition (Table 2).

Threat from each chain was assessed by way of a pressure assessment (*sensu* exposure-effect) approach (see Robinson *et al.*, 2013, for full details of the methodology). The pressure assessment methodology was designed with the concept of risk assessment in mind, such that the assessment criteria we developed could be used to evaluate the likelihood and consequences of a specific or combination of impact chains. The assessment was based on expert judgement (Cooke and Goossens, 2004) given by 40 participants from 17 institutions and 13 countries from around the EU and more broadly. Data were collected using the World Café methodology (Brown, 2002; Elliot *et al.*, 2005), and participants qualitatively assessed each impact chain using a categorical assessment of five criteria: (1–2) two describing the exposure of the ecological component to a sector–pressure combination; (3) one describing the severity of the interaction; and (4–5) two describing recovery (Figure 2; Table 1). Participants were supported by a comprehensive literature review

Table 1. The pressure assessment criteria and categories used to evaluate each impact chain (after Robinson *et al.*, 2013) and the numerical risk scores assigned to each category.

	Description	**Percent overlap (%)**	**Standardized value (proportion of max)**
Spatial extent	The spatial extent of overlap between a pressure type and ecological characteristic		
Widespread	Where a sector overlaps with an ecological component by 50% or more (max is 100%).	75	1.00
Local	Where a sector overlaps with an ecological component by >5% but <50%. A raw value taken as the midpoint between the range boundaries	27.5	0.37
Site	Where a sector overlaps with an ecological component by >0% but <5%. A raw value taken as the midpoint between the range boundaries	2.5	0.03
		Months per year	
Frequency	How often a pressure type and ecological characteristic interaction occurs measured in months per year		
Persistent	Where a pressure is introduced throughout the year	12	1.00
Common	Where a pressure is introduced up to 8 months of the year	8	0.67
Occasional	Where a pressure is introduced up to 4 months of the year	4	0.33
Rare	Where a pressure is introduced up to 1 month of the year	1	0.08
		Severity per interaction	
Degree of Impact	An acute (A) interaction is an impact that kills a large proportion of individuals and causes an immediate change in the characteristic feature. A chronic (C) interaction is an impact that could have detrimental consequences if it occurs often enough and/or at high enough levels. A low severity (L) interaction never causes high levels of mortality, loss of habitat, or change in the typical species or functioning irrespective of the frequency and extent of the event(s)		
Acute	Severe effects after a single interaction	1	1.00
Chronic	Severe effects occur when the frequency of introductions exceed a specified number of interactions. Here, that critical value was specified as 8 occurrences (or $1/8 = 0.125$)	0.125	0.13
Low	Severe effect not expected. For precautionary reasons, we assume a potential effect after 100 introductions	0.01	0.01
		Persistence (years)	
Persistence	The period over which the pressure continues to cause impact following cessation of the activity introducing that pressure		
Continuous	The pressure continues to impact the ecosystem for at least 100 years	100	1.00
High	The pressure continues to impact the ecosystem for between 10 and 100 years. A raw value taken as the midpoint between the range boundaries	55	0.55
Moderate	The pressure continues to impact the ecosystem for between 2 and 10 years. A raw value taken as the midpoint between the range boundaries	6	0.06
Low	The pressure continues to impact the ecosystem for between 0 and 2 years. A raw value taken as the midpoint between the range boundaries	1	0.01
		Recovery (years)	
Resilience	The resilience (recovery time) of the ecological characteristic to return to pre-impact conditions. Recovery times for species assessments were based on turnover times (e.g. generation times). For predominant habitat assessments, recovery time was the time taken for a habitat to recover its characteristic species of features given prevailing conditions		
None	The population/stock has no ability to recover and is expected to go "locally" extinct. The recovery in years is predicted to take 100+ years	100	1.00
Low	The population will take between 10 and 100 years to recover. A raw value taken as the midpoint between the range boundaries	55	0.55
Moderate	The population will take between 2 and 10 years to recover. A raw value taken as the midpoint between the range boundaries	6	0.06
High	The population will take between 0 and 2 years to recover. A raw value taken as the midpoint between the range boundaries	1	0.01

of primary, secondary, and tertiary information sources and had access to online resources throughout the proceedings. Participants evaluated each impact chain considering prevailing conditions, applied here at a European regional sea scale, not least so that the outcomes of the assessment could support the objectives of the MSFD (EC, 2008). Each regional sea group reached agreement in the assessment of each impact chain. Some impact chains were excluded from the final assessment based on the absence of a sector (and thus its pressures) in the regional sea. As such, a separate network of impact chains was developed for each regional sea (see Knights *et al.*, 2013, for full details of the network model).

Assessing risk and recovery in large ecosystems

Our approach builds on a long series of antecedents of productivity susceptibility analysis (e.g. Stobutzki *et al.*, 2001; Hobday *et al.*, 2011; Samhouri and Levin, 2012). We applied numerical scores to each qualitative assessment category (Table 1) and used combinations of the assessment criteria to describe two axes of information: "impact risk (IR)" and "recovery lag (RL)" (Figure 2). IR was constructed using a combination of exposure (2) and sensitivity (1) criteria, which describe the spatial extent and temporal (frequency) overlap of a sector–pressure within an ecological component, and the severity of the interaction where overlap occurs (degree of impact). These criteria were combined into the aggregate criterion, we refer to as IR, where the greater the IR score, the greater the threat to a component (Figure 2). It is important to note that each assessment criterion was evaluated independently before being combined into an aggregate score. This was intentional such that the effect of each criterion on the combined risk score could be evaluated separately, but which can lead to equivalent scores from different combinations, e.g. "Acute-Occasional-Widespread" and "Acute-Persistent-Low" (Table 2).

RL was described using the combination of pressure persistence (the number of years before the pressure impact ceases following cessation of the sector introducing it) and ecological component resilience (recovery time) following the cessation of the pressure impact. This aggregate criterion gives an indication of the time required for potential improvement in ecosystem state to be seen following the management of a specific impact chain, where the greater the RL value, the longer period required for an ecological component to recovery back to its pre-impacted state.

As assessment criteria had a varying number of assessment categories (as many as 5 and as few as 3), scores for each category were standardized using percentage scores, where the worst case equates to a score of 1 (Table 1) and other categories calculated as fractions of that total. Each axis receives equivalent weight in estimating threat and under this framework, the IR and/or RL for an ecological component increases with distance from the origin. The assessment allows the "worst" impact chain or chains to be identified (either in terms of IR and/or RL) in isolation or grouped in combinations, e.g. by sector or pressure.

IR and RL scores were calculated for each impact chain as the product (multiplication) of the assigned categorical scores (Table 2) to enable direct comparison and for the purposes of calculating the contribution of IR and RL to "total risk" (see Piet *et al.*, in press). However, to indicate recovery time in years following an impact, RL standardized values were converted into minimum time to recovery in years based on the ranges given in Table 2. Recovery time (years) was calculated as the sum of the pressure persistence (years) and recovery time (years) ($P + R$) values for a given combination.

IR and RL (years) were then grouped, either by sector, pressure type, or ecological component and the distribution of values presented using boxplots. IR scores can range between 0.002 and 1, where 1 is the worst case, and RL time frames range between 1 and 200 years (Tables 1 and 2).

Table 2. IR, RL standardized scores ($P \times R$), and minimum time (years) for recovery ($P + R$) of ecological components (ECs) for all possible category combinations (category definitions are shown in Table 1).

IR products		**Frequency**			
Extent	**Degree of impact**	*Persistent*	*Common*	*Occasional*	*Rare*
Widespread	*Acute*	1.00000	0.67000	0.33000	0.08000
Local	*Acute*	0.33000	0.22110	0.10890	0.02640
Site	*Acute*	0.03000	0.02010	0.00990	0.00240
Widespread	*Chronic*	0.12500	0.08375	0.04125	0.01000
Local	*Chronic*	0.04125	0.02764	0.01361	0.00330
Site	*Chronic*	0.00375	0.00251	0.00124	0.00030
Widespread	*Low*	0.01000	0.00670	0.00330	0.00080
Local	*Low*	0.00330	0.00221	0.00109	0.00026
Site	*Low*	0.00030	0.00020	0.00010	0.00002

RL products		**Resilience**		
Persistence	*None*	*Low*	*Moderate*	*High*
Continuous	1.0000	0.5500	0.0600	0.0100
High	0.5500	0.3025	0.0330	0.0055
Moderate	0.0600	0.0330	0.0036	0.0006
Low	0.0100	0.0055	0.0006	0.0001

Minimum recovery time (years)		**Resilience**		
Persistence	*None*	*Low*	*Moderate*	*High*
Continuous	200	110	102	101
High	110	20	12	11
Moderate	102	12	4	3
Low	101	11	3	1

Results

Using expert judgement, we identified and evaluated 3347 sector–pressures that can affect the ecological components of Europe's regional seas. The distribution of sector–pressures was split between predominant habitat types (1817) and mobile species, such as fish, seabirds, and marine mammals (1530) with the number of impact chains affecting each component varying between regional seas as a result of differences in the types of sectors operating in each sea, and thus the type and number of pressures introduced.

IR scores were generally low, with little variation between regions irrespective of the sector or pressure considered (Figure 3). The median IR score per chain per region ranged from 0.003 in the Baltic and Black Seas and NE Atlantic and 0.013 in the Mediterranean Sea (see Table 2 for possible combinations). Outliers were, however, many and in some cases the IR values exceed 0.69, indicating that the presence of acute severity, spatially widespread and persistent introductions of some pressures (Figure 3, Table 2). Grouping impact chains by sector indicated that the IR for the majority of pressures they introduce is relatively low (<0.01; Figure 3), indicating relatively low severity impacts and/or spatially or temporally restricted impacts. Fishing was the sector posing the greatest risk, exhibiting multiple outliers with IR values >0.4, indicating many widespread and frequent impact chains with severe consequences. Similar outliers were common to fishing in all regional seas, suggesting that the impact mechanisms are the same irrespective of regional differences in the sector activities (Figure 3).

RL was more varied than the IR scores for the same sector-grouped chains. Median values were relatively low and consistent across all regions, indicating that recovery to pre-impacted

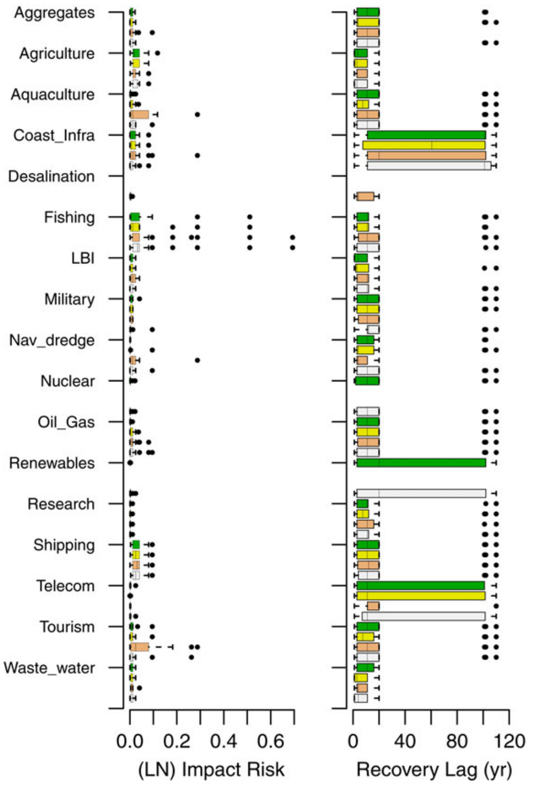

Figure 3. Distribution of IR and RL scores grouped by sector in each of four European regional seas (ordered as Baltic Sea, Black Sea, Mediterranean Sea and NE Atlantic). The maximum IR and RL score for any chain is 0.7 and 1.0, respectively. No bar indicates the absence of the sector in this region. Middle lines of boxplots represent the median values; hinge lengths (end of box) represent the 25% quartiles from the median; whiskers represent the 1.5 times the interquartile range (IQR) beyond the hinge. Outliers are shown as black dots. The same format applies to subsequent boxplots.

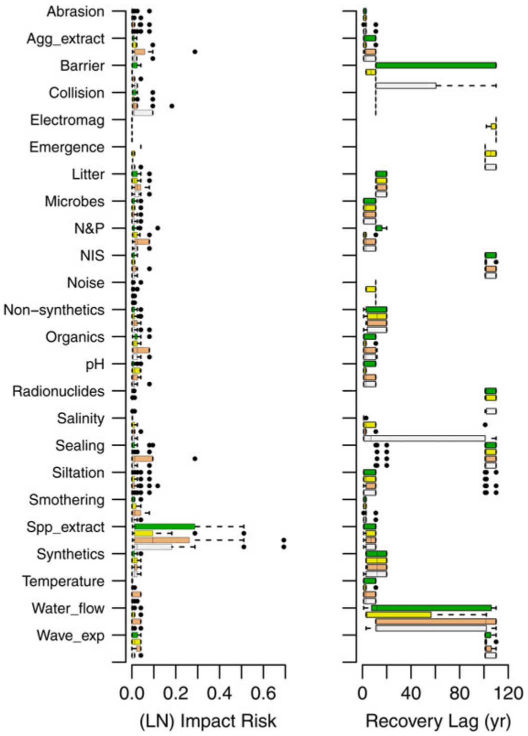

Figure 4. Distribution of IR and RL scores grouped by pressure type in each of four European regional seas (ordered as Baltic Sea, Black Sea, Mediterranean Sea and NE Atlantic). The maximum IR and RL score for any chain is 0.7 and 1.0, respectively. No bar indicates the absence of the pressure in the region. Boxplot information is given in the legend of Figure 3.

condition would occur in 11 years (Figure 3, Table 2), although nearly every sector introduces at least one pressure that takes ecological component(s) >100 years to recover from. In contrast to the IR scores (which were predominantly low; 99% had values <0.05), there was a greater proportion of impact chains with intermediate or high RL time frames of >100 years. In fact, of the 3347 impact chains considered, 14% had an RL of >100 years (458 chains).

Grouping impact chains by the pressure type identified which pressures pose the greatest IR to the ecosystem. Median IR scores were low always; 0.003 in the Baltic Sea and NE Atlantic, 0.011 in the Mediterranean Sea and 0.005 in the Black Sea (Figure 4). Greatest impact scores were associated with the pressure type "species extraction" (0.51–0.69), indicating widespread, common/persistent, and acute impacts throughout all regions (Table 2).

RL was highly dependent on the pressure type. Relatively short minimum recovery times (between 1 and 11 years) were associated with physical pressures [i.e. abrasion, aggregate extraction (agg_extract), collision, noise, smothering, and species extraction (spp_extract)] in all regions (Figure 4). In contrast, biotic pressures [e.g. non-indigenous species (NIS)], contaminant pressures (e.g. radionuclides, marine litter), and hydrological pressures (e.g. water flow regimes, wave exposure) were characterized by long RL times of >100 years before a return to pre-impacted conditions (Figure 4). In some cases, there was little difference in recovery time associated with a particular pressure type between regional seas (e.g. non-synthetic or synthetic contaminants). For other pressure types, such as nitrogen and phosphorus enrichment (N&P) and barriers to species movement (Barriers), there were marked differences between regions, where recovery times were relatively long in one region but short in all other regions. For example, recovery following N&P was estimated to take a minimum of 11 years in the Baltic Sea, but only 2–3 years in all other regions (Figure 4), with differences due to the susceptibility and recovery potential of different ecological components as well as changes in the persistence of the pressure type in that region.

Grouping impact chains by ecological components indicated that many sector–pressure combinations are low IRs (Figure 5). There were, however, a greater number of outliers compared with groupings by sector or pressure, indicating variability in the impact of specific sector–pressure combinations on an ecological component. In

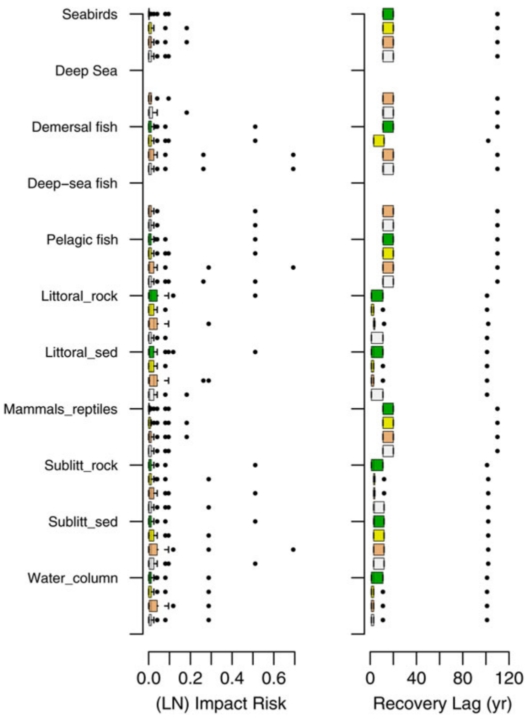

Figure 5. Distribution of IR and RL scores grouped by ecological component in each of four European regional seas (ordered as Baltic Sea, Black Sea, Mediterranean Sea and NE Atlantic). The maximum IR and RL score for any chain is 0.7 and 1.0, respectively. No bar indicates that the ecological component is not present in this region. Boxplot information is given in the legend of Figure 3.

many of these cases, IR scores exceeded 0.5 (acute, widespread, and common or persistent) and the majority of ecological components impacted by an acute severity impact chain that is either locally persistent or occasionally widespread (0.28; Table 2).

Recovery times of the ecological components of different regional seas were largely comparable (Figure 5). For most sector–pressure combinations, recovery times of ecological components were in the region of 1 and 20 years depending on the ecological component in question. Median minimum recovery times were generally longer (11–20 years) for mobile species (i.e. seabirds, deep sea habitats and fish, demersal and pelagic fish, and marine mammals and reptiles) than predominant habitat types (1–4 years for all habitats except the deep sea which requires a minimum of 11–12 years; Figure 5).

In addition or instead of considering all impact chains in a holistic assessment, the impact of a single sector (grouped by pressure type) on the ecosystem can be singled-out for assessment. We illustrate this using the sector "fishing" and the ecological component, "sublittoral sediment", although data can be grouped by any sector, pressure type, or ecological component. Fishing introduced a suite of 13 different pressure types, many of which were relatively low in impact, and from which, the ecosystem is able to recover quickly (Figure 6). Unsurprisingly, species extraction (spp_extract) is the pressure type with the greatest IR, but noting that the recovery time following this pressure type is estimated to be relatively fast ($\sim$11 years for recovery), driven by the low persistence of this pressure despite relatively low resilience scores for some ecological components. Conversely, pressures such NIS were characterized as relatively low in terms of IR (median = 0.003), and extremely slow recovery times (minimum time = 102 years), driven by the difficulties of eradicating invasive species (Galil, 2003).

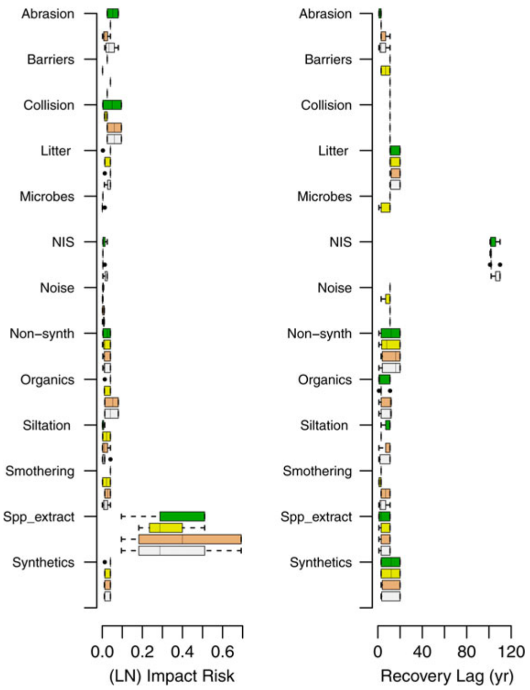

Figure 6. Distribution of IR and RL scores to all ecological components from fishing grouped by pressure in each of four European regional seas (ordered as Baltic Sea, Black Sea, Mediterranean Sea and NE Atlantic). The maximum IR and RL score for any chain is 0.7 and 1.0, respectively. Boxplot information is given in the legend of Figure 3.

Grouping impact chains by sector or pressure for a single ecological component can be used to illustrate specific risks. Focusing on sublittoral sediments (Figure 7), the IR from the majority of sectors is low, although some sectors such as aggregate extraction, aquaculture, fishing, and navigational dredging introduce impact chains of higher risk. Fishing, in particular, introduces impact chains of especially high risk in the Baltic Sea, Mediterranean Sea, and NE Atlantic regions, indicating widespread, frequent, and severe interactions with the seafloor as a result of this sector. Grouping by pressure type revealed the pressures driving those high impact scores, i.e. aggregate extraction and species extraction, and pressures of particular regional importance such as sealing in the Mediterranean Sea (a pressure linked to a number of sectors such as coastal infrastructure and tourism recreation) (Figure 7).

Discussion

We have illustrated how a generic exposure-effect framework can be used to assess the risk to and recovery of ecosystems from human activities on a scale relevant to current environmental policy. We

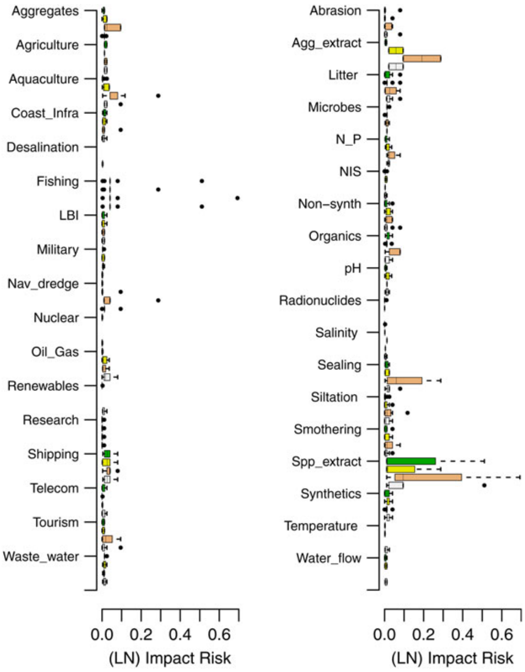

Figure 7. Distribution of IR and RL scores to sublittoral sediments grouped by sector and pressure in each of four European regional seas (ordered as Baltic Sea, Black Sea, Mediterranean Sea and NE Atlantic). Sectors/pressures posing no risk are excluded from the plot. The maximum IR score for any chain is 0.7. Boxplot information is given in the legend of Figure 3.

do this using two datasets: (i) that describes the relationships (linkages) between sectors, pressures, and ecological components of regional sea ecosystems (Knights *et al.*, 2013), and (ii) a qualitative assessment of each linkage using an expert judgement approach (Robinson *et al.*, 2013). The result is two axes of information describing: (i) IR, the likelihood of a negative interaction between a sector and the environment (via the pressure mechanism) and its severity, and (ii) RL, the post-impact rate of recovery to pre-impact condition. The assessment reveals that often, the IR from sector activities is relatively low, but there are a number of impact chains introduced by several sectors of high IR and potentially causing significant harm to the marine environment. Recovery from impact was more variable, but indicated that often, recovery to pre-impact conditions may require many years for some ecological components.

Our framework adopted perhaps the most extensive description of links between human activities and the ecosystem to date (Knights *et al.*, 2013; White *et al.*, 2013). The holistic assessment is therefore relevant to environmental policy and conservation objectives that require an ecosystem approach (McLeod and Leslie, 2009). Here, more than 3500 impact chains were considered forming a complex network of linkages (Knights *et al.*, 2013), which was simplified by grouping chains by "sector", "pressure type", or "ecological component". We presented the results in two ways to demonstrate the flexibility of the approach to identify the impact chains posing the greatest risk and/or slowest recovery. First, in broad terms considering all sectors, pressures, and ecological components, then second, in a more targeted way wherein risk and recovery from a specific sector's impacts or to a single ecological component were assessed. The criteria used to assess each impact chain were relatively coarse (Robinson *et al.*, 2013), but changes in IR/RL could be differentiated within and between groupings (e.g. sector, pressure type, component), allowing managers to take the first step in screening for risks (Knights *et al.*, 2014a); a process which can then be followed by managers prioritizing impact chains for management (Bottrill *et al.*, 2008; Piet *et al.*, in press) based on IR and/or the expected time frame for recovery, assuming that management is effectively implemented, enforced, and complied with (Knights *et al.*, 2014b). Given that management resources are often finite and therefore insufficient to address all issues (Joseph *et al.*, 2009), the framework therefore can act as a decision-support tool (Fletcher, 2005). Managers can then defend management trade-off decisions based on scientific evidence by linking the management measure to a specific conservation objective, as well as identifying the societal and economic costs and benefits of that decision from the outset, which are deemed critical components to the success of an ecosystem approach (Altman *et al.*, 2011; Game *et al.*, 2013; Knights *et al.*, 2014a).

The risk assessment was underpinned by a structured expert judgement analysis of linkages, which is effective for achieving consensus between groups of individuals (Brown, 2002; Cooke and Goossens, 2004). A significant benefit of such an approach is that it can be applied in all systems; even those that are datapoor, and undertaken at relatively low financial cost to the stakeholder (Fletcher *et al.*, 2010). This is of particular value to regions such as the Mediterranean Sea and Black Sea where they not only face the challenge of implementing EBM as obligated under regional sea environmental policy, but have the added complication that the resources (e.g. stocks that straddle international boundaries) are also exploited by stakeholders not bound by the same environmental regulations or ambition levels creating uncertainty and may counteract any management measure(s) implemented by the EU Member State(s) (Stokke, 2000). To counteract the uncertainty surrounding the exploitation of resources by non-EU stakeholders, the assessment can be undertaken using a precautionary approach and use data such as anecdotal evidence to support the pressure evaluation in lieu of empirical data. A manager is then not precluded from making an assessment of regional priorities, but includes uncertainty such that risk to ecosystems is not underestimated.

We applied the risk assessment to the suite of sectors, pressures, and broad ecological components that are common to global marine ecosystems; the ecological components assessed are representative of a healthy ecosystem (Costanza and Mageau, 1999) and have been identified as relevant characteristics of Good Environmental Status (GES) under the MSFD. We can therefore interpret directly from our analysis the risk to the ecosystem from different sectors (Fletcher *et al.*, 2010; Samhouri and Levin, 2012). Application of the risk assessment framework identified the sectors and pressures that are recognized as primary drivers of change in the ecosystem and its components. There were cross-regional similarities in risk and included well-recognized primary sector drivers of ecosystem change such as commercial fishing (e.g. Piet and Jennings, 2005; Coll *et al.*, 2010) and coastal infrastructure (Bulleri and Chapman, 2009), and perhaps less well-recognized sectors such as navigational dredging (Suedel *et al.*, 2008) and tourism (Davenport and Davenport, 2006). Many of the pressure types with higher risk scores are also well recognized, such as selective extraction from fishing (Pauly *et al.*, 1998) and nitrogen and phosphorus run-off from agriculture (Zillen *et al.*, 2008). These were linked to high-risk sectors (e.g. Graneli *et al.*, 1990; Smayda, 1990), which is unsurprising given that direct links can be made between sector–pressures and ecological components (Knights *et al.*, 2013; Liu *et al.*, 2007). As the underlying assessment of the linkages, considered prevailing conditions, results indicate that the regulation of some sector activities have failed to limit their impact as intended (e.g. Khalilian *et al.*, 2010), and elsewhere, harmful impacts have been ignored (Walker *et al.*, 2003).

The assessment was also able to identify and prioritize sectors and pressures that are of region-specific concern. For example, in the Baltic Sea, the effects of N&P are longer lasting than in other regions (Figure 4). Although direct impacts on ecosystem components are relatively low risk, indirect effects are numerous and of greater concern but which were not assessed here. Nutrient enrichment by persistent point source introductions coupled with extremely low turnover rates in soils and sediments has led to nutrients being released for decades beyond cessation of discharges in the Baltic Sea region (HELCOM, 2010) and can have lasting effects on many characteristics of the ecosystem (Graneli *et al.*, 1990; Smayda, 1990; Moncheva *et al.*, 2001; Diaz and Rosenberg, 2008). As such, eutrophication is a heavily targeted issue in the Baltic Sea, with management in place to limit or prevent further introductions of nutrients (HELCOM, 2010).

The number of high-risk impact chains introduced by different sectors reinforces the need for holistic management, which adopts a combination of management measures to achieve the objectives of the ecosystem approach (Tallberg, 2002; Knights *et al.*, 2013). The protection of some components is likely to be easier to achieve than for others (Khalilian *et al.*, 2010). For example, an improvement in sublittoral habitat state (Figure 7) would likely require the management of fishing, aggregates, aquaculture, navigational dredging, and research (including scientific research and bio-prospecting) sectors (Figure 7), whereas pelagic fish species are threatened by fishing, tourism, research, and aquaculture. Reductions in risk would therefore likely require different (and most likely more complex) levels of control. Identifying combinations of management measures to reduce risk are outside the scope of this paper (see Piet *et al.*, submitted to this journal for such an assessment), but the analysis does indicate that the complexity of management strategies required to reduce risk will be dependent, not only on the region, but also the conservation objective. Although not undertaken here, the approach could be used to evaluate management strategies by assessing the reduction in risk to the ecosystem or targeted characteristics. Risk reductions could be achieved in several ways via changes in exposure or sensitivity or a combination of the two (Smith *et al.*, 2007). Managers would then be able to make trade-offs and develop more socially acceptable management strategies (Hassan *et al.*, 2005), which can lead to greater compliance (Tallberg, 2002), a reduction in enforcement costs (Sutinen and Soboil, 2003), and an increased likelihood of reaching the environmental objective.

A limitation of the approach was that intensity was not explicitly included within the pressure assessment, although part of the definition of the sensitivity criterion "degree of impact" (see Robinson *et al.*, 2013, for a full description). This was reflected in the regional assessments by identification of the pressures "Introduction of synthetic compounds" and "Introduction of non-synthetic compounds" as higher RL issues (Figure 4). Although both pressure types have the potential to cause widespread and catastrophic impacts when and where they occur (Peterson *et al.*, 2003; Korpinen *et al.*, 2012), the intensity of introduction tends to be relatively low and generally fails to exceed the concentration required for adverse impacts (see low IR scores; Figure 4) despite widespread, low-intensity introductions being common (Robinson *et al.*, 2013). The assessment is therefore precautionary, in that some of the issues highlighted may not be of immediate concern unless a rare or catastrophic event was to occur (Peterson *et al.*, 2003).

Limited fiscal resources, ever increasing demands for resources (Hallerberg *et al.*, 2007; Halpern *et al.*, 2008) and the complex relationship between humans and their environment (Liu *et al.*, 2007) are significant challenges to EBM. Risk assessment is gaining momentum as a decision-support tool that allows managers and policy-makers to prioritize human drivers of environmental change (Fletcher, 2005; Fletcher *et al.*, 2010; Hobday *et al.*, 2011; Samhouri and Levin, 2012) and makes a basic contribution towards EBM objectives. The development of a reliable risk assessment has been challenging because of the inherent complexity associated with multiple sectors targeting multiple ecosystem characteristics (resources) making attributing risk to specific sectors and their activities difficult. The approach illustrated here provides a rapid, structured, transparent assessment of current risk to ecosystems so that resource managers on the national, international, or regional stage can identify the most harmful activities and potential management measures suggested and corresponding science-based time frames for improvement such that confidence in the stewardship of resources by managers is built (Knights *et al.*, 2014a). Coupled with an evaluation of the costs and benefits regarding the impact of a measure on the environment, societal, and economic metrics (Hassan *et al.*, 2005) will increase the likelihood that the overarching objective of EBM, sustainable use, is achieved.

Supplementary data

Supplementary material is available at the *ICESJMS* online version of the manuscript.

Acknowledgements

This study was funded by the EU FP7 programme "Options for Delivering Ecosystem-based Marine Management" (ODEMM; grant number 244273; www.liv.ac.uk/odemm). We also thank Rob Marrs (University of Liverpool) and Tony Smith (CSIRO) for their helpful comments on this paper.

References

Airoldi, L., and Beck, M. W. 2007. Loss, status and trends for coastal marine habitats of Europe. Oceanography and Marine Biology, 45: 345–405.

Altman, I., Blakeslee, A. M. H., Osio, G. C., Rillahan, C. B., Teck, S. J., Meyer, J. J., Byers, J. E., *et al.* 2011. A practical approach to implementation of ecosystem-based management: a case study using the Gulf of Maine marine ecosystem. Frontiers in Ecology and the Environment, 9: 183–189.

Astles, K. L., Holloway, M. G., Steffe, A., Green, M., Ganassin, C., and Gibbs, P. J. 2006. An ecological method for qualitative risk assessment and its use in the management of fisheries in New South Wales, Australia. Fisheries Research, 82: 290–303.

Bax, N. J., and Williams, A. 2001. Seabed habitat on the south-eastern Australian continental shelf: context, vulnerability and monitoring. Marine and Freshwater Research, 52: 491–512.

Bottrill, M. C., Joseph, L. N., Carwardine, J., Bode, M., Cook, C. N., Game, E. T., Grantham, H., *et al.* 2008. Is conservation triage just smart decision making? Trends in Ecology and Evolution, 23: 649–654.

Breen, P., Robinson, L. A., Rogers, S. I., Knights, A. M., Piet, G., Churlova, T., Margonski, P., *et al.* 2012. An environmental assessment of risk in achieving good environmental status to support regional prioritisation of management in Europe. Marine Policy. doi: 10.1016/j.marpol.2012.02.003

Brown, J. 2002. The World Café: a Resource Guide for Hosting Conversations That Matter. Whole Systems Associates, Mill Valley, CA.

Bulleri, F., and Chapman, M. G. 2009. The introduction of coastal infrastructure as a driver of change in marine environments. Journal of Applied Ecology, 47: 26–35.

Campbell, M. L., and Gallagher, C. 2007. Assessing the relative effects of fishing on the New Zealand marine environment through risk analysis. ICES Journal of Marine Science, 64: 256–270.

Coll, M., Piroddi, C., Steenbeek, J., Kaschner, K., Lasram, F. B., Aguzzi, J., Ballesteros, E., *et al.* 2010. The biodiversity of the Mediterranean Sea: estimates, patterns, and threats. PLoS One, 5: e11842

Cooke, R. M., and Goossens, L. H. J. 2004. Expert judgement elicitation for risk assessments of critical infrastructures. Journal of Risk Research, 7: 643–656.

Costanza, R., and Mageau, M. 1999. What is a healthy ecosystem? Aquatic Ecology, 33: 105–115.

Davenport, J., and Davenport, J. L. 2006. The impact of tourism and personal leisure transport on coastal environments: a review. Estuarine Coastal and Shelf Science, 67: 280–292.

Diaz, R. J., and Rosenberg, R. 2008. Spreading dead zones and consequences for marine ecosystems. Science, 321: 926–929.

Dunne, J. A., Williams, R. J., and Martinez, N. D. 2002. Network structure and biodiversity loss in food webs: robustness increases with connectance. Ecology Letters, 5: 558–567.

EC. 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). Official Journal of the European Union, pp. 19–40.

Elliot, J., Heesterbeek, S., Lukensmeyer, C. J., and Slocum, N. 2005. Participatory Methods Toolkit: a Practioner's Manual. King Baudouin Foundation and the Flemish Institute for Science and Technology.

Elliott, M. 2002. The role of the DPSIR approach and conceptual models in marine environmental management: an example for offshore wind power. Marine Pollution Bulletin, 44: Iii–Vii.

Fletcher, W. J. 2005. The application of qualitative risk assessment methodology to prioritize issues for fisheries management. ICES Journal of Marine Science, 62: 1576–1587.

Fletcher, W. J., Shaw, J., Metcalf, S. J., and Gaughan, D. J. 2010. An ecosystem-based fisheries management framework: the efficient, regional-level planning tool for management agencies. Marine Policy, 34: 1226–1238.

Francis, R. I. C. 1992. Use of risk analysis to assess fishery management strategies - a case-study using orange roughy (*Hoplostethus atlanticus*) on the Chatham Rise, New Zealand. Canadian Journal of Fisheries and Aquatic Sciences, 49: 922–930.

Galil, B. S. 2003. Control and eradication of invasive aquatic invertebrates. *In* Encyclopedia of Life Support Systems (EOLSS). Ed. by F. Gherardi, C. Corti, and M. Gualtieri. EOLSS Publishers, Paris, France.

Game, E. T., Kareiva, P., and Possingham, H. P. 2013. Six common mistakes in conservation priority setting. Conservation Biology, 27: 480–485.

Graneli, E., Wallstrom, K., Larsson, U., Graneli, W., and Elmgren, R. 1990. Nutrient limitation of primary production in the Baltic Sea area. Ambio, 19: 142–151.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., *et al.* 2008. A global map of human impact on marine ecosystems. Science, 319: 948–952.

Hallerberg, M., Strauch, R., and von Hagen, J. 2007. The design of fiscal rules and forms of governance in European Union countries. European Journal of Political Economy, 23: 338–359.

Hassan, R., Scholes, R., and Ash, N. 2005. Ecosystem and Human Well-being. *Millennium ecosystem assessment*. Island Press, Washington, DC.

HELCOM. 2010. Maritime Activities in the Baltic Sea - An Integrated Thematic Assessment on Maritime Activities and Response to Pollution at Sea in the Baltic Sea Region. ICES Document 123.

Hobday, A. J., Smith, A. D. M., Stobutzki, I. C., Bulman, C., Daley, R., Dambacher, J. M., Deng, R. A., *et al.* 2011. Ecological risk assessment for the effects of fishing. Fisheries Research, 108: 372–384.

Holman, I. P., Nicholls, R. J., Berry, P. M., Harrison, P. A., Audsley, E., Shackley, S., and Rounsevell, M. D. A. 2005. A regional, multi-sectoral and integrated assessment of the impacts of climate and socio-economic change in the UK: II Results. Climatic Change, 71: 43–73.

Hope, B. K. 2006. An examination of ecological risk assessment and management practices. Environment International, 32: 983–995.

Joseph, L. N., Maloney, R. F., and Possingham, H. P. 2009. Optimal allocation of resources among threatened species: a project prioritization protocol. Conservation Biology, 23: 328–338.

Khalilian, S., Froese, R., Proelss, A., and Requate, T. 2010. Designed for failure: a critique of the common fisheries policy of the European Union. Marine Policy, 34: 1178–1182.

Knights, A. M., Culhane, F., Hussain, S. S., Papadopoulou, K. N., Piet, G. J., Raakær, J., Rogers, S. I., *et al.* 2014a. A step-wise process of decision-making under uncertainty when implementing environmental policy. Environmental Science and Policy, 39: 56–64.

Knights, A. M., Koss, R. S., and Robinson, L. A. 2013. Identifying common pressure pathways from a complex network of human activities to support ecosystem-based management. Ecological Applications, 23: 755–765.

Knights, A. M., Piet, G. J., Jongbloed, R., and Robinson, L. A. 2014b. An exposure-effect risk assessment methodology to evaluate the performance of management scenarios: case study examples from Europe's regional seas. University of Liverpool, Liverpool. 1–38 pp.

Korpinen, S., Meski, L., Andersen, J. H., and Laamanen, M. 2012. Human pressures and their potential impact on the Baltic Sea ecosystem. Ecological Indicators, 15: 105–114.

La Jeunesse, I., Rounsevell, M., and Vanclooster, M. 2003. Delivering a decision support system tool to a river contract: a way to implement the participatory approach principle at the catchment scale? Physics and Chemistry of the Earth, 28: 547–554.

Leslie, H. M., and McLeod, K. L. 2007. Confronting the challenges of implementing marine ecosystem-based management. Frontiers in Ecology and the Environment, 5: 540–548.

Liu, J. G., Dietz, T., Carpenter, S. R., Alberti, M., Folke, C., Moran, E., Pell, A. N., *et al.* 2007. Complexity of coupled human and natural systems. Science, 317:1513–1516.

McLeod, K. L., and Leslie, H. 2009. Ecosystem-Based Management for the Oceans. Island Press, Washington, DC.

Moncheva, S., Gotsis-Skretas, O., Pagou, K., and Krastev, A. 2001. Phytoplankton blooms in Black Sea and Mediterranean coastal ecosystems subjected to anthropogenic eutrophication: similarities and differences. Estuarine Coastal and Shelf Science, 53: 281–295.

Odermatt, S. 2004. Evaluation of mountain case studies by means of sustainability variables - A DPSIR model as an evaluation tool in the context of the North-South discussion. Mountain Research and Development, 24: 336–341.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., and Torres, F. 1998. Fishing down the marine food webs. Science, 279: 860–863.

Peterson, C. H., Grabowski, J. H., and Powers, S. P. 2003. Estimated enhancement of fish production resulting from restoring oyster reef habitat: quantitative valuation. Marine Ecology-Progress Series, 264: 249–264.

Piet, G. J., and Jennings, S. 2005. Response of potential fish community indicators to fishing. ICES Journal of Marine Science, 62: 214–225.

Piet, G. J., Jongbloed, R. H., Knights, A. M., Tamis, J. E., Paijmans, A., van der Sluis, M., de Vries, P., *et al.* In press. Integrated management strategy evaluation based on risk assessment. Biological Conservation.

Robinson, L. A., White, L., Culhane, F., and Knights, A. M. 2013. ODEMM Pressure Assessment Userguide V.2. ODEMM Guidance Document Series No. 4. University of Liverpool, Liverpool. 15 pp.

Rounsevell, M. D. A., Dawson, T. P., and Harrison, P. A. 2010. A conceptual framework to assess the effects of environmental change on ecosystem services. Biodiversity and Conservation, 19: 2823–2842.

Samhouri, J., and Levin, P. S. 2012. Linking land- and sea-based activities to risk in coastal ecosystems. Biological Conservation, 145: 118–129.

Scheren, P. A. G. M., Kroeze, C., Janssen, F. J. J. G., Hordijk, L., and Ptasinski, K. J. 2004. Integrated water pollution assessment of the Ebrie Lagoon, Ivory Coast, West Africa. Journal of Marine Systems, 44: 1–17.

Smayda, T. 1990. Novel and nuisance phytoplankton blooms in the sea: Evidence for a global epidemic. *In* Toxic Marine Phytoplankton. Ed. by E. Graneli, B. Sundstom, L. Edler, and D. M. Anderson. Elsevier, New York.

Smith, A. D. M., Fulton, E. J., Hobday, A. J., Smith, D. C., and Shoulder, P. 2007. Scientific tools to support the practical implementation of ecosystem-based fisheries management. ICES Journal of Marine Science, 64: 633–639.

Stobutzki, I., Miller, M., and Brewer, D. 2001. Sustainability of fishery bycatch: a process for assessing highly diverse and numerous bycatch. Environmental Conservation, 28: 167–181.

Stokke, O. S. 2000. Managing straddling stocks: the interplay of global and regional regimes. Ocean and Coastal Management, 43: 205–234.

Suedel, B. C., Kim, J., Clarke, D. G., and Linkov, I. 2008. A risk-informed decision framework for setting environmental windows for dredging projects. Science of the Total Environment, 403: 1–11.

Sutinen, J. G., and Soboil, M. 2003. The performance of fisheries management systems and the ecosystem challenge. *In* Responsible Fisheries in the Marine Ecosystem, pp. 291–310. Ed. by M. Sinclair, and G. Valdimarsson. CASI Publishing, Signal Mountain.

Tallberg, J. 2002. Paths to compliance: enforcement, management, and the European Union. International Organization, 56: 609–643.

Tallis, H., Levin, P. S., Ruckelshaus, M., Lester, S. E., McLeod, K. L., Fluharty, D. L., and Halpern, B. S. 2010. The many faces of ecosystem-based management: making the process work today in real places. Marine Policy, 34: 340–348.

Tansley, A. G. 1935. The use and abuse of vegetational concepts and terms. Ecology, 16: 284–307.

Walker, W. E., Harremoes, P., Rotmans, J., van der Sluijs, J. P., van Asselt, M. B. A., Janssen, P., and Krayer von Krauss, M. P. 2003. Defining uncertainty. A conceptual basis for uncertainty management in model-based decision support. Integrated Assessment, 4: 5–17.

White, L. J., Koss, R. S., Knights, A. M., Eriksson, A., and Robinson, L. A. 2013. ODEMM Linkage Framework Userguide (Version 2). University of Liverpool, Liverpool.

Williams, A., Dowdney, J., Smith, A. D. M., Hobday, A. J., and Fuller, M. 2011. Evaluating impacts of fishing on benthic habitats: a risk assessment framework applied to Australian fisheries. Fisheries Research, 112: 154–167.

Zillen, L., Conley, D. J., Andren, T., Andren, E., and Bjorck, S. 2008. Past occurrences of hypoxia in the Baltic Sea and the role of climate variability, environmental change and human impact. Earth-Science Reviews, 91: 77–92.

Zhang, C. I., Hollowed, A. B., Lee, J. B., and Kim, D. H. 2011. An IFRAME approach for assessing impacts of climate change on fisheries. ICES Journal of Marine Science, 68: 1318–1328.

Handling editor: Mark Gibbs