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Contribution to the Themed Section: 'Risk Assessment'

Introduction

Risk assessment and risk management: a primer for marine scientists

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Risk assessment is the management approach or framework of choice in many disciplines, including health care and research, engineering design, and particularly the insurance sector which relies on the best available forward projections of natural hazards and accidents. The marine management community, which includes researchers, practitioners, and resource managers responsible for individual targeted stocks, aquaculture activities, and the marine environment in general, has been slower to take up quantitative risk assessment approaches. Whilst there are prominent examples where risk assessment and management approaches have been applied, they are relatively few. This article theme set presents examples of such and identifies tools and approaches that can be applied to coastal and oceanic marine systems worldwide. The methods developed and the lessons learned from these studies can be used to guide researchers, practitioners, and resource managers. It is hoped that this article theme set will provide an overview of the current state of risk assessment as applied to marine resource management, and stimulate new thinking on how risk assessment approaches can be applied.

Keywords: ecological risk assessment, environmental impact assessment, expert elicitation, marine and coastal risk assessment.

An overview of approaches to risk assessment

The prominent sociologist Ulrick Beck has linked the modernization of many nations over the last century with the development of the "Risk Society" in which governments, communities, organizations, and individuals focus much of their day-to-day efforts managing risk and, where possible, transfer risk onto others. Beck (1992) argues that this fixation on managing and transferring risk is a defining attribute of our post-modern society, particularly in industrialized nations. Consistent with this view, a number of scientific, public health, sociological disciplines, and technical practices have adopted strong risk management approaches and frameworks (e.g. EFSA, 2012, 2013, 2014, and see Mao *et al.*, 2010). A topical, but unpopular example would be the finance sector whose well-evolved risk management processes are arguably advantageous to particular parts of the finance sector, but less helpful to the remainder of the international investment community (e.g. Moshirian, 2011).

Policy and regulations in the health sector are increasingly risk-based. Health policy is now commonly assessed in terms of indicators that focus on risk to life expectancy, or average number of years forgone or added as a result of proposed policy instruments (Yokota and Thompson, 2004). Risk to remaining years of life is often monetized and directly compared with the cost of healthcare to assess the efficiency and effectiveness of health policy. These approaches are explicitly risk-based as they seek to understand the risks to human lives, or the risks that are mitigated by policy initiatives. Similarly, the engineering design of infrastructure is also increasingly risk-based (e.g. Dai *et al.*, 2002). Structural design needs to balance the likelihood of failure against construction cost, especially in zones where natural hazards such as floods, earthquakes, or tornadoes can occur (Cornell *et al.*, 2002). In such locations, it is generally economically inefficient to have all structures completely immune to all *possible* hazards and, therefore, infrastructure is commonly built to be immune to *probable* hazards. Assessing the difference between

possible and *probable* requires a probabilistic analysis and the results of such risk analyses are often formalized in building standards and codes. For example, the Australian Building Codes now account for very low likelihood extreme cyclone events (known as hurricanes or typhoons in the northern hemisphere) for some parts of Australia. These Codes, which apply to all structures, were developed based on the estimated risk to a range of locations across the north of Australia (Standards Australia, 2006). As a further example, road tunnels in many nations must now conform to safety levels that are several orders of magnitude safer than the likelihood of loss of life on the adjacent sections of highways (Miclea *et al.*, 2007). This increased level of immunity or safety reflects road users' preferences for road tunnels to be safer than the conjoining open highway (Standards Australia, 2011). This risk standard is then used to define what safety equipment is installed in tunnels. Once again, major management decisions are based on the results of risk assessments.

The engineering risk assessment approach is also enshrined in many national and international standards, most recently the ISO 31000 series. ISO 31000 defines the risk assessment process as consisting of determining the risk context, identifying, analysing and evaluating, and then treating risks. Similarly, risk-based approaches have been proposed that incorporate future uncertainties associated with climate change (Jones, 2001).

The insurance sector is the stalwart of risk management. Insurers are highly leveraged around quantitative estimates of the risk of natural and industrial hazards (e.g. Santomero and Babbel, 1997). This is because insurance companies, and especially re-insurance companies, tread a fine balance between earning income from premiums and financial outlays following catastrophic events. As premiums are collected before hazard events, and outlays redeemed following them, insurers seek to set premium levels so that enough revenue can be earned over the long term. These quantitative estimates or predictions of future hazard events are generated through the application of actuarial techniques, which can be regarded as at the forefront of quantitative risk assessment (Embrechts *et al.*, 1999).

In contrast, risk assessments are not commonly applied to the management of natural systems and environments. It can be argued that, at present, ecological risk assessment is a term that is narrowly used for the assessment of impacts of chemical contaminants to the environment, that is, in ecotoxicology. This is although, both globally and regionally, the largest threat or hazard to ecological systems and processes is mostly loss or physical alteration of habitat, invasive species, or direct exploitation (as for targeted species), and not chemical contamination. Therefore, it follows that ecological risk assessment, perhaps the most powerful framework for assessing anthropogenic changes to the environment, is not currently being directed towards assessing and managing the greatest threats facing natural systems. One of the reasons for this is the widespread use of environmental impact assessments (EIAs).

For decades, EIAs have been one of the primary policy instruments for environmental management and have become a globally consistent approach to managing impacts of human activities, including in the coastal and marine environment (e.g. Tullos, 2009, although see the caveats raised by Hedgpeth, 1973). However, much of the advice generated by coastal and marine science practitioners, through the generation of EIAs and similar assessments, are *implicitly* risk-based but often do not *explicitly* follow risk assessment methodologies. This can sometimes be a substantial limitation in environmental assessment methods, but can also sometimes be an opportunity to improve EIAs. The reason this can become problematic is that governing agencies and natural resource managers often operate within explicit risk management frameworks and, therefore, are at times forced to apply quasi-risk assessments in the form of EIAs to formal administrative processes that are expected to be risk-based. It follows, therefore, that embracing a risk assessment approach could lead to increased uptake of scientific advice by natural resource managers, and ultimately lead to better environmental management outcomes. In any case, in the light of the importance of the likelihood of possible consequences occurring, it would be helpful if we moved away from EIAs that contain a comprehensive identification of the possible sources of hazard and consequences but have cursory treatment of the likelihood of the hazards occurring. To illustrate this, Table 1 presents a generalized comparison of the information contained in a typical coastal or marine EIA with information contained in a risk assessment. Despite areas of commonality, there are key differences in the requirements.

Fisheries science and management practitioners have recently begun to apply formal risk management processes to fishing activities (e.g. Hobday *et al.*, 2011). In many ways, this has been a natural extension of quantitative stock modelling and assessment as these approaches already involve the application of statistical approaches for parameter estimation. The uptake of formal risk assessment by fisheries practitioners has presumably been accompanied by an upskilling of individuals and teams. Ideally, this upskilling would have involved consideration of the plethora of literature and guidelines detailing best-practice risk assessment methodologies. However, having said this, a particular characteristic of the discipline of risk assessment is that the terminology and methodologies are often applied in a loose manner—it seems that everyone is a risk assessor. This may be a consequence of Beck's Risk Society where it could be argued that everyone is in fact by default, a risk assessor and manager. However, it does not follow that despite our universal pre-occupation with risk assessment and transference, that we all follow formal and defensible methodologies for achieving these outcomes.

Risk is most often defined as the product of the likelihood or probability of an event occurring, and the consequences of the event if it were to occur. Alternative definitions have been proposed; for example, the influential risk communicator Peter Sandman (www.psandman.com) defines risk as:

$$\text{Risk} = \text{hazard} + \text{outrage},$$

as opposed to the more formal definition of risk as:

$$\text{Risk} = \text{likelihood} \times \text{consequence}.$$

Table 1. Comparison of the characteristics of EIAs and risk assessments.

	Context identified	Risk identification	Risk analysis	Risk evaluation	Risk treatment
Risk assessment	✓	✓	✓	✓	✓
EIA	✓	✓	?	?	✓

The Sandman definition has developed from the perspective of public relations management of industrial accidents whereby the key outcome to be avoided is public outrage. The classical definition, commonly attributed to Blaise Pascal in the 17th century (Bernstein, 1996), defines risk as directly and simultaneously dependent on both the likelihood (commonly expressed as a probability) and the consequence of the hazard occurring.

It is also important to note that risk perception in humans is a psychological process through which an individual digests, correlates, and assesses information about a hazard (Shackleton *et al.*, 2011). Decisions about the perceived severity of the risk are based on, for example, an individual's circumstances, their knowledge of the risk, their personal experiences and beliefs, social norms, and a consideration of the possible impacts that action or inaction may have. Importantly, a scientific assessment of risk may not always be the primary source of information in the risk perception process, either at the individual or societal levels, particularly if this information is not communicated in an appropriate manner. For example, a study of coastal communities vulnerable to flooding found that information from family, friends and local community groups was perceived as more important in assessing flood risk than information from media reports or government agencies (Harvatt *et al.*, 2011). It is vitally important, therefore, that risk assessment procedures include stakeholders in the process and that outcomes are communicated in a clear and intuitive manner.

Despite the long-standing origins of probabilistic risk assessment, it can be argued that many scientific disciplines followed the trail blazed by classical physicists and chemists (underpinned by Newtonian deterministic views of cause and effect) for much of the twentieth century. This dominance acted to downplay the relevance and importance of more statistical-based or probabilistic methodologies for understanding cause and effect such as risk assessment. Having said this, the rise of quantum mechanics as a means of explaining shortfalls in Newtonian physics has helped to promote the application of more probabilistic approaches such as risk assessment. A consequence of this focus on deterministic mechanisms was also that Bayesian statistical approaches, arguably the most appropriate framework for risk assessment, were also rarely applied until relatively recently (e.g. Siu and Kelly, 1998). As highlighted above, the approach of assessing risk in terms of likelihood and consequence is the foundation of a number of risk management disciplines, including financial, emergency, asset, and business continuity management. However, despite the long and distinguished history of risk assessment, its uptake in coastal and marine applications has lagged behind its overall uptake by at least a decade (compare Figure 1 with Figure 1 in Mao *et al.*, 2010).

The core quantitative tasks in risk assessment are the estimation of the likelihood and consequences of a source of risk. The first step in a risk assessment is generally to identify the possible consequences, ranging from immediate and obvious to more far-field and cumulative. The identification of possible cascading consequences is generally a tractable task. This is especially the case where numerical models are able to simulate possible consequences. This can also be achieved through engaging technical specialists to identify potential sources of risk. However, this process of expert elicitation needs to be undertaken with care as being a technical specialist is not a sufficient condition for being risk-intelligent (see, for example, http://www.projectionpoint.com for details on risk intelligence, and see EFSA, 2014). In other words, many technical specialists are clearly experts in their discipline, but can be poor at estimating the likelihood of future events or conditions occurring.

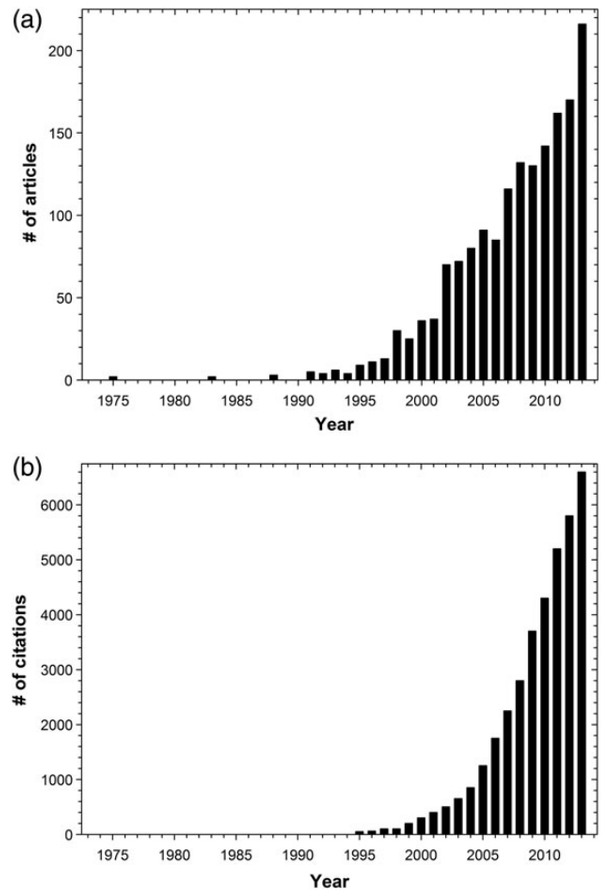

Figure 1. Absolute number of articles (a) and citations (b) during the period 1945 – 2013 that are returned by a search for the string "risk assessment" and "marine" in the Thomson – Reuters Web of Science database. The *x*-axis begins at 1975 because the numbers were zero before then. The search was conducted on 18 August 2014.

Therefore, developing a list of *possible* consequences is a relatively straightforward task. For example, anyone who has been involved with high profile or controversial coastal or marine infrastructure development proposals or natural resource exploitation proposals will have found that opponents to, for example, coastal developments or increases in fisheries allowable catches are very capable of generating long lists of *possible* future impacts. Furthermore, such opponents often then jump to advocating for the precautionary principle (Lauck *et al.*, 1998) to be applied as a justification for not approving projects under the argument that there are so many possible impacts that the risk will be unacceptable. This is a misapplication of risk assessment (and the precautionary principle) since, in such cases, there is no defensible investigation of the likelihood of the impacts occurring and, therefore, no distinction between *possible* and *probable* impacts. In other words, it is relatively easy to come up with a long list of all possible consequences or impacts of a proposal. However, this alone can be somewhat unhelpful unless the corresponding estimate of the likelihood is provided so that *possible* impacts can be distinguished from the *probable* impacts. This does not mean to say that low likelihood but possible impacts need to be ignored; rather it recognizes that estimates of risk require consideration of both the consequence and the likelihood.

Estimating the likelihood of particular consequences is often problematic. This is especially the case for low likelihood, high consequence events. For high likelihood, low consequence events, there are often considerable data and time-series available for analysis. Hence, while low likelihood/high consequence risks are quantitatively the same risk as high likelihood/low consequence risks, the latter are generally easier to assess and hence our confidence in these risk predictions is often greater as we typically have direct personal experience with these events. This fact also means that even subject matter experts can implicitly bias risk assessments towards more frequent and better-known events. For example, there can widespread differences in opinion, even among experts, over the frequency of natural hazards such as earthquakes (e.g. Shearer and Stark, 2012).

There are four general approaches for estimating the likelihood of specified impacts occurring: (i) estimates based on the measured impacts of similar activities in similar environments or contexts, (ii) estimates based on results validated numerically, semi-empirically, or using empirical models, (iii) estimates based on accepted theory of cause–effect mechanisms, and (iv) expert elicitation and opinion of one or more individuals. The robustness or defensibility of estimates of the likelihood of impacts occurring will be maximized if more than one of these methods is applied. For example, if expert opinion alone is used, then this opinion should be developed from the basis of theory, previously applied models, and/or similar examples presented in the scientific literature (see EFSA, 2014).

In effect, the assessment of the likelihood of future events occurring is a prediction. While the basis of the prediction may be at least partly developed from previous occurrences of the impacts or consequences under investigation, fundamentally it is a future-looking prediction and, therefore, subject to the well-documented difficulties that humans have in predicting future events, especially those that occur infrequently (Gregory *et al.*, 1996). It is, therefore, critical that the basis for any prediction be systematic, methodologically sound, and well documented. Such systematic methodology is at the core of formal risk assessment methodologies, and this highlights the criticality that risk assessment practitioners have a deep understanding of these processes. For example, while workshops may be a good approach to elucidating the possible consequences of projects, or changes to regulations, workshops alone may not be suitable for generating estimates of the likelihood of consequences occurring. That is because workshop participants may not be well equipped to compare very low and very high likelihood future events without bias (Yang *et al.*, 2009).

As a result of the difficulties in assessing likelihood, the most robust approach is to follow the prioritization presented above. However, in some cases, detailed previous examples may be absent and, for ecological systems, numerical or even empirical or analytical models may also be unavailable. For physical processes, the movement of water in aquatic environments is well described by the Navier–Stokes equations of motion that can be adequately encapsulated in numerical hydrodynamic models. The same cannot be said for ecological systems as they cannot be fully described by a single set of coupled partial-differential equations. It is this inconsistent approach to assessing likelihood that is a key difference between formal risk assessment and EIA.

A clear benefit in applying risk-based approaches is that the identification of *probable* impacts can be differentiated from a long list of *possible* impacts. However, perhaps one of the most valuable outcomes of a risk assessment exercise is in the identification of gaps in the knowledge/data required to quantify the risk. Thus, even if the exercise cannot adequately quantify the risk *per se*, it will almost always result in a ranked list of research questions that must be pursued to support the risk assessment process.

An article theme set: Risk assessment and risk management in the marine sciences

In the context of the above, and given the widespread adoption of formal risk assessment frameworks in many disciplines, we thought it timely to assess the uptake and application of quantitative risk assessment and risk management approaches in marine and coastal resource and environmental management. To this end, this article theme set presents eight case studies in which risk assessment has been applied to inform and guide the management of coastal and marine systems.

Taranger *et al.* (2015) provide details of the development and implementation of a quantitative risk assessment approach for investigating multiple risks associated with Norwegian aquaculture activities. The results of these analyses provide valuable knowledge for the ongoing management of the aquaculture sector in Norway.

Stelzenmueller *et al.* (2015) applied risk assessment approaches, including the use of Bayesian belief network models, to develop spatially explicit information to underpin marine spatial planning. This approach was able to incorporate impacts and recovery potential for different fishing fleets with different vessel and gear characteristics.

Fletcher (2015) provides a personal view and lessons learned from applying risk assessments to the management of several key stocks and ecosystems. Key lessons learned include the importance of engaging and empowering stakeholders through assessment approaches that are more intuitive so that they can actively understand and participate in the risk assessment process. Similarly, Cortés *et al.* (2015) detail the results of an investigation into how risk frameworks can be applied to the analysis of population dynamics of cartilaginous fish populations. The results of a comparative assessment of different approaches applied to the same stock are provided.

Azmi *et al.* (2015a, b) provide two reports based on a series of biosecurity and pest invasion studies. The global marine biosecurity research and management community have recognized that directing resources to manage incursions is most effective if resources are deployed according to risk. However, this can only be effectively achieved if the assessment of the risk is accurate. These two studies provide examples of how this can be achieved.

Risk screening is commonly applied as the first step in quantitative risk assessments. Cotter *et al.* (2015) undertook an ecological risk screening exercise for fisheries off the Southwest coast of England. This involved extensive stakeholder workshop tasks to both elicit information, and ensure engagement with key stakeholders. The ecological risk screening approach was effective in eliciting and integrating disparate information that can then be used to prioritize management resources.

Knights *et al.* (2015) provide a demonstration of how causal pressure-state linkage approaches can be incorporated into a risk management framework. This approach can be used to investigate the relative magnitude of different impact pathways and thereby identify ecosystem components most at-risk. Finally, Astles (2015) highlights the importance of addressing multiple scales and risk pathways in both the ecological system and the human system, and the need for risk communication throughout both the assessment and management processes. This study provides an approach for transitioning from effective risk assessment to risk management, demonstrated through a case study of an urban estuary.

All of these case studies demonstrate the considerable advantages and utility that risk-based approaches offer. It follows, therefore, that there is considerable potential and scope for risk-based approaches to be applied to the management of marine fisheries, aquaculture, spatial planning, and other activities that occur in coastal and marine systems, in coordination with the direct management of living resources and habitats.

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References

Astles, K. L. 2015. Linking risk factors to risk treatment in ecological risk assessment of marine biodiversity. ICES Journal of Marine Science, 72: 1116–1132.

Azmi, F., Hewitt, C. L., and Campbell, M. L. 2015a. A hub and spoke network model to analyse the secondary dispersal of introduced marine species in Indonesia. ICES Journal of Marine Science, 72: 1069–1077.

Azmi, F., Primo, C., Hewitt, C. L., and Cambell, M. L. 2015b. Assessing marine biosecurity risks when data is limited: bioregion pathway and species-based exposure analyses. ICES Journal of Marine Science, 72: 1078–1091.

Beck, U. 1992. Risk Society: Towards a New Modernity. Sage, New Delhi.

Bernstein, P. L. 1996. Against the Gods: The Remarkable Story of Risk. John Wiley and Sons, New York. ISBN 0-471-12104-5

Cornell, C., Jalayer, F., Hamburger, R., and Foutch, D. 2002. Probabilistic Basis for 2000 SAC Federal Emergency Management Agency Steel Moment Frame Guidelines. Journal of Structural Engineering, 128: 526–533.

Cortés, E., Brooks, E. N., and Shertzer, K. W. 2015. Risk assessment of cartilaginous fish populations. ICES Journal of Marine Science, 72: 1057–1068.

Cotter, J., Lart, W., de Rozarieux, N., Kingston, A., Caslake, R., Le Quesne, W., Jennings, S., *et al.* 2015. A development of ecological risk screening with an application to fisheries off SW England. ICES Journal of Marine Science, 72: 1092–1104.

Dai, F. C., Lee, C. F., and Ngai, Y. Y. 2002. Landslide risk assessment and management: an overview. Engineering Geology, 64: 65–87.

EFSA Panel on Animal Health and Welfare (AHAW). 2012. Guidance on risk assessment for animal welfare. EFSA Journal, 10: 2513. 30 pp. doi:10.2903/j.efsa.2012.2513.

EFSA SC (EFSA Scientific Committee). 2013. Scientific opinion on priority topics for the development of risk assessment guidance by EFSA's Scientific Committee. EFSA Journal, 11: 3345. 20 pp. doi:10.2903/j.efsa.2013.3345

EFSA (European Food Safety Authority). 2014. Guidance on expert knowledge elicitation in food and feed safety risk assessment. EFSA Journal, 12: 3734. 278 pp. doi:10.2903/j.efsa.2014.3734.

Embrechts, P., Resnick, S. I., and Samorodnitsky, G. 1999. Extreme value theory as a risk management tool. North American Actuarial Journal, 3: 30–41.

Fletcher, W. J. 2015. Review and refinement of an existing qualitative risk assessment method for application within an ecosystem-based management framework. ICES Journal of Marine Science, 72: 1043–1056.

Gregory, R., Slovic, P., and Flynn, J. 1996. Risk perception, stigma, and health policy. Health and Place, 2: 213–220.

Harvatt, J., Petts, J., and Chilvers, J. 2011. Understanding householder responses to natural hazards: flooding and sea-level rise comparisons. Journal of Risk Research, 14: 63–83.

Hedgpeth, J. W. 1973. The impact of impact studies. Helgoläander wissenschaftliche Meeresuntersuchungen, 24: 436–445.

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: 2–3.

Jones, R. N. 2001. An environmental risk assessment/management framework for climate change impact assessments. Natural Hazards, 23: 1997–1230.

Knights, A. M., Piet, G., Jongbloed, R. H., Tamis, J. E., White, L., Akoglu, E., Boicenco, L., *et al.* 2015. An exposure-effect approach for evaluating ecosystem-wide risks from human activities. ICES Journal of Marine Science, 72: 1105–1115.

Lauck, T., Clark, C. W., Mangel, M., and Munro, G. R. 1998. Implementing the precautionary principle in fisheries management through marine reserves. Ecological Applications, 8: S72–S78.

Mao, N., Wang, M-H., and Ho, Y. S. 2010. A bibliometric study of the trend in articles related to risk assessment published in Science Citation Index. Human and Ecological Risk Assessment, 16: 801–824.

Miclea, P. C., Chow, W. K., Shen-Wen, C., Junmei, L., Kashef, A., and Kang, K. 2007. International tunnel fire-safety design practices. ASHRAE Journal, 49: 50–60.

Moshirian, F. 2011. The global financial crisis and the evolution of markets, institutions and regulation. Journal of Banking and Finance, 35: 502–511.

Santomero, A., and Babbel, D. F. 1997. Financial risk management by insurers: an analysis of the process. The Journal of Risk and Insurance, 64: 231–270.

Shackleton, E. C. R., Potts, J., and Carter, B. 2011. Residents' perceptions of coastal flood risk and its management through Coastal Defence Strategies at Emsworth, United Kingdom. Littoral 2010. Royal Geographical Society, London. http://dx.doi.org/10.1051/litt/201113001

Siu, N. O., and Kelly, D. L. 1998. Bayesian parameter estimation in probabilistic risk assessment. Reliability Engineering and System Safety, 62: 89–116.

Shearer, P. M., and Stark, P. B. 2012. Global risk of big earthquakes has not recently increased. Proceedings of the National Academy of Sciences of the United States of America, 109: 717–721.

Standards Australia. 2006. Australian Standard AS 4055–2006, Wind loads for housing. Sydney (Australia).

Standards Australia. 2011. Australian Standard AS 4825–2011, Tunnel fire safety. Sydney (Australia).

Stelzenmüller, V., Fock, H. O., Gimpel, A., Rambo, H., Diekmann, R., Probst, W. N., Callies, U., *et al.* 2015. Quantitative environmental risk assessments in the context of marine spatial management: current approaches and some perspectives. ICES Journal of Marine Science, 72: 1022–1042.

Taranger, G. L., Karlsen, Ø., Bannister, R. J., Glover, K. A., Husa, V., Karlsbakk, E., Kvamme, B. O., *et al.* 2015. Risk assessment of the environmental impact of Norwegian Atlantic salmon farming. ICES Journal of Marine Science, 72: 997–1021.

Tullos, D. 2009. Assessing the influence of environmental impact assessments on science and policy: an analysis of the Three Gorges Project. Journal of Environmental Management, 90: S208–S223.

Yang, Z., Coble, K. H., and Hudson, M. D. 2009. The role of individual personality type in subjective risk elicitation outcomes. Journal of Risk Research, 12: 209–222.

Yokota, F., and Thompson, K. M. 2004. Value of information analysis in environmental health risk management decisions: past, present, and future. Risk Analysis, 24: 635–650.

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Contribution to the Themed Section: 'Risk Assessment'

Original Article

Risk assessment of the environmental impact of Norwegian Atlantic salmon farming

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Norwegian aquaculture has grown from its pioneering days in the 1970s to be a major industry. It is primarily based on culturing Atlantic salmon and rainbow trout and has the potential to influence the surrounding environment and wild populations. To evaluate these potential hazards, the Institute of Marine Research initiated a risk assessment of Norwegian salmon farming in 2011. This assessment has been repeated annually since. Here, we describe the background, methods and limitations of the risk assessment for the following hazards: genetic introgression of farmed salmon in wild populations, regulatory effects of salmon lice and viral diseases on wild salmonid populations, local and regional impact of nutrients and organic load. The main findings are as follows: (i) 21 of the 34 wild salmon populations investigated indicated moderate-to-high risk for genetic introgression from farmed escaped salmon. (ii) of 109 stations investigated along the Norwegian coast for salmon lice infection, 27 indicated moderate-to-high likelihood of mortality for salmon smolts while 67 stations indicated moderate-to-high mortality of wild sea trout. (iii) Viral disease outbreaks (pancreas disease, infectious pancreatic necrosis, heart and skeletal muscle inflammation, and cardiomyopathy syndrome) in Norwegian salmon farming suggest extensive release of viruses in many areas. However, screening of wild salmonids revealed low to very low prevalence of the causal viruses. (iv) From ~500 yearly investigations of local organic loading under fish farms, only 2% of them displayed unacceptable conditions in 2013. The risk of eutrophication and organic load beyond the production area of the farm is considered low. Despite several limitations, especially limited monitoring data, this work represents one of the world's first risk assessment of aquaculture. This has provided the Norwegian government with the basis upon which to take decisions for further development of the Norwegian aquaculture industry.

Keywords: environmental impact, eutrophication, genetic interaction, organic load, pathogens, risk assessment, salmon lice.

Introduction

Background

The Atlantic salmon (*Salmo salar* L.) farming industry was first started in Norway in the early 1970s and has now grown to become one of the country's largest export industries by economic value. In addition to Atlantic salmon, which is by far the most significant species farmed in Norway, there are also commercial farming of rainbow trout [*Oncorhynchus mykiss* (Walbaum)] and other marine species such as Atlantic cod (*Gadus morhua* L.) and halibut [*Hippoglossus hippoglossus* (L.)]. In 2012, the production of Atlantic salmon and rainbow trout in Norway was 1 232 095 and 74 583 tons, respectively, and a total of 1006 marine farms

was licensed. These farms are distributed along much of Norway's coastline.

Aquaculture of salmonids in Norway, and other countries where these species are farmed in significant numbers, is primarily based around the production of eggs and juveniles in freshwater facilities on land, combined with grow out of fish in open marine cages. During the last decades, technical standards for the production of aquaculture infrastructure has improved dramatically. However, the primary methods for cultivation of finfish have remained similar, with the size of sea cages (up to 160 m in circumference) and the number of stocked smolt (up to 200 000 individuals per cage) increasing.

The rapid expansion of the aquaculture industry, both in Norway and other regions where this form of open-cage production has increased, has not occurred without environmental challenges. However, although a significant body of evidence suggests various environmental impacts of aquaculture, the rapid expansion of this industry means that management guidelines and targets to address potential negative effects have generally not developed in association with the rapid expansion of the industry. Therefore, there is a need for more coordinated efforts to identify hazards related to open sea cage farming and evaluate environmental risks.

Risk analysis

Several approaches have been suggested and discussed for risk analysis of marine ecosystems and marine aquaculture activities (Anon., 2006, 2010; Nash, 2007; GESAMP, 2008; Samuel-Fitwi *et al.*, 2012), and similar approaches of risk analysis and assessment have been adapted to animal welfare including welfare of farmed fish (e.g. EFSA, 2012). According to GESAMP (2008), a risk analysis should first involve hazard identification, then risk assessment of these hazards including the assessment of release, exposure, and consequences, followed by risk estimation/evaluation. The latter preferably related to politically defined thresholds of acceptability or level of protection. Subsequently, this can be followed up by appropriate risk management and appropriate risk communication.

A full risk analysis is based on the ability to quantify both the probability of a certain event and its consequences, but in biological systems it is normally very difficult to quantify these factors precisely. Hence, risk analyses in biological systems are often conducted using broad qualitative categories, by scoring the probability and consequences from low to high (e.g. GESAMP, 2008). This can in turn be based on some semi-quantitative assessment or on expert opinion as suggested by Anon. (2006).

In 2009, the Norwegian government established a set of environmental goals for sustainability in the "Strategy for an Environmentally Sustainable Norwegian Aquaculture Industry" (Anon., 2009b; Table 1). In response to this, the Institute of Marine Research, Norway, initiated a risk assessment of Norwegian salmon farming in 2010, and yearly since (Taranger *et al.*, 2011a, b, 2012a, 2013, 2014). These risk assessments were based on identified hazards and specific endpoints or proxies related to environmental impacts of salmon farming (Table 2). The endpoints/proxies were in turn derived from the governmental goals for environmental sustainability mentioned above. Moreover, evaluation thresholds for some of these endpoints/proxies (acceptance levels of impact or level of protection) were proposed (Taranger *et al.*, 2012b), and subsequently used in the risk assessments in 2013 and 2014. Here, we describe the way in which these assessments have been conducted, the methodological limitations and challenges, as well as future needs to data and analytical tools.

Table 1. The five primary goals for the future development of the Norwegian aquaculture industry as established by the Norwegian government in 2009.

Goals	
Goal 1: Disease	Disease in fish farming will not have a regulating effect on stocks of wild fish, and as many farmed fish as possible will grow to slaughter age with minimal use of medicines.
Goal 2: Genetic interaction	Aquaculture will not contribute to permanent changes in the genetic characteristics of wild fish populations.
Goal 3: Pollution and discharges	All fish farming locations in use will maintain an acceptable environmental state and will not have higher emissions of nutrient salts and organic materials than the receiving waters can tolerate.
Goal 4: Zoning	The aquaculture industry will have a location structure and zoning which reduces impact on the environment and the risk of infection.
Goal 5: Feed and feed resources	The aquaculture industry's needs for raw materials for feed will be met without overexploitation of wild marine resources.

Hazard identification

The first step in a risk assessment is to identify the most important hazards. A range of criteria for hazard identification was proposed by GESAMP (2008). This includes an analysis on how potential hazards relates to undesirable changes in the environment/ecosystem. To this end, potential hazards are evaluated for their possible severity, extent and duration of the change, either based on past experiences, analogue situations, or models. Some of the environmental challenges (i.e. hazards) identified include ecosystem and benthic community effects of organic loading and nutrients (Buschmann *et al.*, 2006; Kutti *et al.*, 2008; Bannister *et al.*, 2014), transfer of parasites to native populations (Krkošek *et al.*, 2005, 2013a, b; Jackson *et al.*, 2013; Skilbrei *et al.*, 2013; Torrissen *et al.*, 2013; Serra-Llinares *et al.*, 2014), disease interactions (Glover *et al.*, 2013b; Madhun *et al.*, 2014a), and ecological (Jonsson and Jonsson, 2006) and genetic interactions with wild populations (Crozier, 1993; Clifford *et al.*, 1998b; Skaala *et al.*, 2006; Glover *et al.*, 2012, 2013a).

Based on the accumulating evidence of the severity, geographical extent and duration and/or reversibility of the various impacts related to open sea cage salmon farming in Norwegian coastal waters, we have based the current risk assessment on the following hazards: (i) genetic introgression of escaped farmed salmon into wild populations, (ii) impact of salmon lice (*Lepeophtheirus salmonis*) on wild salmonid populations, (iii) potential disease transfer from farmed salmon to wild salmonid populations, and (iv) local and regional impacts of organic load and nutrients from marine salmon farms.

Impact of farmed escapees on the genetic integrity of wild Norwegian populations

Risk assessment

In the following chapter, we have considered the following elements of risk assessment; release, exposure, and consequences, in the following manner. Release assessment is defined as the number of farmed salmon escaping into the natural environment, both as reported and unreported numbers of escapees. Exposure assessment is defined as the physical mixing of farmed escaped salmon on the

Table 2. Identified hazards, process of concern, and endpoint of concern for goals 1–3 for the future development of the Norwegian aquaculture industry as established by the Norwegian government in 2009.

Hazard	Process of concern	Endpoint of concern
Genetic interaction (Goal 2)	Farmed escaped salmon successfully interbreed with wild salmon populations	Changes observed in the genetic characteristics of wild salmon populations
Salmon lice (Goal 1)	Salmon lice from fish farming affects wild fish	Salmon lice from fish farming significantly increase the mortality of wild salmonids
Viral diseases (Goal 1)	Disease transmission from fish farming affects wild fish	Viral transmission from fish farming significantly increase the mortality of wild salmonids
Discharges of organic material: (i) local effects (ii) regional effects (Goal 3)	Emissions of organic materials to the surrounding environment	(i) Unacceptable change in sediment chemistry and faunal communities in the production zone (ii) Significant change in bottom communities beyond the production zone—regional impact
Discharges of nutrients: (i) local effects (ii) regional effects (Goal 3)	Emissions of nutrients to the surrounding environment	(i) Nutrients from fish farms results in local eutrophication (ii) Nutrients from fish farms results in regional eutrophication

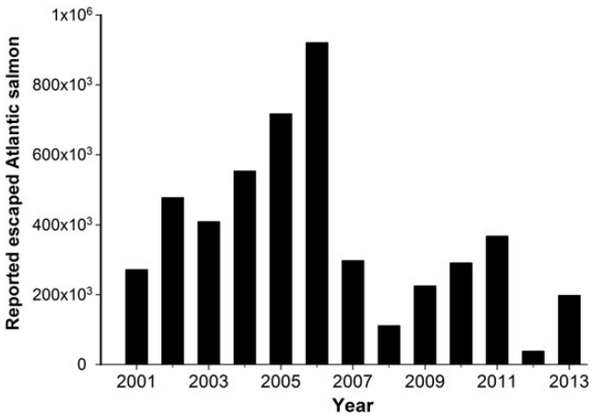

Figure 1. The number of farmed salmon escapes reported to the Norwegian Directorate of Fisheries by fish farmers for the period 2001–2013. Data were taken from the Norwegian Directorate of http://www.fiskeridir.no/.

spawning grounds of wild populations, and the subsequent level of genetic introgression resulting from successful spawning. Finally, we have defined consequence assessment as the consequence of genetic introgression for both the short fitness consequences and the long evolutionary consequences on the native populations.

One of the challenges to the continued development of a sustainable aquaculture industry is containment, and each year, thousands or hundreds of thousands of farmed salmon escape into the natural environment in Norway (Figure 1). Furthermore, the official statistics for numbers of escapees reported to the Norwegian Directorate of Fisheries underestimate the real number of escapees. This has been documented through extensive simulated release experiments and statistical modelling (Skilbrei *et al.*, 2015), and is clearly supported by the fact that the legal authorities in Norway have implemented DNA tracing methods to identify the farm of origin for escapees where they have not been reported (Glover *et al.*, 2008; Glover, 2010; Zhang *et al.*, 2013). While the majority of escapees disappear post-escape (Hansen, 2006; Skilbrei, 2010a, b, 2013), each year, significant numbers of farmed salmon are nevertheless observed in rivers inhabited by wild populations (Fiske *et al.*, 2006, Fiske, 2013). It is therefore considerable potential for genetic interaction between these escapees and native populations.

The Atlantic salmon displays considerable population genetic structure throughout its native range. This variation is partitioned in a hierarchical manner, with the largest genetic differences being observed between populations located in different continents or countries, and the smallest differences being observed among neighbouring populations within regions (Ståhl, 1987; Taggart *et al.*, 1995; Verspoor *et al.*, 2005). This structure reflects various processes, for example recolonization patterns, genetic isolation by distance (Glover *et al.*, 2012), and landscape features which modify population connectivity within regions (Dillane *et al.*, 2008). In addition to differences in allele frequencies of molecular genetic markers, Atlantic salmon populations display different life history characteristics. While much of this phenotypic variation is environmentally caused, some of these differences are influenced by underlying genetic variation, and it is generally accepted that these differences potentially reflect adaptations to their native rivers (Taylor, 1991; Garcia de Leaniz *et al.*, 2007; Fraser *et al.*, 2011).

Norwegian farmed Atlantic salmon dominates global production, originates from over 40 Norwegian rivers, and has been subject to approximately ten or more generations of domestication selection (Gjedrem, 2010). Breeding programmes have successfully selected for fish that outgrow their wild counterparts multiple times under farming conditions (Glover *et al.*, 2009; Solberg *et al.*, 2013a, b). In addition to traits that have been directly selected for in the breeding programmes, genetic changes in non-targeted traits have also been observed, for example in predator awareness (Einum and Fleming, 1997), stress tolerance (Solberg *et al.*, 2013a), and gene transcription (Roberge *et al.*, 2006). In addition, decreased genetic variation, as revealed by molecular genetic markers (Norris *et al.*, 1999; Skaala *et al.*, 2004), and lower estimates of heritability for growth (Solberg *et al.*, 2013a), has been observed in farmed populations. Reduced genetic variation in molecular genetic markers reflects founder effects and genetic drift, driven by limited farmed population sizes, while reduction in heritability for growth is likely to be a result of successful directional selection for this trait over multiple generations.

An early study in Ireland estimated introgression of farmed escaped salmon in a native population based upon escapement

from a nearby farm (Clifford *et al.*, 1998b). However, despite the fact the genetic changes in native wild populations have been observed in molecular genetic markers as a result of farmed salmon introgressing in Canadian (Bourret *et al.*, 2011), Irish (Crozier, 1993, 2000; Clifford *et al.*, 1998a, b), and Norwegian rivers (Skaala *et al.*, 2006; Glover *et al.*, 2012), until a recent breakthrough in a study of 20 Norwegian rivers (Glover *et al.*, 2013a), the cumulative level of introgression of farmed salmon has not been calculated for any native Atlantic salmon population. This is due to the fact that estimation of cumulative introgression of farmed salmon is statistically challenging (Besnier *et al.*, 2011). In the Norwegian study of 20 rivers (Glover *et al.*, 2013a), a combination of Approximate Bayesian Computation, and genetic data for wild-historical, wild-contemporary and a diverse pool of farm samples that were genotyped for a set of collectively informative single-nucleotide polymorphic markers (Karlsson *et al.*, 2011) was used to estimate introgression for a period of 2–4 decades.

Overall, the study by Glover *et al.* (2013a) revealed less introgression of farmed Atlantic salmon in many Norwegian populations (Figure 2) than may be expected based upon the reported numbers of escapees in these populations, and estimations from introgression models (Hindar *et al.*, 2006). The authors concluded that spawning success of farmed escaped salmon has been generally low in many Norwegian rivers, a suggestion consistent with earlier estimates of spawning success in controlled experiments (Fleming *et al.*, 1996, 2000). Nevertheless, results from the study demonstrated high levels of admixture in some native populations, and together with an earlier study using microsatellites, reported decreased genetic differentiation among populations over time (Glover *et al.*, 2012, 2013a). The latter of which is consistent with suggestions that widespread introgression of farmed salmon will lead to erosion of population genetic structure among native populations (Mork, 1991).

Estimating the genetic consequences of farmed salmon introgression on life history traits, population fitness, and long-term evolutionary capacity of wild populations is more challenging than estimating introgression. This is in part due to the fact that wild populations display large natural variation in, for example, marine survival, and at the same time are influenced by a wide range of anthropogenic factors (Parrish *et al.*, 1998), which may potentially mask biological changes caused by introgression of farmed salmon. Nevertheless, comparative studies in Ireland and Norway have demonstrated additive genetic variation for fitness in the wild, with offspring of farmed salmon displaying lower survival than fish of native origin (McGinnity *et al.*, 1997, 2003; Fleming *et al.*, 2000; Skaala *et al.*, 2012). Similar studies conducted on other salmonid species in response to releases of hatchery fish have also arrived at similar conclusions (Araki *et al.*, 2008; Araki and Schmid, 2010).

In summary, the presence of farmed escaped salmon on the spawning grounds of native populations, and the potential for genetic interactions between escapees and wild conspecifics, is of concern. This is because farmed escapees may be genetically different from the recipient wild population for several reasons. (i) Farmed salmon usually do not originate from the same wild population into which they migrate post-escape and will therefore display population genetic differences to the native population. (ii) Farmed salmon have been subject to directional selection and thus differ to all wild salmon for those traits. (iii) Through relaxation of natural selection and inadvertent adaption to the domestic environment, farmed salmon have undergone domestication selection and will also differ to wild salmon.

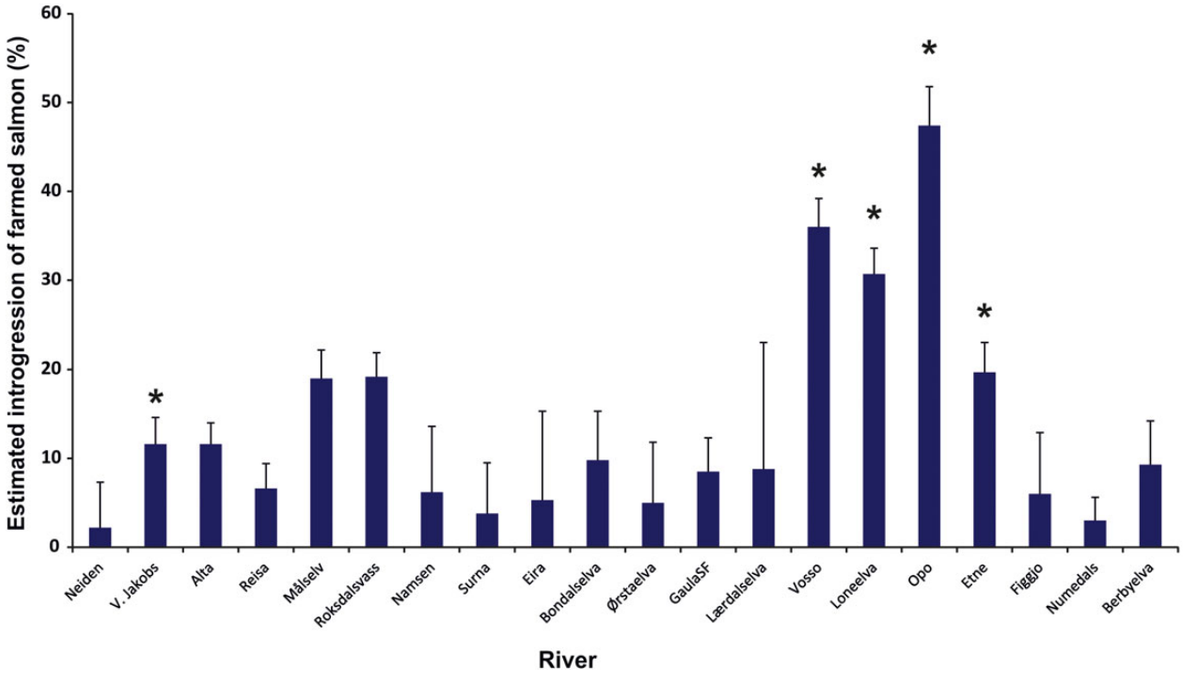

Figure 2. Estimated cumulative introgression of farmed Atlantic salmon in 20 Norwegian Atlantic salmon populations in the period 1970–2008 based upon Approximate Bayesian computation using genetic data. Figure is produced using estimations of admixture from Table 3 in Glover *et al.* (2013a). The computed median level of introgression is 9.1%.

Risk estimation

As part of a national strategy for an environmentally sustainable aquaculture industry (Anon., 2009b), the Norwegian government established the following management goal to prevent genetic interactions of farmed escapees with wild salmon populations: "Aquaculture will not contribute to permanent genetic changes in the genetic characteristics of wild fish stocks" (Table 1). This political target, which forms the basis for the risk estimation, is clearly open for scientific interpretation. However, it was interpreted in a conservative sense for the estimation of risk (Table 2). Thus, any observed genetic change in allele frequencies of molecular genetic markers caused by introgression of farmed salmon would be regarded as permanent genetic change, and therefore in violation of the management goal for sustainability established by the government. The rationale behind this interpretation was first and foremost because molecular genetic markers would be required to directly measure genetic changes in the wild populations. Furthermore, while natural selection will influence the genetic composition of any population, including those where farmed salmon have successfully introgressed, it is unlikely that natural selection will revert the population back to its exact genetic composition before introgression of farmed salmon. This is despite the possibility that natural selection may potentially restore fitness in the natural population.

The documentation of genetic change in a wild population is most directly achieved through the analysis of molecular genetic markers. However, while introgression of farmed Atlantic salmon has been estimated for 20 wild salmon populations in Norway in a 3–4 decade period from 1970 onwards (Glover *et al.*, 2013a), genetic data to estimate introgression of farmed salmon does not exist for the great majority of Norwegian populations. Furthermore, the analysis was being used to address risk of continued and future genetic changes in relation to today's aquaculture industry rather than changes that have already occurred through historical introgression. Therefore, the frequency of farmed escaped salmon observed in wild populations, which is correlated with genetic introgression of farmed escapees over time (Glover *et al.*, 2012, 2013a) was chosen as the indicator to estimate risk of further genetic changes in each wild population for the estimation of risk (Taranger *et al.*, 2012a).

From 2012 onwards, risk was estimated against the below categories for probability of further genetic changes in wild populations caused by introgression of farmed salmon:

No or low risk of genetic change: $<$4% incidence of farmed salmon

Moderate risk of genetic change: 4–10% incidence of farmed salmon

High risk of genetic change: $>$10% incidence of farmed salmon

The threshold values were set according to knowledge of natural straying (reviewed by Stabell, 1984) and knowledge about the present correlation between frequency of farmed fish and corresponding genetic introgression (Glover *et al.*, 2012, 2013a). Stabell (1984) showed that most fish returned to their natal river, although in two of the experiences referred to, straying rates were as high as 10% and nearly 20%. We have chosen a threshold value for no or low risk for genetic change at a frequency of farmed fish in the river corresponding to the lower part of the natural straying estimates (4%), while 10% as high risk of genetic change from the upper part of the distribution. Especially, the threshold value for the upper limit is uncertain and might be modified according to new knowledge about the corresponding correlation between frequency of farmed fish and actual introgression (Glover *et al.*, 2013a; Taranger *et al.*, 2014).

The frequency of farmed salmon in each river surveyed was based upon autumn data where the frequency has been reported for a series of Norwegian rivers (Fiske, 2013). To estimate risk, the frequency of farmed salmon observed in autumn survey was recomputed into an "incidence of farmed salmon" per population using a formula for normalizing data from summer angling catches and autumn surveys (Diserud *et al.*, 2010). This was done because the percentage of farmed salmon in autumn samples is usually higher than in summer angling catches (Fiske *et al.*, 2006), which is in part because farmed salmon enter rivers later than wild salmon. Thus, without normalization of data, the frequency of farmed salmon in summer and autumn surveys are not directly comparable. While many rivers have both summer and autumn estimates, some only have one or the other estimate and therefore require transforming into what has been defined as the "incidence of farmed salmon". The formula for normalizing data from summer and autumn surveys to create the "incidence of farmed salmon" were obtained from Diserud *et al.* (2010) and are presented below:

$$\arcsin\left(\sqrt{(\text{incidence of farmed salmon})}\right) = 0.116 + 0.888 \times \arcsin\left(\sqrt{(\text{summer frequency})}\right)$$

$$\arcsin\left(\sqrt{(\text{incidence of farmed salmon})}\right) = 0.044 + 0.699 \times \arcsin\left(\sqrt{(\text{autumn frequency})}\right)$$

For the risk assessment, the mean incidence of farmed salmon was estimated in 34 rivers distributed along the Norwegian coast using autumn survey data collected in the period 2010–2012. Only rivers having autumn survey data from a minimum of 2 of the 3 years in this survey period were included in the risk assessment. Based on these data, the risk for genetic changes as a result of farmed salmon introgression was low, moderate, or high for 13 (38%), 11 (32%), and 10 (29%) surveyed rivers, respectively (Figure 3).

Limitations

There are a number of challenges and limitations to the conducted risk assessment and estimation, its approach, and calibration against threshold values and potential impacts. One limitation is the fact that the observed frequency of farmed escapees in rivers has been used as the proxy for potential genetic changes in wild populations caused by introgression of the farmed salmon. This is a limitation because the correlation between the frequency of escapees observed in a river, and documented genetic introgression is only modest ($R^2 = 0.47$, $p = 0.0007$) (Glover *et al.*, 2013a). As a result, some rivers display higher and lower levels of genetic introgression from farmed than would be estimated by analysis of the frequency of escapees on the spawning grounds. The consequence of this is that the observed frequency of farmed escapees in each population will not accurately reflect the true risk of genetic changes for all populations, and only by using genetic methods directly will the risk be able to be quantified accurately.

The underlying causes of the lack of a strong relationship between the observed frequency of escapees and genetic introgression are important to identify to help improve the accuracy of the risk assessment in the future. From their genetic study of introgression in 20 Norwegian rivers, Glover *et al.* (2013a) identified both technical and biological elements that are likely to influence the

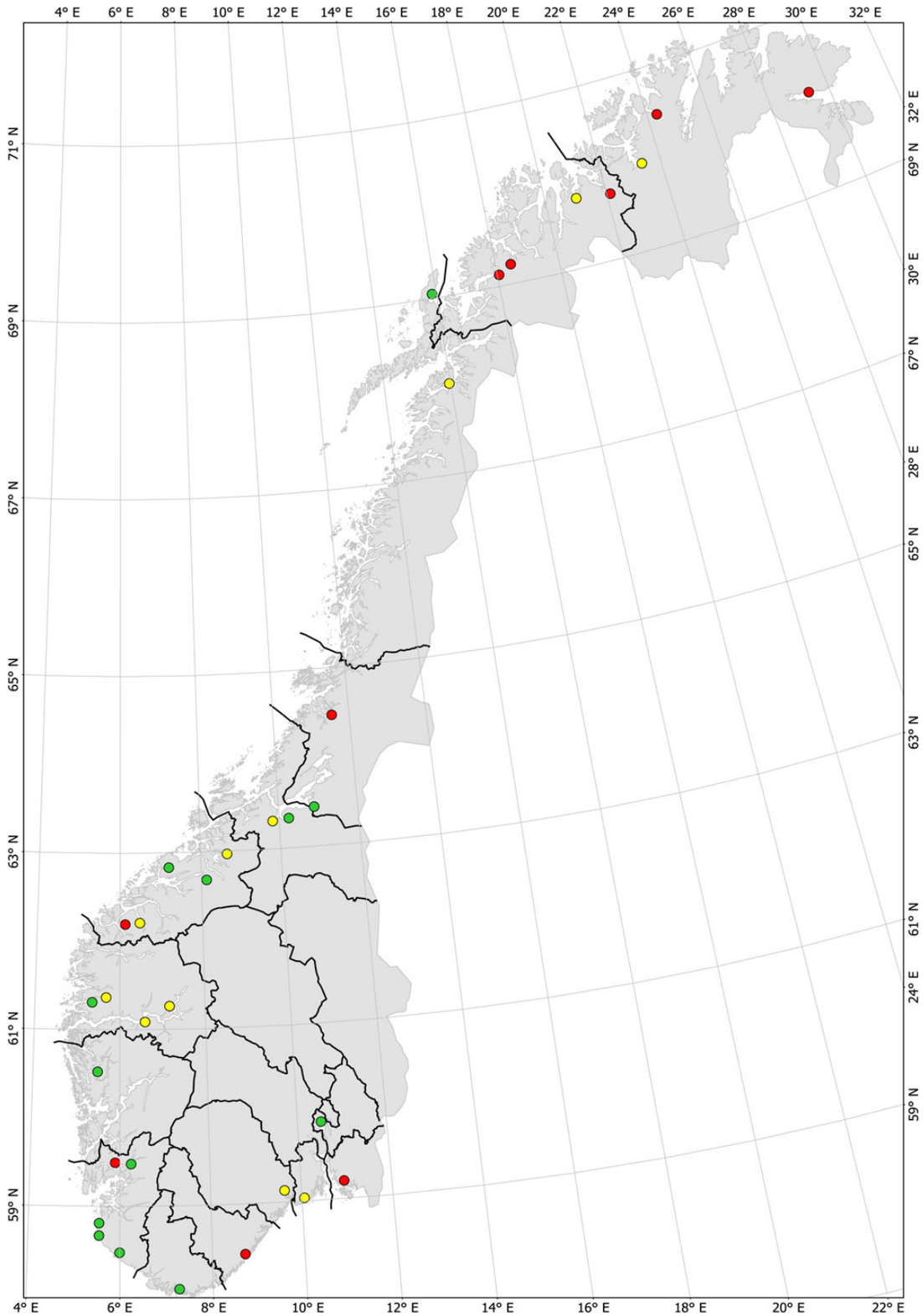

Figure 3. Assessment of risk for genetic changes in 34 wild Atlantic salmon populations in Norway in the period 2010 – 2012. Green point = No or low risk, yellow point = moderate risk, and red point = high risk.

Table 3. Some identified gaps in current knowledge with respect to understanding the potential negative consequences of introgression of farmed salmon in native populations.

Question	Hypothesis to be tested
What is the fitness differential between the offspring of wild, hybrid, and farmed salmon, including multiple generation back-crossed individuals, in different rivers and environments?	To what degree can the results from the few comparative experiments of survival in the wild be generalized for all types of rivers and populations, and is it possible to generalize results?
How strong is natural selection, and will the offspring of farmed salmon readapt to the natural environment?	Lower survival of the offspring of farmed salmon and hybrids in the wild also implies that natural selection will purge poorly adapted individuals from the recipient population, but how strong is natural selection and what is the time-scale of this potential re-adaptation process?
What are the threshold limits of introgression?	What degree of genetic introgression will be tolerated in wild populations before biological, life history and ecological characteristics of the population, and population productivity are compromised?
What is the underlying genomic architecture of domestication and local adaptation in salmonids?	What genetic changes have occurred during domestication, is it possible to identify genetic markers linked with these changes, and fitness in the wild to provide more accurate measurements of functional genetic changes in native populations?

strength of the relationship between the observed frequency of escapees and genetic introgression. Two of the primary components suggested were potential inaccuracies in the frequency of escapees reported for each river (e.g. limited, biased, or non-standardized sampling or reporting), and the fact that the density of the native population, especially on the spawning grounds, may also influence relative success of farmed escapees through spawning competition (Fleming *et al.*, 1996, 2000). In the future, models may be used which include covariables in addition to just the incidence of farmed salmon to predict genetic changes.

A further limitation of the present risk assessment is that it was only conducted for 34 rivers for the 2013 risk assessment. These rivers were chosen as they had autumn survey data published (Fiske, 2013) and thus readily available for assessment of risk for the period 2010–2012. In Norway, there are over 400 salmon rivers and for ~220 of them the status of the stocks are assessed (Anon., 2013b). Thus, the rivers investigated in the current risk assessment only represent a small proportion of those in Norway. Therefore, it is not possible to make clear regional inferences regarding introgression of farmed salmon, only for a small number of specific rivers. Data for the frequency of farmed salmon exist for a larger number of rivers than are currently included in this risk assessment. However, the quality of some of these data, the reporting and availability of the data are highly variable. It was for this reason that the risk assessment was only conducted for the 34 rivers.

Clearly, there is a significant need to increase efforts to expand and improve the monitoring of escaped salmon in a larger number of Norwegian rivers using data gathered and reported in a standardized manner. This will initially be able to improve estimates of the proportion of escaped salmon and will also provide a better foundation for the collection of representative samples for subsequent use in genetic analysis to validate introgression in rivers. An effort to coordinate data collection of escapees has been initiated within Norway in 2014, and in the future it is predicted that the risk assessment will be conducted in a much larger number of rivers.

In addition to technical and data availability challenges linked with the risk assessment and its implementation, there are gaps in current knowledge which limit the ability to identify threshold tolerance limits for introgression of escapees and the level of potentially detrimental effects on the wild populations. Current knowledge points toward a potential negative effect of introgression of farmed Atlantic salmon on the fitness and future evolutionary capacity of recipient wild populations. This is when taking into consideration data available from experimental comparisons of farmed and wild salmon especially in the natural environment (McGinnity *et al.*, 1997, 2003; Fleming *et al.*, 2000; Skaala *et al.*, 2012), background knowledge of salmon biology, life history and ecology, and extensive information from hatchery-fish supplementation for both Atlantic salmon as well as other salmonid species in both the Atlantic and Pacific (Araki *et al.*, 2008; Araki and Schmid, 2010). Nevertheless, significant gaps in understanding of the biological consequences of introgression of farmed salmon remain. These need to be quantified in the future to make a full assessment of risk of biological consequences following introgression of escapes. The major points are summarized in Table 3.

Salmon lice impact on wild salmonids

Salmon lice (*Lepeophtheirus salmonis*) from salmon farms are recognized as an important hazard to wild anadromous salmonids in Norwegian coastal waters (Serra-Llinares *et al.*, 2014). Salmon lice on farmed salmon produce large amounts of planktonic larvae stages that spread via the water currents and can infect migrating Atlantic salmon post-smolts, as well as sea trout (*Salmo trutta*) and Arctic charr (*Salvelinus alpinus*) that stay in coastal waters (Jones and Beamish, 2011). Hydrodynamic models coupled with biological data show that salmon lice can be transported up to 200 km over a 10-d period, although most dispersed 20–30 km (Asplin *et al.*, 2011; Serra-Llinares *et al.*, 2014). The number of salmon lice allowed on farmed salmon is tightly controlled by Norwegian legislation (www.mattilsynet.no). However, the large number of farmed salmon, with ~300 million smolts put into sea cages every year along the Norwegian coast, results in worse case releases in the order of more than a billion salmon lice larvae daily from salmon farms in Norway (Taranger *et al.*, 2014).

New analyses reveal strong correlation between salmon farms and lice infections on wild salmonids in Norwegian coastal waters (Helland *et al.*, 2012; Serra-Llinares *et al.*, 2014). The Norwegian salmon lice monitoring programme on wild salmonids demonstrate annual lice epidemics, most likely connected to the density of salmon farms in the surrounding areas as well as the seasonal dynamics of salmon lice infections on farmed salmon (Jansen *et al.*, 2012; Serra-Llinares *et al.*, 2014; Taranger *et al.*, 2014). A series of experiments has shown that salmon lice may affect anadromous salmonids (reviewed in Finstad and Bjørn, 2011; Anon., 2012; Torrissen *et al.*, 2013).

To assess the risk of salmon lice infection on wild populations, we have considered the following elements of risk assessment: release, exposure, and consequences in the following manner. The release

assessment is based on estimating the production and distribution of infectious salmon lice. The exposure assessment is based on estimating the lice infection on wild salmonid populations using different methods for direct measurements of salmon lice infections on salmon and sea trout. The consequence assessment is the effect salmon lice have on salmonid populations in terms of estimated likelihood of increased marine mortality and/or reduced reproduction based on the exposure assessment.

Risk assessment

All salmon farming sites in Norway report the numbers of salmon lice on the fish weekly when the temperature is >4°C. This is reported together with fish biomass and number of individual salmon per cage (reported each month). Based on the number of sexually mature female salmon lice on the fish on each farming site, the number of infectious salmon lice larvae produced from the different salmon farms are calculated (Jansen *et al.*, 2012). However, at present we do not have enough information and validated models to accurately estimate the impact of the salmon lice infections on wild populations based on reported data from the fish farms (Taranger *et al.*, 2013, 2014).

The lice infection on wild salmonid populations is estimated using different methods as part of a national monitoring programme (Serra-Llinares *et al.*, 2014; Taranger *et al.*, 2014). These methods encompass catch of sea trout and Arctic charr in traps or nets (Bjørn *et al.*, 2011a), and salmon post-smolts caught in special surface trawls in fjord systems (Holm *et al.*, 2000; Bjørn *et al.*, 2007b; Holst *et al.*, 2007). In addition, groups of small sentinel cages containing on average 30 farmed salmon post-smolts are placed in the fjords to monitor the salmon lice infection rate. The fish are kept in the cages for 3 weeks before lice are counted on all the fish, and the procedure repeated three times during spring and summer (Bjørn *et al.*, 2011a, 2013). The sampling programme is focused on areas with high salmon farming activity (Figure 4), as well as some fjords that are protected against salmon farming. The assessment in the period 2010–2013 is based on data from 1 to 5 sites per fjord in 13–16 fjord systems annually, and with increasing numbers of fish sampled at each site in the later years (Table 4). More details about the national salmon lice monitoring programme are provided elsewhere (Helland *et al.*, 2012; Bjørn *et al.*, 2013; Serra-Llinares *et al.*, 2014).

To conduct the risk assessment on the potential impact on salmonid smolts that migrate from the rivers in spring and early summer on the one hand, and the risk of sea trout and Arctic charr that stay in fjords and coastal waters during summer, the national monitoring programme covers two different periods. These periods are adjusted for different timing of smolt migration and seawater residence along the Norwegian coast (Anon., 2011), with an earlier sampling window in the southern part of Norway and later further north. This corresponds to two assumed "critical periods", the first during spring when the salmonid smolts leave the rivers and enters the estuaries and fjords (Period 1), and the second period to estimate the accumulated infection rate on sea trout and Arctic charr that remain in fjords and on the coast during summer (Period 2). See also Anon. (2011) for further information on median migration dates and migration speeds for Atlantic salmon post-smolts in Norwegian fjords.

A range of laboratory studies demonstrate the impact of salmon lice on salmon post-smolts (Grimnes and Jakobsen, 1996; Finstad *et al.*, 2000, 2010; Heuch *et al.*, 2005; Wagner *et al.*, 2008). It has been shown that 0.04–0.15 lice per g fish weight can increase stress levels, reduce swimming ability and create disturbances in water and salt balance in Atlantic salmon (Nolan *et al.*, 1999; Wagner *et al.*, 2003, 2004; Tveiten *et al.*, 2010). In sea trout, ~50 mobile lice are likely to give direct mortality (Bjørn and Finstad, 1997), and only 13 mobile lice, or~0.35 lice per g fish weight might cause physiological stress in sea trout (weight range of 19–70 g; Wells *et al.*, 2006, 2007). Moreover, ~0.05–0.15 lice per g fish weight were found to negatively affect sexually maturing Arctic charr (Tveiten *et al.*, 2010).

According to a review by Wagner *et al.* (2008), infections of 0.75 lice per gram fish weight, or ~11 salmon lice per fish, can kill a recently emigrated wild smolt of ~15 g if all the salmon lice develop into preadult and adult stages. Studies of naturally infected wild

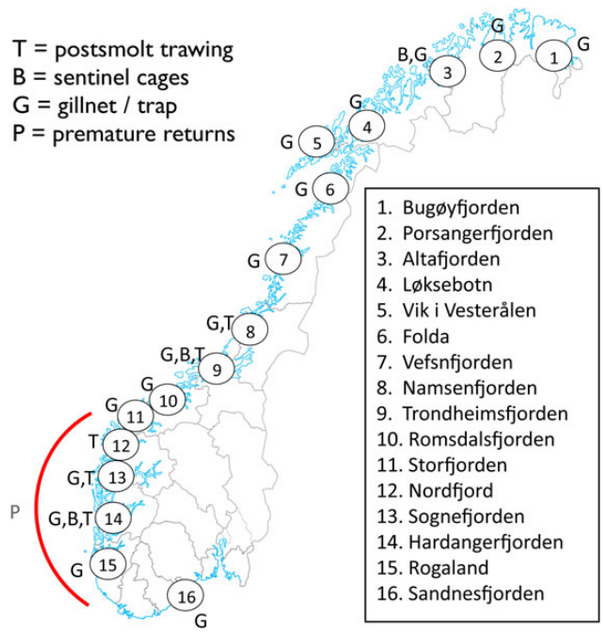

Figure 4. Sampling localities in Norwegian salmon lice monitoring programme applying various techniques such as gillnets and traps, post-smolt trawling, sentinel cages, and recording of premature return to rivers of sea trout during 2010–2013. The total number of sampling localities/sites with gillnets and traps has varied between 26 and 32 in the period.

Table 4. The number of sampling locations and number of wild-caught salmonids (mostly sea trout) investigated for salmon lice infestations in the Norwegian salmon lice monitoring programme during 2010–2013.

	Sites and number sampled in Period 1 (only fish < 150 g)				Sites and number sampled in Period 2 (all fish sizes)			
Year	2010	2011	2012	2013	2010	2011	2012	2013
Locations	26	31	29	23	26	32	29	23
n	218	422	944	1711	623	806	1144	2368

Period 1 covers smolt migration in spring and early summer, whereas Period 2 covers summer period to assess the accumulated effects on sea trout and Arctic charr. Both periods are adjusted different timing of smolt migration and seawater residence along the Norwegian coast, with an earlier sampling window in the southern part of Norway and later further north.

salmon post-smolts indicate that only those with <10 lice survived the infection (Holst *et al.*, 2007). This is consistent with field studies on salmon lice infections in salmon post-smolts in the Norwegian Sea. Over a decade of surveys, no post-smolts was found with >10 salmon lice, and fish with up to 10 mobile lice were observed to be in poor condition with low blood count and poor growth (Holst *et al.*, 2007). New studies of naturally infected wild salmon post-smolts also show that sea lice are fatal at high infections (Berglund Andreassen, 2013). More work in this field is in progress.

Population-wise effects of salmon lice have been demonstrated on wild salmonids in Ireland and Norway. This was studied by protecting individually tagged Atlantic salmon smolts against salmon lice (using Substance EX or Slice) before they were released into the sea near their respective home rivers (Finstad and Jonsson, 2001; Hazon *et al.*, 2006; Skilbrei and Wennevik, 2006; Hvidsten *et al.*, 2007; Skilbrei *et al.*, 2008, 2013; Jackson *et al.*, 2011, 2013; Anon., 2012; Gargan *et al.*, 2012; Krkošek *et al.*, 2013a, b; Vollset *et al.*, 2014). These studies suggest that salmon lice infections increase the marine mortality in areas with intensive salmon farming activity.

Risk estimation

The risk assessment on salmon lice is based on Goal 1 in the policy document on sustainable aquaculture by the Norwegian government from 2009 (Anon., 2009b, Table 1) stating that "Disease in fish farming will not have regulating effect on populations of wild fish...". This target has in this risk estimation been interpreted as "Salmon lice from fish farming significantly increase the mortality of wild salmonids" (Table 2).

A salmon lice risk index, attempting to estimate the increased mortality due to salmon lice infections, was proposed by Taranger *et al.* (2012a) and are based on the assumption that small salmonid post-smolts (<150 g body weight) will suffer 100% lice-related marine mortality, or return prematurely to freshwater for sea trout, in the wild if they are infected with >0.3 lice g^{-1} fish weight. Furthermore, the lice-related marine mortality is estimated to 50%, if the infection is between 0.2 and 0.3 lice g^{-1} fish weight, 20% if the infection rate is between 0.1 and 0.2 lice g^{-1} fish weight, and finally 0% lice-related mortality if the salmon lice infection is <0.1 lice g^{-1} fish weight.

For larger salmonids (over 150 g), we assume that lice-related mortality or compromised reproduction will be 100% in the group if they have >0.15 lice g^{-1} fish weight, 75% for lice infections between 0.10 and 0.15 lice g^{-1} fish weight, 50% for lice infections between 0.05 and 0.10 lice g^{-1} fish weight, 20% for lice infections between 0.05 and 0.01 lice g^{-1} group, and finally 0% if the salmon lice infection is <0.01 lice g^{-1} fish weight.

For both indices, increased mortality risk or compromised seawater growth or reproduction at population level are calculated as the sum of the increased mortalities/compromised reproduction for the different "infection classes" in the sample, reflecting the distribution of the intensity of salmon lice infections of the different individuals sampled. This assumes that individuals caught with traps, gillnets or trawls are representative for the various salmonid populations in that fjord area. The risk was further scored according to the system proposed by Taranger *et al.* (2012a); as low (<10% estimated increase in mortality; green colour), moderate (yellow) for those with between 10 and 30% increase, and high (red) if the increase is calculated as >30%.

The current assessment is based on these scorings, and we use data from the national monitoring programme on salmon infections in wild salmonids (Bjørn *et al.*, 2010, 2011b, 2012, 2013). Separate result tables are presented for the risk for Atlantic salmon smolts (Period 1; Table 5) and for sea trout and the Arctic charr populations (Period 2; Table 6). The results are sorted by county from south to north and by fjord system.

The results indicate considerable variation in risk between years and sampling locations. Moreover, these data strongly indicate a much higher risk for sea trout (and also Arctic charr in the Northern regions) compared with Atlantic salmon post-smolts and reveal moderate-to-high risk of population-reducing effects on sea trout in most counties with high salmon farming activity. The risk of population-reducing effects for Atlantic salmon varies much more between years and sites, and was low at most sites in 2010 and 2013, but moderate and high at several sites 2011 and 2012.

Limitations

The assessment in the period 2010–2013 is based on data from 13 to 16 fjord systems annually. Despite large field effort (Bjørn *et al.*, 2011b, 2012, 2013), the geographical coverage is insufficient in terms of the distribution of salmon farms and wild salmonid populations along the Norwegian coast. There are also problems considering how well the different sampling methods are representative for the different anadromous populations in that area. We have limited data on salmon lice infections in migrating Atlantic salmon smolts, so the risk assessment for salmon is mainly conducted by the use data on salmon lice infections on sea trout caught in traps and gillnets in Period 1 as proxy for the risk to Atlantic salmon post-smolts. Lice infections on trout may not be directly proportional to lice infections on migrating salmon smolt. It is likely that differences in, for example, migratory behaviour and marine ecology exposes salmon and sea trout smolts for different sea lice infection risk (Anon., 2011), even within the same fjord system (Bjørn *et al.*, 2007a, 2011b, c, 2013; Serra-Llinares *et al.*, 2014). The link between individual lice infections and population effects is also very uncertain. There is therefore uncertainty of the current risk assessment both for Atlantic salmon, and for sea trout and Arctic charr. Moreover, the current data are presented without any estimates of uncertainty, which must be included in future analyses.

Future

The "Strategy for an environmentally sustainable aquaculture industry" (Anon., 2009b) states that no disease, including lice, should have a regulatory effect on wild fish. The monitoring of salmon lice infection of wild salmonids is an important verification of whether this goal is achieved, and whether the measures taken are appropriate and sufficient. An indicator system that allows detection of possible problems needed therefore to be established. Hitherto, this assessment has been based on data from the national monitoring of sea lice. An analysis of the historical data in this monitoring series (2004–2010) shows that both the extent and nature of the data have had some weaknesses that limit the ability to analyse and understand the observed variation in infections on wild fish (Helland *et al.*, 2012; Serra-Llinares *et al.*, 2014). Therefore, monitoring and risk assessment based only on lice counts on wild salmonids is not considered sufficient.

Consequently, a rather radical change in the monitoring, advisory and management system for lice has therefore been proposed (Taranger *et al.*, 2012b, 2013, 2014; Bjørn *et al.*, 2013; Serra-Llinares *et al.*, 2014). This system is based on (i) detection and forecasting of increased production of infectious salmon lice using models, (ii) verification of infection pressure through risk-based

Table 5. Estimated risk for lice-related mortality (%) of Atlantic salmon post-smolts based on lice levels on sea trout caught in traps or gillnets and with weight <150 g in Period 1 at the sites from south to north in Norway in 2010–2013.

County	Fjord	Site	2010	2011	2012	2013
Aust-Agder	Sandnesfjord	Sandnes	0	0	0	
Rogaland	Ryfylke	Hellvik		0	0	0
		Vikedal [a2012]		36	20	
		Nedstrand				3
		Forsand		0	0	
Hordaland	Hardanger	Granvin	0	0		
		Ålvik		54	51	0
		Rosendal	0	69	53	13
		Etne	0	0	16	1
Sogn og Fjordane	Sognefjorden	Balestrand	0	0	2	0
		Vik				0
		Brekke/Dingja	0	35	23	0
Møre og Romsdal	Romsdal	Eresfjord	0	0	0	
		Sandnesbukta				22
		Isfjord	0	0	0	0
		Bolsøy [a2010–2012]	2	10	22	15
		Vatnefjorden				0
		Frænfjorden				7
	Storfjord	Sylte	0	0	37	
		Sykkylven	0	0	0	
		Ørsta	0	5	9	
Sør-Trøndelag	Trondheimsfjorden	Skatval [a2010]	6	0	2	0
		Agdenes	0	90	94	0
		Hitra	0	5	0	0
Nord-Trøndelag	Namsen	Tøtdal	0	0	0	0
		Sitter [a2010–2011]	32	24	71	15
		Vikna			7	98
Nordland	Eidsfjord	Vik [a2010–2012]	0	32	50	
	Folda	Ballkjosen	0	25	13	
		Sagfjord	0	7	7	
	Vefsn	Fagervika	0	3		
		Leirfjord	0	0		
	Velfjord	Indre Velfjord [a]			4	
		Ytre Velfjord [a]			4	
Troms	Salangen	Løksa [a2012]		0	4	
Finnmark	Altafjord	Talvik [a2012]	0	0	1	12
		Skillefjord	5	0	3	26
	Bugøyfjord	Bugøyfjord				
	Porsanger	Handelsbukt	0	0		0
		Kåfjord/Repvåg [a2013]	0	0		0

The colour code refers to the assessment of potential population-reducing effect (red = high, yellow = moderate, and green = low).
[a]Small number of fish caught, all fish were used in the assessment irrespective of body size.

Table 6. Estimated risk for salmon lice-related mortality or compromised reproduction (%) of sea trout based on observations in Period 2 at the various sites from south to north in Norway in 2010–2013.

County	Fjord	Site	2010	2011	2012	2013
Aust-Agder	Sandnesfjord	Sandnes	0	2	2	
Rogaland	Ryfylke	Hellvik		0	0	0
		Vikedal		15	0	
		Nedstrand				7
		Forsand		3	0	
Hordaland	Hardanger	Granvin	0	14		
		Ålvik		17	40	32
		Rosendal	55	67	87	38
		Etne	54	3	74	32
Sogn og Fjordane	Sognefjorden	Balestrand	1	0	3	0
		Vik				
		Brekke / Dingja	46	19	72	19
Møre og Romsdal	Romsdal	Eresfjord	0	21	8	
		Sandnesbukta				71
		Isfjord	7	15	25	26
		Bolsøy	14	13	32	73
		Vatnefjorden				98
		Frænfjorden				81
	Storfjord	Sylte	0	0	0	
		Sykkylven	3	10	34	
		Ørsta	25	16	35	
Sør-Trøndelag	Trondheimsfjorden	Skatval	0	13	2	11
		Agdenes	26	40	34	35
		Hitra	8	47	88	41
Nord-Trøndelag	Namsen	Tøtdal	0	9	14	55
		Sitter	65	40	59	62
		Vikna			57	60
Nordland	Eidsfjord	Vik	34	54	59	
	Folda	Ballkjosen	52	45	66	
		Sagfjord	4	52	32	
	Vefsn	Fagervika	19	4		
		Leirfjord	3	0		
	Velfjord	Indre Velfjord			34	
		Ytre Velfjord			28	
Troms	Salangen	Løksa		22	20	
Finnmark	Altafjord	Talvik	3	47	5	18
		Skillefjord	4	55	24	51
	Bugøyfjord	Bugøyfjord		10		
	Porsanger	Handelsbukt	0	0		5
		Kåfjord/Repvåg	0	0		27

Mortality estimates are based on all fish sampled in the period, with different thresholds for small (<150 g) and large (>150 g) fish. The colour code refers to the assessment of potential population-reducing effect (red = high, yellow = moderate, and green = low).

and adapted surveillance on wild salmonids and (c) extended risk assessment based on a considerably larger dataset and fine scale hydrodynamic lice dispersal modelling to assess the effect on wild populations, which then allows adjustment of measures taken by management and industry to reduce this effect to levels within the objective of the strategy. This is now possible due to better knowledge about the relationship between intensive salmon farming activity and infection pressure (Helland *et al.*, 2012; Jansen *et al.*, 2012; Serra-Llinares *et al.*, 2014; Taranger *et al.*, 2014), and better and more accessible farming and environmental data (Jansen *et al.*, 2012; Taranger *et al.*, 2014).

A preliminary analysis indicates that under realistic conditions of lice infections, water currents, temperature and salinity, and relative lice infections may be predicted using a coupled hydrodynamic-biological lice dispersion model. With further calibrations and validation, such a system can probably be developed at least for specific areas along the Norwegian coast. As validation, calibration, and implementation of such a risk-based monitoring system is done, more of the monitoring could be based on the model and less on catch of wild salmonids. The preliminary model results are encouraging in terms of validation and calibration the model predictions against observed infection of wild salmonids. However, considerable research and development remains, where the main challenges are:

- The system for detecting problem areas based on farming data and sea lice infections of notification of problem areas need to be further developed and operationalized, and systems for risk-based and adapted surveillance on wild salmonids must be developed.
- Coupled hydrodynamic-biological lice dispersion models must be validated and calibrated against observed infection levels on wild salmonids in the field.
- Knowledge about the ecological effects of a given infection pressure on stocks of wild salmonids (population-reducing effects) must be increased so that more precise predictions can be developed.

Disease transfer from farmed salmon to wild fish

Background

Infectious diseases represent a major problem in Norwegian fish farming, despite successful development and application of vaccines against a range of pathogens. In addition to lice (considered separately above), viral diseases currently represent the largest disease problems in Norwegian aquaculture (Johansen, 2013). In the period 2005–2012, the four most frequent viral diseases [infectious pancreatic necrosis (IPN), pancreas disease (PD), heart and skeletal muscle inflammation (HSMI), and cardiomyopathy syndrome (CMS)] had 400–500 outbreaks annually (Johansen, 2013). The main reason for the dominance of viral diseases is the lack of effective vaccines. Bacterial diseases, on the other hand, cause only ~20 outbreaks annually, reflecting that the currently used bacterial vaccines provide good protection (Austin and Austin, 2007). Among the parasites, parvicapsulosis due to the myxosporean *Parvicapsula pseudobranchicola* is a problem mainly in northern Norway, whereas heavy gill infections with the microsporidian *Paranucleospora theridion* and the amoeba *Paramoeba perurans* occur mostly in the southern parts of the country. The significance of infections with the former two parasites is unclear, while amoebic gill disease (AGD) has so far been detected only during fall in 3 years (2006, 2012–2013).

For most pathogens, clear evidence for transmission from farmed to wild fish is limited (Raynard *et al.*, 2007). Most of the diseases that currently cause problems in fish farms are likely enzootic, originating from wild fish. This implies that these infections occur or occurred in the past at some "background" level in wild stocks. Such considerations complicate an estimation of the impact of aquaculture, since the "normal" prevalence range of many important disease agents is unknown. However, in two cases exotic pathogens have been introduced in association with farming activities. These have clearly affected wild Atlantic salmon populations.

The ectoparasite *Gyrodactylus salaris* (Monogenea) was first detected in Norway in 1975 (Johnsen *et al.*, 1999). There have been several introductions of *G. salaris* to Norway (Hansen *et al.*, 2003) linked to the import of salmonids from Sweden. Later, the parasite has spread (or has been spread) to many rivers (Johnsen *et al.*, 1999). By 2005, *G. salaris* had been detected in 45 rivers and 39 freshwater farms (Mørk and Hellberg, 2005). Norwegian Atlantic salmon stocks are very susceptible to *G. salaris*, and gyrodactylosis in farmed salmon may lead to 100% mortality if not treated (Bakke, 1991; Bakke *et al.*, 1992; Bakke and MacKenzie, 1993). Mortality in rivers is high, with the density of Atlantic salmon parr being reduced by 50–99% (Johnsen *et al.*, 1999).

Aeromonas salmonicida, the causative agent of furunculosis, was introduced to Norway in 1964, when furunculosis was detected in a single farm that received rainbow trout from Denmark. The disease then spread to other farms and wild fish within a limited area, being detectable there until 1979. A second introduction occurred in 1985, in connection with an import of Atlantic salmon smolts from Scotland. The disease then spread rapidly to farms and wild fish, and in 1992 a total of 550 salmon farms and 74 river systems were affected (Johnsen and Jensen, 1994). This rapid spread of the disease was likely facilitated by frequent escapement events involving infected fish (Johnsen and Jensen, 1994). Mortality due to furunculosis was registered in many rivers among escaped salmon, wild salmon, and trout. Mortality in farmed fish was high, reaching 50%, but the disease was first controlled by antibiotics and subsequently effectively with oil-based vaccines (Sommerset *et al.*, 2005; Johansen, 2013).

These two examples show the devastating effects that introductions of exotic pathogens can have. Even when disregarding agents only known from non-salmonids, there is a large number of potential pathogens (i.e. hazards) infecting salmonids elsewhere that could have significant impact on both salmon farming and wild fish populations in Norway if introduced (Raynard *et al.*, 2007; Brun and Lillehaug, 2010). Import of live fish represents the major threat to both fish farming and wild stocks, since this may lead to the introduction of exotic pathogens. However, *G. salaris* infections have not been detected in Atlantic salmon hatcheries in recent years (Hytterød *et al.*, 2014 and references therein), and the parasite does not survive in seawater. Furunculosis outbreaks in farms are rare, since most farmed salmon is protected through vaccination. Regarding disease transfer from farmed salmon to wild salmonids, these diseases are currently considered to be under control.

The detection of disease in wild fish and estimating disease impact on wild populations is difficult. Clinically affected fish usually disappear quickly in nature (e.g. predated). Epizootics with mass mortality of fish are rare, but have occurred in Norway (Bakke and Harris, 1998; Sterud *et al.*, 2007) and elsewhere (Hyatt *et al.*, 1997; Gaughan *et al.*, 2000). Such episodes are usually caused either by an exotic pathogen introduced to naïve host populations (Bakke and Harris, 1998) or by exceptional environmental

conditions such as high temperature (Sterud *et al.*, 2007). However, infection with native (enzootic) agents under normal environmental conditions can cause disease in individuals and affect an individual's survival or investment in reproduction. Hence, all pathogens may contribute to the regulation of wild populations at some level (May and Anderson, 1979; May, 1983), although the impact may vary and is often the result of a complex interaction between hosts, pathogens, environment, and predators (Dobson and Hudson, 1986; Combes, 2001).

Risk assessment

There is relatively little data available on the infection status of Norwegian wild salmonid stocks with respect to the most important pathogens that affect farmed salmon (e.g. viral agents). The available data mainly concerns returning adult salmon and some local sea trout populations screened with molecular methods (Kileng *et al.*, 2011; Garseth *et al.*, 2012, 2013a, b, c; Biering *et al.*, 2013; Madhun *et al.*, 2014a, b). Studies on the occurrence of viral infections in early life stages of salmonids are only fragmentary (e.g. Plarre *et al.*, 2005).

Due to the limited data available, the disease status (outbreak statistics) in Norwegian fish farming is used as a proxy of the infection pressure from farmed salmon to wild salmonids. Information regarding disease outbreaks on Norwegian fish farms is gathered by the Norwegian Veterinary Institute (NVI) and published annually in their Fish Health Reports (e.g. Johansen, 2013). These data record official diagnoses from NVI, as well as information from the local fish health services. This information is likely to be biased towards the more serious diseases, particularly those that are required by law to be reported to government authorities. Subclinical infections may be common and may also contribute to the spread of pathogens. However, these infections are usually not detected. Despite shortcomings, these data are the best currently available information and give a reasonably good indication of the disease status of the majority of farmed fish in Norway.

Most diseases in Norwegian salmon and rainbow trout farms are represented by only a few outbreaks, often representing geographically separate cases (Johansen, 2013). However, some diseases have a large number of outbreaks/diagnoses, and are those most likely to cause elevated infection pressures that may affect wild populations. At present the most common diseases in Norwegian salmon farming are the viral diseases PD, IPN, CMS and HSMI (Table 7; Johansen, 2013). In addition, AGD due to *Paramoeba perurans* is an emerging problem (Hjeltnes, 2014). The listed viral diseases have caused some 400 or more outbreaks each year since 2005. Outbreaks are often more frequent in certain regions and at certain times of the year, leading to a consideration also of spatial and temporal variation in the potential infection pressure (Table 8).

Table 7. Number of disease outbreaks for the most important diseases in Norwegian salmon farming (Johansen, 2013).

	2005	2006	2007	2008	2009	2010	2011	2012
PD	45	58	98	108	75	88	89	137
HSMB	83	94	162	144	139	131	162	142
IPN	208	207	165	158	223	198	154	119
CMS	71	80	68	66	62	49	74	89

We have considered the following elements of risk assessment regarding the viral agents salmonid alphavirus (SAV), infectious pancreatic necrosis virus (IPNV), piscine myocarditis virus (PMCV), and piscine reovirus (PRV); release, exposure, and consequences as follow. The release assessment is the assumed infection pressure as proxied by the outbreak statistics. The exposure assessment is a consideration of the spatial and temporal concurrence of wild salmonids with release. A consequence assessment should consider two aspects: (i) evidence for virus transmission and (ii) impact of viral infections. However, the impact is in all the considered cases are unknown, and only evidence for virus transmission can be discussed.

Pancreas disease: salmonid alphavirus

PD in Atlantic salmon and rainbow trout is caused by SAV. In Norway, there are currently two regionalized PD epidemics caused by SAV3 (south of Hustadvika, 63° N) and SAV2 (north of 63° N). Experimental studies show transmission of SAV via water, and epidemiological studies provide evidence for horizontal farm to farm spread (Nelson *et al.*, 1995; McLoughlin *et al.*, 1996; Kristoffersen *et al.*, 2009; Stene, 2013; Stene *et al.*, 2014). The virus has been shown to survive for several weeks in the environment (Graham *et al.*, 2007) and thus may be carried long distances with currents (Stene, 2013; Stene *et al.*, 2014). SAV2 may have a different outbreak pattern than SAV3, since outbreaks tend to occur later in the year (Johansen, 2013). For the southern region (SAV3), the period in which smolts migrate and adult salmon return coincides with many SAV3 outbreaks (Table 8; Johansen, 2013). In the northern region (SAV2), most outbreaks occur later in the year. This may signify that most of the smolt migration precedes peak virus spread in the SAV2 region. On the other hand, returning salmon and sea trout are likely more exposed, but screening indicates that very few wild fish are infected (Biering *et al.*, 2013). Infected escaped salmon can enter rivers in fall, possibly exposing wild fish including naïve juveniles to the virus (Madhun *et al.*, 2014a).

Screening of sea trout (Biering *et al.*, 2013; Madhun *et al.*, 2014b) indicates that sea trout in areas with high frequency of PD outbreaks are not infected with SAV. This is in accordance with injection

Table 8. Overview of the main periods where salmon and sea trout reside in coastal areas.

	Coastal area	J	F	M	A	M	J	J	A	S	O	N	D
Smolt migration	South Norway					xx	xx	x					
	Central Norway					x	xx	x					
	Northern Norway						x	xx					
Return	South Norway					x	xx	xx	x				
	Central Norway					x	xx	xx	xx	x			
	Northern Norway					x	xx	xx	xx	x			
Sea Trout (sea)	South Norway				x	xx	xx	xx	x	x	x		
	Central Norway				x	xx	xx	xx	x	x	x		
	Northern Norway				x	x	xx	xx	x	x			

Southern Norway: south Norway up to Sogn and Fjordane, Central Norway: Møre and Romsdal-Trøndelag; Northern Norway: Nordland-Finnmark. x = a few fish in coastal areas, xx = large numbers of fish in coastal areas.

experiments, which suggest that sea trout is more resistant to SAV than salmon (Boucher *et al.*, 1995). SAV infections have been detected in wild salmonids and wild flatfish (Nylund, 2007; Snow *et al.*, 2010; Biering *et al.*, 2013), but PD has not been observed in wild fish.

Altogether, there are yet no data that confirm SAV transmission from farmed salmon to wild fish, but transmission of virus to wild fish is considered likely due to the large number of outbreaks and the documented efficient horizontal transmission of SAV. The probability of transmission of SAV to wild salmon is considered to be moderate for migrating smolts in the southern PD-region due to the temporal overlap between outbreaks and migration, whereas it is considered to be low in the northern PD-region as most of the migration is finished before the major outbreak period. For returning salmon, the probability of infection is considered to be low in both PD-regions based on the available screening results. The probability of SAV transmission to wild salmon is considered to be low in areas with no or few outbreaks. The probability of infection of sea trout during the marine phase is also considered to be low.

Infectious pancreatic necrosis: infectious pancreatic necrosis virus

IPNV is a robust, long-lived birnavirus that infect many different fish species in both fresh water and seawater (e.g. Reno, 1999). The virus is enzootic in Norway. IPN cause significant losses in fish farming in most areas in Norway (Johansen, 2013). However, there are indications of a downwards trend in outbreaks and losses, which might be caused by the increased use of IPN resistant fish (Johansen, 2013). The virus is shed into the water by infected fish, and is spread to other farms by water currents (see, e.g. Mortensen, 1993; Wallace *et al.*, 2005; Raynard *et al.*, 2007; Johansen *et al.*, 2011). A higher prevalence of IPNV has been found in wild fish near salmon farms with clinical outbreaks of IPN, compared with fish at distant sites (Wallace *et al.*, 2005, 2008). Fish surviving an IPNV infection often become persistent carriers of the virus, but viral shedding from carriers has not been demonstrated (Johansen *et al.*, 2011). The prevalence of IPNV in wild fish is low (Brun, 2003; Wallace *et al.*, 2005), and farmed fish are probably a major source of virus in the marine environment. Disease outbreaks in wild salmon have not been described, but mortality in wild marine fish due to IPN has been reported elsewhere (Stephens *et al.*, 1980; Mcallister *et al.*, 1984).

Due to the large number of annual outbreaks, the demonstrated robustness and infectivity of the virus, as well as the wide range of hosts, the probability of IPNV transmission from farmed to wild fish is considered as moderate. The extent and consequences of such transmission are unknown.

Heart and skeletal muscle inflammation: piscine reovirus

HMSI affects farmed salmon along the entire coast of Norway. HSMI outbreaks mainly occur 5–9 months after sea-transfer. The causative agent is an *Orthoreovirus*, PRV (Palacios *et al.*, 2010; Løvoll *et al.*, 2012). The disease can be produced experimentally using infected tissue, infected cell culture, or by cohabitation (Kongtorp *et al.*, 2004; Martinez-Rubio *et al.*, 2012, 2013). PRV is present in high densities in salmon with HSMI, but high infection intensity can be found also in clinically healthy salmon. PRV infections have been detected in wild salmon along the entire coast of Norway and have been detected in sea trout (Biering *et al.*, 2013; Garseth *et al.*, 2013b). Analyses of PRV genotypes in wild Atlantic salmon, farmed salmon, and sea trout have suggested an extensive spread of the virus along the coast, and establishment in wild populations. This spread is probably due to extensive transportation of fish between areas over a long period (Garseth *et al.*, 2013c). However, there are no reports of HSMI in PRV infected wild salmonids (e.g. Garseth *et al.*, 2013b). In Norway sea trout are only rarely (1.4–3%) infected (Garseth *et al.*, 2012; Biering *et al.*, 2013) with PRV. PRV infections have been detected in some marine fish species, but the virus genotype is unknown (Wiik-Nielsen *et al.*, 2012). It is not known how long and in what quantities PRV is shed from infected fish nor viral survival in seawater. However, modelling suggests that the virus can be transported over longer distances than SAV (Aldrin *et al.*, 2010; Kristoffersen *et al.*, 2013). The latter findings suggest that the virus is relatively stable and may spread over large areas.

Since PRV infections are widespread in farmed salmon, may readily be transmitted, and is detectable in >10% of the wild salmon examined, it is considered likely that PRV is transmitted from farmed to wild salmon. HSMI occurrence in wild salmon and other wild fish are unknown (Garseth *et al.*, 2013b, Madhun *et al.*, 2014b).

Cardiomyopathy syndrome: Piscine myocarditis virus

CMS is a serious disease in salmon, and is caused by a Totivirus, PMCV (Løvoll *et al.*, 2010; Haugland *et al.*, 2011). CMS can be transmitted experimentally by injecting heart tissue homogenates from diseased fish, PMCV from cell culture, and by cohabitation (Haugland *et al.*, 2011). Infections are long-lasting, with a gradual development of cardiac pathology.

PMCV has been detected in farmed Atlantic salmon along the entire Norwegian coast, but is not as widespread as PRV. PMCV infection has also been detected in a few wild salmon in Norway (Garseth *et al.*, 2012), and CMS-like lesions have been observed in the hearts of wild salmon before the discovery of the virus (Poppe and Seierstad, 2003). Large-sized wild salmon represent the only known natural reservoir for PMCV. There are no studies on shedding of PMCV from diseased fish or carriers, or on virus survival in water.

Due to the large number of hosts, prolonged infections, and the documented virus spread in cohabitation experiments, we assume that the virus is present in the environment and that the infection pressure around farms harbouring the virus is elevated. Examination of wild returning salmon detected only very few (3/1350) infected with PMCV (Garseth *et al.*, 2012; Biering *et al.*, 2013). Such a low prevalence, which may represent natural rather than fish farming-related infections, suggests that PMCV transmission from farmed to wild salmon is infrequent. Therefore, the probability of infection in wild salmon due to virus released from farms is considered low. Due to a general lack of data, particularly regarding young fish, it is not possible to assess the impact of PMCV infection in wild salmon.

Limitations of the analysis

There is a scarcity of data on infections and lack of evidence for disease in wild salmonids for the four viral agents considered. A large number of outbreaks suggest extensive spread of virus, and consideration of timing also often substantiates an exposure of wild salmonids to the viral agents. A serious limitation in the risk assessment is a lack of information on infections due to these agents in wild fish, particularly in salmon smolts. These may be less susceptible than their farmed peers, but may also suffer mortality due to the infections. Such mortality would likely occur through predation (i.e. virus induced). In either case, the returning 1 or 2 sea-winter salmon could be found to be virus free, as is indeed generally the

case regarding SAV. Juvenile fish is often particularly susceptible to viral infections. Escaped virus infected farmed salmon can enter rivers in fall (e.g. Madhun *et al.*, 2014a), where juvenile salmonids may be exposed to released virus. The impact of SAV, PRV, and PMCV infections on juvenile salmonids is unknown. The abundance of escaped salmon in Norwegian rivers (see above) indeed suggests a high potential for interaction at this stage. Hence, due to lack of data, consequence assessments cannot be done for the four viral agents considered.

Here, we have focused on the four major diseases currently prevalent in Norwegian salmon farming. All are viral diseases, and a large number of outbreaks are reported annually. However, low prevalence does not necessarily mean low impact, and diseases that at present are under control might surge and become a threat to wild populations. Subclinical or apparently benign infections may also have unforeseen ecological effects in nature by affecting survival (i.e. predator avoidance) or recruitment.

We have not considered the possible impact of exotic pathogens on Norwegian salmon farming. There are a large number of potential pathogens (i.e. hazards) infecting salmonids elsewhere that could affect farmed and wild salmonids in Norway if introduced (Raynard *et al.*, 2007; Brun and Lillehaug, 2010). Such introductions may be irreversible, and difficult or impossible to contain, and the main risk factor is the movement of live fish or fertilized eggs.

Future work

To evaluate the effects of salmon and rainbow trout farming on the infection status of wild salmonids, there is a need to increase our knowledge about the complete pathogen repertoire (viruses, bacteria, fungi, and parasites) present in wild fish in areas of high- and low-intensity fish farming. Long-term monitoring of selected wild populations would allow detection of changes in the infection status in the population. Experimental challenge experiments with SAV, PRV, and PMCV must be performed on juvenile Atlantic salmon and brown trout in freshwater. This would better allow a consideration of the threat posed by escaped salmon ascending rivers during fall. A more extensive genotyping of virus from wild and farmed fish would improve our understanding of both virus spread and genetic changes in pathogens that may occur in the high host density of farming areas.

Organic load and nutrients from salmon farms

The salmonid aquaculture industry has continuously restructured since 1999, with reductions in the number of farms, increased farm size and relocation of farms to deeper fjord (50–300 m) and current rich coastal aquaculture sites. During this period, the production has doubled (Gullestad *et al.*, 2011), with typical salmon farm produces between 3000 and 5000 tons in a 18 months period in sheltered coastal waters and as much as 14 000 tons at more dynamic coastal sites. This rapid development has led to increased concerns about the environmental impacts both at present and future predicted finfish production levels.

Increased awareness of elevated discharges of nutrients, excess feed, and faeces to the marine environment has resulted in greater scrutiny of the aquaculture industry (Mente *et al.*, 2006; Taranger *et al.*, 2012a). To this end, the Ministry of Fisheries and Coastal Affairs have stated that "the environmental impact of aquaculture must be kept at an acceptable level and be within the assimilative capacity of the area" (Anon., 2009b).

Assessing the risk of organic enrichment and nutrient overloading of Norwegian finfish aquaculture at local and regional scales focus both on benthic and pelagic systems and is based on a combination of scientific knowledge and industry monitoring data, coupled with best professional judgment, and the precautionary principle.

Risk assessment

Release assessment

Intensive farming of finfish in open sea cages results in the release of organic and inorganic effluents (i.e. carbon, nitrogen, and phosphorus) in the form of waste feed, faeces, and metabolic by-products to the surrounding aquatic environments (Holmer *et al.*, 2005; Strain and Hargrave, 2005). At current production levels in Norway, salmonid farming (1.3 M tons cultivated fish in 2012) releases ~34 000 tons of N, 60 000 tons of C, and 9750 tons of P annually (Taranger *et al.*, 2013).

Expose assessment

Accumulation of these effluents into the marine system can negatively impact the ecosystem by contributing to eutrophication of pelagic systems, fertilization of benthic macrophytes in the euphotic zone, and organic enrichment of benthic systems (Strain and Hargrave, 2005). However, the area of influence (local or regional locations) and degree of enrichment of the environment depends on a number of factors including the size of the farm (i.e. the biomass of fish), the ambient environmental conditions (i.e. hydrodynamics, water depth, wave exposure, topography and substrate type) and the husbandry practices at the individual fish farms (Holmer *et al.*, 2005). In Norway, detailed knowledge about the environmental effects of organic and nutrient enrichment from finfish aquaculture is mainly based on studies around sheltered and coastal fjord aquaculture sites.

At deep aquaculture sites, fish farming effluents can be traced into the wider environment and into benthic foodwebs up to at least 1 km from the farming site (Kutti *et al.*, 2007b; Olsen *et al.*, 2012). At low deposition levels, organic enrichment of benthic sediments (up to 500 m from the farming location) stimulates secondary production in soft bottom communities, resulting in shifts in benthic faunal community structure (Kutti *et al.*, 2007a; Kutti, 2008; Bannister *et al.*, 2014). In addition, excessive loading of organic effluents to sediments often leads to dramatic changes in biogeochemical processes leading to grossly anoxic conditions (Valdemarsen *et al.*, 2012). The emissions of dissolved nutrients from finfish farms are quickly diluted in the water column at dynamic sites and elevated nutrient levels are hardly detected 200 m away from the farm (H. Jansen, IMR, unpublished data; Sanderson *et al.*, 2008).

Consequence assessment

Many studies have investigated benthic impacts of fish farming on soft sediment benthic systems, demonstrating that intensive fish farming modifies biogeochemical processes (Holmer and Kristensen, 1992; Holmer and Frederiksen, 2007; Norði *et al.*, 2011). Remineraliztion of the highly labile organic waste (i.e. fish feed and faeces) results in increased sediment oxygen demand and altered metabolic pathways, and a shift from aerobic (i.e. heterotrophic respiration) to anaerobic (i.e. sulphate reduction and methanogenesis) microbial degradation (Holmer and Kristensen, 1992; Holmer *et al.*, 2003; Valdemarsen *et al.*, 2009). Excessive organic enrichment can thus lead to highly modified sediment conditions (Valdemarsen *et al.*, 2012), impacting the structure and biomass of faunal communities (Kutti *et al.*, 2007b; Hargrave *et al.*, 2008; Valdemarsen *et al.*, 2010; Bannister *et al.*, 2014). Increased release

of dissolved nutrients from fish farming activities may stimulate phytoplankton growth and plankton blooms (Gowen and Ezzi, 1994) and may change the composition of seaweed communities in the littoral zone (Rueness and Fredriksen, 1991; Bokn *et al.*, 1992; Munda, 1996; Pihl *et al.*, 1999; Worm and Sommer, 2000; Krause-Jensen *et al.*, 2007).

To a lesser extent, there are studies that have investigate the effects of intensive fish farming to other habitats and biota including maerl beds (Hall-Spencer *et al.*, 2006; Sanz-Lazaro *et al.*, 2011; Aguado-Giménez and Ruiz-Fernández, 2012), coral reefs (Bongiorni *et al.*, 2003; Villanueva *et al.*, 2006), seaweeds and seagrass beds (Worm and Sommer, 2000; Diaz-Almela *et al.*, 2008; Holmer *et al.*, 2008), megafaunal communities (Wilding *et al.*, 2012), and pelagic and dermersal fish (Tuya *et al.*, 2006; Fernandez-Jover *et al.*, 2007, 2011; Dempster *et al.*, 2011). A consensus of these studies is that if the assimilative capacities of these environments are exceeded, then impacts on individual species, habitats, and ecosystems will be pronounced.

Risk estimation

Organic loading on a local scale

On a local scale the endpoint: *unacceptable change in faunal communities and sediment chemistry in the production zone* is estimated. The criterion of unacceptable change is determined by Norwegian authorities and all salmon farms in Norway are monitored through mandatory investigations (MOM system; Hansen *et al.*, 2001). The MOM-B investigations are performed regularly under and in the closest vicinity of the fish cages and are based on qualitatively determined indicators such as chemical parameters (pH and redox potential), sensory parameters, and presence and/or absence of macro-infauna. The performance of these indicators against predefined thresholds categorizes the farming locations into different environmental conditions (1. low-, 2. medium-, 3. high-organic loading, and 4. organic overloading). The environmental condition 4 represents an unacceptable state when production cannot continue before the farming location has recovered. Data from the monitoring of Norwegian salmon farms are obtained from the Norwegian Directorate of Fisheries, including 2761 electronically reported MOM-B investigations undertaken beneath Norwegian fish farms between 2009 and 2013. The percentage of farms in an unacceptable ecological state (4) has been stable and <3% the last few years (Figure 5). This is probably a result of better localization of farms. The risk of unacceptable change in sediment chemistry and fauna communities in the production zone is low. However, according to national set thresholds for management of the production zone, low impact on this scale does not reflect pristine conditions, but merely that the farm is managed within acceptable conditions in regard to its local impact.

Several rigorous scientific examinations of benthic impacts have been conducted near salmon farms in western Norway (Hordaland) where benthic carbon loadings, benthic fauna responses, and sediment biogeochemical processes were studied (Kutti *et al.*, 2007a, b; Kutti, 2008; Valdemarsen *et al.*, 2012; Bannister *et al.*, 2014). In addition, there are generic scientific knowledge in respect to the flow of organic waste into benthic foodwebs along the Norwegian coast (Olsen *et al.*, 2012).

Organic loading on a regional scale

The MOM-C system is an extended investigation of several sites (1–5) in the extended influence zone around farms and consists of quantitative measurements of the organic enrichment and the impact on biodiversity in infauna communities. The MOM-C investigations of fauna communities are following the Norwegian Standard (NS 9410) and farming sites are categorized into different environmental states (i.e. very good, good, moderate, poor, and very poor) according to nationally set thresholds (Molvær *et al.*, 1997). Based on the hydrographical conditions around the farm a distant point should be indentified in the most likely accumulation area beyond the production zone. The ecological condition at this distant site could be used as a proxy to estimate the risk of the endpoint; *significant change in bottom communities beyond the production zone (regional impact)*.

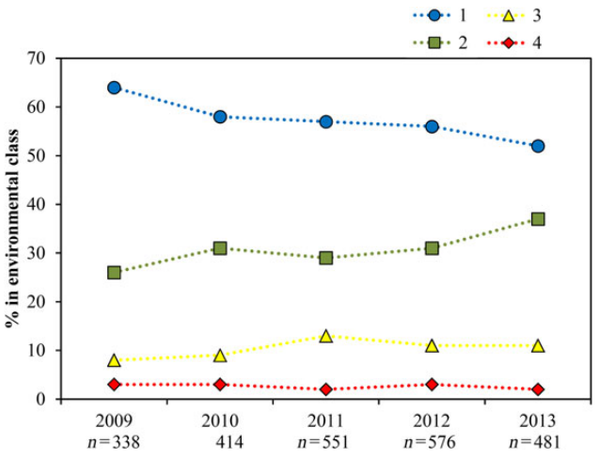

Figure 5. Impact of organic load from Norwegian marine finfish farms monitored by the mandatory MOM-B investigations (NS9410:2007) in the period 2009–2013. Data are given as percentage number of farms with ecological condition: 1 (blue), low organic loading; 2, (green), moderate organic loading; 3 (yellow), high organic loading (maximum allowed loading); 4 (red), overloading of the site, n = number of reported MOM-B investigations (data from Norwegian Directorate of Fisheries).

Data from MOM-C investigations on 122 salmon farms, which represents ~10% of the farms currently operating in Norwegian waters, are compiled from the Norwegian Directorate of Fisheries. The data show that distant sites at 95% of the farms had a high or very high ecological classification according to national set thresholds for Shannon–Wiener diversity index (H′) (Molvær *et al.*, 1997) while 5% was classified in moderate conditions.

To provide an estimate of risk for impact on a regional scale, data have been retained from case studies in the Hardangerfjord (Husa *et al.*, 2014a) and regional monitoring in some other areas in Norwegian coastal waters (Vassdal *et al.*, 2012) according to parameters and thresholds defined in the Norwegian implementation of the European Water Framework Directive (Molvær *et al.*, 1997; WFD, 2000/60/EC; Anon., 2009a). These data show that the ecological conditions in fauna communities and oxygen values in deep regional basins are high to very high in fjords with high salmon farming activity. These findings were also supported by analysis of the relative importance of the extra contribution of organic farm waste to decomposing communities in the deep basins in the Hardangerfjord, estimating that current farming production increased oxygen consumption by 10% and decreased oxygen levels in bottom water with 0.09% (Aure, 2013). However, we do not have sufficient data from the entire Norwegian coastline to make a full risk estimation of the impact of organic loading on a regional scale.

Nutrient emissions on a local scale

Local impact from nutrients and fine particulate material in the euphotic zone are currently not monitored around Norwegian fish farms, and we therefore have no data to estimate the endpoint; *nutrients from fish farms results in local eutrophication*.

Nutrient emissions on a regional scale

To estimate the endpoint, *nutrients from fish farms results in regional eutrophication*, we do not have sufficient data from Norwegian coastal waters to fulfil a complete risk estimation. However, three years monitoring of nutrient values and chlorophyll *a* in the Hardangerfjord area (Husa *et al.*, 2014b) and in Rogaland County, a sensitive area for fish farming due to lower water exchange (Vassdal *et al.*, 2012), show that ecological conditions for these parameters are within national acceptances thresholds (Molvær *et al.*, 1997) suggesting high or very high water quality.

These data coupled with modelling estimations on potential increase in phytoplankton production (Figure 6; Skogen *et al.*, 2009) suggest low risk of regional impacts from aquaculture in Norway. The potential increase in phytoplankton production is based on knowledge about the water transport mechanisms, coupled with typical natural values of nitrogen and phosphorous in the Norwegian Coastal Current and the calculated extra contribution to nutrient concentrations from fish farms in each Norwegian county. Assuming that theoretically all the nitrogen released from fish farms is assimilated in phytoplankton growth, an increase in the natural phytoplankton biomass were calculated and compared with the threshold of a 50% increase in phytoplankton biomass that is defined as eutrophication by OSPAR (Anon., 2010).

Limitations

The risk assessment in the period 2010–2013 is based on a limited dataset, restricted scientific knowledge, and national monitoring methods that require upgrading/revising. On a local scale, the use of the MOM-B monitoring dataset for assessing local impacts of organic enrichment should be used cautiously. The MOM-B system is built on a qualitative assessment limited in its efficacy outside of soft sediment habitats. Using the "MOM system" to monitor the environmental effects to other benthic habitats such as hard, mixed and sandy bottom habitats, seaweed and kelp habitats, or other sensitive habitats including sponge aggregations and cold water corals reefs will lead to uncertain monitoring results. Therefore, reported benthic conditions underneath fish farms should be used cautiously in the risk assessment approach. The number of MOM-C investigations available for this risk assessment was limited to 122 fish farming sites.

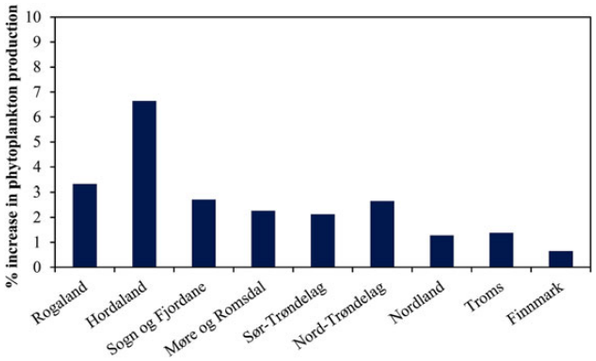

Figure 6. Estimated percentage increase in phytoplankton production due to emissions of dissolved nitrogen from finfish farms in 2012 in each Norwegian county, based on 100% exploitation of the nitrogen to carbon fixation.

Considering there are more or less 1000 fish farming locations along the Norwegian Coastline, the use of these investigations to provide an overview of the impact of organic enrichment from aquaculture within the influence area, should be approached cautiously, given they only represent ~10% of current production sites. Scientific data are restricted to fjord habitats in western Norway, there is a dearth of scientific studies representing the different geographical settings (i.e. southern, mid, and northern Norway) and the different benthic habitat types (coastal sandy sediments, hard bottom habitats, and sensitive and vulnerable habitats).

Furthermore, at regional scales national monitoring programmes have only started to begin (from 2008); therefore, existing knowledge and data of regional impacts from fish farming are mainly restricted to two counties and limited time. These limitations result in a precautionary approach used in the above risk assessment.

Future work

To achieve the goal of the Ministry of Fisheries and Coastal Affairs that the environmental impact of aquaculture must be kept at an acceptable level and be within the assimilative capacity of the area, further scientific research and coastal monitoring efforts are needed. It is crucial that further knowledge is developed on understanding the interaction of organic and nutrient waste release on different habitat types (ca. hard bottom habitats, coastal sandy sediment habitats, and benthic boreal systems). In addition, habitats of ecological significance and sensitive species (ca. coral reefs, sponge aggregations, maerl beds, seagrass meadows, and spawning areas) require detailed investigations to understand their responses to organic and nutrient loadings, thus allowing more informed decisions on ecological impacts to be made. Furthermore, given the heterogeneity of benthic substrates along the Norwegian coastline, and the existing limitations of the MOM-B monitoring standard, improved monitoring tools need to be established to enable monitoring of local impacts from fish farms on non-soft sediment substrates. MOM-C investigations should be performed more often and at a greater number of stations along a gradient from enrichment. Furthermore, sampling for therapeutants and fatty acids should be incorporated in the MOM-C investigations to detect the pressure of drugs on the environment and also to determine if the impacts detected are in fact related to fish farming activities.

Moreover, there is also a need to develop new modelling tools that can predict the dispersal of organic and nutrient wastes from fish farms, which will enable better placement of monitoring stations for both MOM-C and regional monitoring to increase the likelyhood of detecting impacts. Finally, regional monitoring programmes should incorporate greater sampling coverage (i.e. sampling locations) and frequency along the entire Norwegian coastline. Monitoring programmes should also identify possible risk areas for regional impacts and place further emphasis on monitoring in these habitats.

Discussion

The main approach in the risk assessment on the environmental impact of fish farming in Norway was to review the state-of-the-art on various hazards and potential risk factors, review national

monitoring data with relevance for the risk assessment, and score qualitatively the risk of impact in broad categories; low, moderate, and high risk of impact when monitoring data were available (Taranger *et al.*, 2011a, b, 2012b, 2013).

When the first risk assessment of the environmental impact of Norwegian fish farming was initiated in 2010 (Taranger *et al.*, 2011a), only very broad definitions and goals of aquaculture sustainability were presented in a strategy document by the Norwegian government (Anon., 2009b). The goals for sustainability put forward in this document are listed in Table 1, and covers the topics; diseases/parasites, genetic interactions with wild populations, pollution and discharges, marine site structure and zoning, as well as sustainability of feed and feed resources. These goal are very generic and do not specify sustainability indicators and related thresholds for societal/political acceptance of the environmental impact of fish farming in Norway.

The Institute of Marine Research in Norway and the Norwegian Veterinarian Institute were requested by the Norwegian Ministry of Fisheries and Coastal Affairs in 2011 to propose scientifically based sustainability indicators and related thresholds to score the severity of the potential environmental impacts of aquaculture. This included indicators for the risk of genetic introgression of escaped farmed salmon as well impact of salmon lice from fish farming on wild salmonid populations. As a result, indicators and suggested thresholds to score the environmental impact as low, moderate, or high were proposed in 2012 (Taranger *et al.*, 2012a). These recommendations were in part approved and implemented into a governmental report on Norwegian seafood policy in 2013 (Anon., 2013c), and in a newly established Quality Norm for the management and protection of wild Atlantic salmon in Norway approved by the Norwegian government in late 2013 (Anon., 2013a).

As a result, the suggested indicators for environmental impact as the associated suggested thresholds for scoring of the impacts have been adopted in the most recent version of the Risk assessment of Norwegian fish farming (Taranger *et al.*, 2014). They are the basis of the current assessment of risk of genetic introgression and impact on salmon lice on wild salmonid populations. The assessments are based on monitoring programmes on the numbers of escaped salmon in Norwegian rivers over the last 3 years, and the level of salmon lice infections on wild salmonids in Norwegian coastal waters during the last 4 years.

New scientific findings have recently emerged, such as the measured level of introgression of escaped farmed salmon into wild salmon populations in 20 Norwegian rivers (Glover *et al.*, 2013a), and new estimates on the additional marine mortality caused by salmon lice on Atlantic salmon from the rivers Dale and Vosso, in Hordaland, Norway (Skilbrei *et al.*, 2013; Vollset *et al.*, 2014). Such findings assist in developing more quantitative assessments of the impact of number of escaped salmon in rivers, and the impact of salmon lice from farming on wild salmonids, respectively.

In contrast to the situation for genetic introgression and impact of salmon lice, there are very limited monitoring data on potential transfer of other diseases and parasites from salmon farming to wild fish in Norway. Hence, the assessment on the risk of disease transfers to wild fish is mainly based on an analysis of the frequency of disease outbreaks in the ~1000 sites for salmon farming along the Norwegian coast. This is supported with a review on the knowledge of risk of disease transmission for the most relevant pathogens, and some available data on prevalence of pathogens and/or any disease outbreaks in wild Atlantic salmon and sea trout populations in Norway (Taranger *et al.*, 2014).

Regarding the local and regional impact of organic load and release of nutrient from marine salmon farming, such environmental impact indicators and associated monitoring programmes are defined and adapted by the Norwegian fish farming authorities regarding the local zone under and close to the farms, whereas the indicators and monitoring programmes for regional effects are being implemented in some counties in Norway (Vassdal *et al.*, 2012; Husa *et al.*, 2014a). The local zone under and close to the farm is monitored with a risk-based frequency using the relatively simple MOM-B method, while the more sensitive MOM-C method with detailed analysis of the species compositions in soft bottom samples near the farms is only applied occasionally (Hansen *et al.*, 2001). Both these methods have limitations, e.g. they require soft bottom, and are currently under revisions.

Data from regional monitoring has only become available in a few counties in the last years, but new programmes are starting up in several counties. The regional monitoring will to a large degree be based upon environmental indexes and environmental quality elements and related threshold for scoring of quality according to the Norwegian implementation of EUs Water Framework Directive (Anon., 2009a).

The current risk assessment of local of organic matter and nutrient release are based on the mandatory MOM-B monitoring from all farms and MOM-C analyses from a limited number of fish farms (Taranger *et al.*, 2014). The regional impact is evaluated based on available models and some investigations in the counties of Hordaland and Rogaland which both have high salmon farming activity compared with the area of available coastal water. As discussed above under the section on impact of organic matter and nutrients, the analyses have limitations, and a more extensive and improved monitoring programme is needed both on local and regional scale.

Moreover, a range of other potential risk factors, such as use of various pharmaceuticals, transfers of xenobiotics with the feed, use of copper as antifouling agent on sea cages, interactions with fisheries and other ecological impacts of sea cage farming, as well as ecological impacts of catch, transport and use of wrasses as cleaner fish against salmon lice are discussed in the risk assessment of Norwegian fish farming (Taranger *et al.*, 2014), but are not included in the current analysis.

Conclusions and summary of main findings

This represents the first risk assessment of cage-based salmonid aquaculture in Norway, which is world's largest producer of farmed Atlantic salmon. While there are several limitations in the approaches used to estimate the risks, as has been discussed in the sections above, this work has provided the Norwegian authorities with a framework upon which to evaluate the most important identified hazards against environmental goals for sustainability. The primary findings from the present risk assessment can be summarized as follows:

- Based upon the observed frequency of farmed escaped salmon on the spawning grounds of wild populations in the period 2010–2012, 21 of the 34 populations included in the risk assessment were in moderate–to-high risk of experiencing genetic changes due to introgression of farmed salmon. However, a recent study of 20 Norwegian rivers has demonstrated that there is only a moderate correlation between the observed frequency of escapees and introgression of farmed salmon (Glover *et al.*, 2013a); therefore, validation of the level of introgression in a

higher number of native populations will be required in the future.

- During the period 2010–2013, salmon lice infections mainly resulting from salmon farming were estimated at a total of 109 stations covering relevant areas of the Norwegian coastline using wild sea trout as a proxy for local infection pressure on wild salmonids. Twenty-seven of these stations indicated moderate or high likelihood of mortality for wild migrating salmon smolts. For sea trout later in the season, 67 of the stations indicated moderate or high likelihood of mortality on wild sea trout.
- The high frequency of the viral disease outbreaks for PD, IPN, heart and skeletal muscle inflammation, and CMS in Norwegian salmon farming suggests extensive release of the causal pathogens for these diseases in many areas. Migrating wild salmon and local sea trout are likely to be exposed to these pathogens. However, the extent of this exposure and consequences remains largely unknown. Screening of wild salmonids has revealed low to very low prevalence of the viruses SAV, IPNV, PMCV, and low prevalence of PRV in salmon. Furthermore, these viruses have never been documented to cause disease in wild Norwegian salmonids. Thus, a general lack of data prohibits complete risk estimation for these diseases.
- From a total of ~500 yearly investigations of local organic loading under fish farms, 2% of them displayed unacceptable conditions in the benthic sediments and faunal composition in 2013, whereas 11% classified with a high organic loading but still within the threshold. The remaining 87% of the farms had a moderate-to-high ecological conditions. The risk of eutrophication and organic over loading in the benthic communities beyond the production area of the farm is considered low based upon case studies and monitoring data from a limited area of the Norwegian coast.

Given the rapid expansion of open sea cage farming in Norway, and internationally, and the range of ecological impacts that either are demonstrated or suspected, there is an urgent need for better knowledge about such impacts, to implement improved monitoring programmes for the most important hazards, and also to improve procedures for risk assessments including useful environmental risk indicators and to facilitate processes that involves definitions on the societal acceptance levels of the various impacts.

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References

Aguado-Giménez F., and Ruiz-Fernández J. M. 2012. Influence of an experimental fish farm on the spatio-temporal dynamic of a Mediterranean maërl algae community. Marine Environmental Research, 74: 47–55.

Aldrin M., Storvik B., Frigessi A., Viljugrein H., and Jansen P. A. 2010. A stochastic model for the assessment of the transmission pathways of heart and skeleton muscle inflammation, pancreas disease and infectious salmon anaemia in marine fish farms in Norway. Preventive Veterinary Medicine, 93: 51–61.

Anon. 2006. Report of the Study Group on Risk Assessment and Management Advice (SGRAMA). ICES Resource Management Committee, ICES CM 2006/RMC: 04, Ref LRC ACFM, ACE ACME 71. ICES, Copenhagen.

Anon. 2009a. Classification of environmental condition of water. Directorate group for implementation of the water framework (in Norwegian). Veileder 01:2009. 180 pp.

Anon. 2009b. Strategy for an Environmentally Sustainable Norwegian Aquaculture Industry (in Norwegian). Ministry of Fisheries and Coastal Affairs, Oslo, Norway. 34 pp.

Anon. 2010. Report of the Working Group on Environmental Interactions of Mariculture (WGEIM). ICES CM 2010SSGHIE:08. 58 pp.

Anon. 2011. Status for the Norwegian salmon stocks in 2011 (in Norwegian). Report from Vitenskapelig råd for lakseforvaltning, nr. 3. 285 pp.

Anon. 2012. Salmon lice and effects on wild salmonids- from individual response to effects on stocks (in Norwegian). Thematic Report from Vitenskapelig råd for lakseforvaltning no. 3. 56 pp.

Anon. 2013a. Quality norm from wild populations of Atlantic salmon (*Salmo salar*) (in Norwegian). Ministry of Climate and Environment, Oslo, Norway. 26 pp.

Anon. 2013b. Status for the Norwegian salmon stocks in 2013 (in Norwegian). Report from Vitenskapelig råd for lakseforvaltning no 5. 136 pp.

Anon. 2013c. The worlds leading seafood nation (in Norwegian). White papers no. 22. Ministry of Fisheries and Coastal Affairs, Oslo, Norway. 139 pp.

Araki H., Berejikian B. A., Ford M. J., and Blouin M. S. 2008. Fitness of hatchery-reared salmonids in the wild. Evolutionary Applications, 1: 342–355.

Araki H., and Schmid C. 2010. Is hatchery stocking a help or harm? Evidence, limitations and future directions in ecological and genetic surveys. Aquaculture, 308: S2–S11.

Asplin L., Boxaspen K. K., and Sandvik A. D. 2011. Modeling the distribution and abundance of planktonic larval stages of *Lepeophtheirus salmonis* in Norway. *In* Salmon lice: an integrated approach to understanding parasite abundance and distribution. Ed. by S. Jones , and R. Beamish . Wiley-Blackwell, Oxford, UK.

Aure J. 2013. Farming and oxygen in the Hardangerfjord (in Norwegian). Havforskningsrapporten, Fisken og Havet Særnummer. pp. 42–44.

Austin B., and Austin D. A. 2007. Bacterial fish pathogens. Diseases of farmed and wild fish. Praxis Publishing Ltd., Chichester, UK.

Bakke T. A. 1991. A review of the inter- and intraspecific variability in salmonid hosts to laboratory infections with *Gyrodactylus salaris* Malmberg. Aquaculture, 98: 303–310.

Bakke T. A., and Harris P. D. 1998. Diseases and parasites in wild Atlantic salmon (*Salmo salar*) populations. Canadian Journal of Fisheries and Aquatic Sciences, 55: 247–266.

Bakke T. A., Harris P. D., Jansen P. A., and Hansen L. P. 1992. Host specificity and dispersal strategy in gyrodactylid monogeneans, with particular reference to *Gyrodactylus salaris* (Platyhelminthes, Monogenea). Diseases of Aquatic Organisms, 13: 63–74.

Bakke T. A., and MacKenzie K. 1993. Comparative susceptibility of native Scottish and Norwegian stocks of Atlantic salmon, *Salmo salar* L., to *Gyrodactylus salaris* Malmberg: Laboratory experiments. Fisheries Research, 17: 69–85.

Bannister R. J., Valdemarsen T., Hansen P. K., Holmer M., and Ervik A. 2014. Changes in benthic sediment conditions under an Atlantic salmon farm at a deep, well-flushed coastal site. Aquaculture Environment Interactions, 5: 29–47.

Berglund Andreassen K. 2013. Effects of infections with salmon lice (*Lepeophtheirus salmonis*) on wild smolts of salmon (*Salmo salar* L.) and trout (*Salmo trutta* L.) (in Norwegian). Master thesis, University of Tromsø, Norway. 61 pp.

Besnier F., Glover K. A., and Skaala Ø. 2011. Investigating genetic change in wild populations: modelling gene flow from farm escapees. Aquaculture Environment Interactions, 2: 75–86.

Biering E., Madhun A. S., Isachsen C. H., Omdal L. M., Einen A. C. B., Garseth A. H., Bjørn P. A., *et al.* 2013. Annual Report on health monitoring of wild anadromous salmonids in Norway. Report from the Norwegian Veterinary Institute and the Institute of Marine Research, Bergen, Norway.

Bjørn P. A., and Finstad B. 1997. The physiological effects of salmon lice infection on sea trout postsmolts. Nordic Journal of Freshwater Research, 73: 60–72.

Bjørn P. A., Finstad B., Asplin L., Skilbrei O., Nilsen R., Serra Llinares R. M., and Boxaspen K. K. 2011a. Method development for surveillance and counting of salmon lice on wild salmonids. Rapport fra Havforskningen 8-2011. 58 pp.

Bjørn P. A., Finstad B., Kristoffersen R., McKinley R. S., and Rikardsen A. H. 2007a. Differences in risks and consequences of salmon louse, *Lepeophtheirus salmonis* (Krøyer), infestation on sympatric populations of Atlantic salmon, brown trout, and Arctic charr within northern fjords. ICES Journal of Marine Science, 64: 386–393.

Bjørn P. A., Finstad B., Nilsen R., Asplin L., Uglem I., Skaala Ø., and Boxaspen K. K., *et al.* 2007b. Norwegian national surveillance of salmon lice epidemics on wild Artlantic salmon, sea trout and Arctic charr in connection with Norwegian national salmon rivers and fjords. NINA report 377. 33 pp.

Bjørn P. A., Finstad B., Nilsen R., Uglem I., Asplin L., Skaala Ø., and Hvidsten N. A. 2010. Norwegian national surveillance 2009 of salmon lice epidemics on wild Atlantic salmon, sea trout and Arctic charr in connection with Norwegian national salmon rivers and fjords . NINA Report 547. 50 pp.

Bjørn P. A., Nilsen R., Serra Llinares R. M., Asplin L., Askeland Johnsen I., Karlsen Ø., Finstad B., *et al.* 2013. Salmon lice infections on wild salmonids along the Norwegian coast in 2013. Final report to Norwegian Food Safety Authority. Report from Havforskningsinstituttet, no. 32–2013.

Bjørn P. A., Nilsen R., Serra Llinares R. M., Asplin L., Boxaspen K. K., Finstad B., Uglem I., *et al.* 2012. Salmon lice infections on wild salmonids along the Norwegian coast in 2012. Final report to Norwegian Food Safety Authority. Report from Havforskningen, 31–2012.

Bjørn P. A., Nilsen R., Serra Llinares R. M., Asplin L., Boxaspen K. K., Finstad B., Uglem I., *et al.* 2011b. Final report to the Norwegian Food Safety Authority of the salmon lice infections along the Norwegian coast in 2011. Report From Havforskningen, 19-2011: 34.

Bjørn P. A., Sivertsgård R., Finstad B., Nilsen R., Serra-Llinares R. M., and Kristoffersen R. 2011c. Area protection may reduce salmon louse infection risk to wild salmonids. Aquaculture Environment Interactions, 1: 233–244.

Bokn T., Murray S. N., Moy F. E., and Magnusson J. B. 1992. Changes in fucoid distribution and abundances in the inner Oslofjord, Norway: 1974–80 versus 1988–90. Acta Phytogeographica Suecia, 78: 117–124.

Bongiorni L., Shafir S., and Rinkevich B. 2003. Effects of particulate matter released by a fish farm (Eilat, Red Sea) on survival and growth of *Stylophora pistillata* coral nubbins. Marine Pollution Bulletin, 46: 1120–1124.

Boucher P., Raynard R. S., Houghton G., and Laurencin F. B. 1995. Comparative experimental transmission of pancreas disease in Atlantic salmon, rainbow trout and brown trout. Diseases of Aquatic Organisms, 22: 19–24.

Bourret V., O'Reilly P. T., Carr J. W., Berg P. R., and Bernatchez L. 2011. Temporal change in genetic integrity suggests loss of local adaptation in a wild Atlantic salmon (*Salmo salar*) population following introgression by farmed escapees. Heredity, 106: 500–510.

Brun E. 2003. Chapter 4: Epidemiology. *In* IPN in salmonids – a review, pp. 51–67. Ed. by B. Skjelstad , E. Brun , and I. Jensen . VESO, Oslo, Norway.

Brun E., and Lillehaug A. 2010. Risk profile of the diseases in Norwegian aquaculture. Report 9–2010 from National Veterinary Institute, Oslo, Norway.

Buschmann A. H., Riquelme V. A., Hernández-González M. C., Varela D., Jiménez J. E., Henríquez L. A., Vergara P. A., *et al.* 2006. A review of the impacts of salmonid farming on marine coastal ecosystems in the southeast Pacific. ICES Journal of Marine Science, 63: 1338–1345.

Clifford S. L., McGinnity P., and Ferguson A. 1998a. Genetic changes in an Atlantic salmon population resulting from escaped juvenile farm salmon. Journal of Fish Biology, 52: 118–127.

Clifford S. L., McGinnity P., and Ferguson A. 1998b. Genetic changes in Atlantic salmon (*Salmo salar*) populations of northwest Irish rivers resulting from escapes of adult farm salmon. Canadian Journal of Fisheries and Aquatic Sciences, 55: 358–363.

Combes C. 2001. Parasitism: The ecology and evolution of intimate interactions. University of Chicago Press, Chicago, USA.

Crozier W. W. 1993. Evidence of genetic interaction between escaped farmed salmon and wild Atlantic salmon (*Salmo salar* L.) in a Northern Irish river. Aquaculture, 113: 19–29.

Crozier W. W. 2000. Escaped farmed salmon, *Salmo salar* L., in the Glenarm River, Northern Ireland: genetic status of the wild population 7 years on. Fisheries Management and Ecology, 7: 437–446.

Dempster T., Sanchez-Jerez P., Fernandez-Jover D., Bayle-Sempere J., Nilsen R., Bjørn P. A., and Uglem I. 2011. Proxy measures of fitness suggest coastal fish farms can act as population sources and not ecological traps for wild gadoid fish. PLoS ONE, 6: e15646, 9 pp.

Diaz-Almela E., Marba N., Alvarez E., Santiago R., Holmer M., Grau A., Mirto S., *et al.* 2008. Benthic input rates predict seagrass (*Posidonia oceanica*) fish farm-induced decline. Marine Pollution Bulletin, 56: 1332–1342.

Dillane E., McGinnity P., Coughlan J. P., Cross M. C., de Eyto E., Kenchington E., and Prodohl P., *et al.* 2008. Demographics and landscape features determine intrariver population structure in Atlantic salmon (*Salmo salar* L.): the case of the River Moy in Ireland. Molecular Ecology, 17: 4786–4800.

Diserud O. H., Fiske P., and Hindar K. 2010. Regional impact of escaped farm salmon on wild salmon populations in Norway. NINA Report 622. 40 pp.

Dobson A. P., and Hudson P. J. 1986. Parasites, disease and the structure of ecological communities. Trends in Ecology & Evolution, 1: 11–15.

EFSA. 2012. EFSA Panel on Animal Health and Welfare: guidance on risk assessment for animal welfare. EFSA Journal, 10: 2513, 30 pp. doi:10.2903/j.efsa.2012.2513.

Einum S., and Fleming I. A. 1997. Genetic divergence and interactions in the wild among native, farmed and hybrid Atlantic salmon. Journal of Fish Biology, 50: 634–651.

Fernandez-Jover D., Jimenez J. A. L., Sanchez-Jerez P., Bayle-Sempere J., Casalduero F. G., Lopez F. J. M., and Dempster T. 2007. Changes in body condition and fatty acid composition of wild Mediterranean horse mackerel (*Trachurus mediterraneus*, Steindachner, 1868) associated to sea cage fish farms. Marine Environmental Research, 63: 1–18.

Fernandez-Jover D., Martinez-Rubio L., Sanchez-Jerez P., Bayle-Sempere J. T., Jimenez J. A. L., Lopez F. J. M., Bjørn P. A., *et al.* 2011. Waste feed from coastal fish farms: a trophic subsidy with compositional side-effects for wild gadoids. Estuarine Coastal and Shelf Science, 91: 559–568.

Finstad B., and Bjørn P. A. 2011. Present status and implications of salmon lice on wild salmonids in Norwegian coastal zones. *In* Salmon lice: an integrated approach to understanding parasite

abundance and distribution, pp. 281–305. Ed. by S. Jones , and R. Beamish . Wiley-Blackwell, Oxford, UK.

Finstad B., Bjørn P. A., Grimnes A., and Hvidsten N. A. 2000. Laboratory and field investigations of salmon lice [*Lepeophtheirus salmonis* (Krøyer)] infestation on Atlantic salmon (*Salmo salar* L.) post-smolts. Aquaculture Research, 31: 795–803.

Finstad B., Bjørn P. A., Todd C. D., Whoriskey F., Gargan P. G., Førde G., and Revie C. 2010. The effect of sea lice on Atlantic salmon and other salmonid species. *In* Atlantic salmon ecology, pp. 253–276. Ed. by Ø. Aas , S. Einum , A. Klemetsen , and J. Skurdal . Wiley-Blackwell, Oxford, UK.

Finstad B., and Jonsson N. 2001. Factors influencing the yield of smolt releases in Norway. Nordic Journal of Freshwater Research, 75: 37–55.

Fiske P. 2013. Surveillance of escaped farmed salmon in rivers in the autumn 2010–2012. NINA Report 989. 33 pp.

Fiske P., Lund R. A., and Hansen L. P. 2006. Relationships between the frequency of farmed Atlantic salmon, *Salmo salar* L., in wild salmon populations and fish farming activity in Norway, 1989–2004. ICES Journal of Marine Science, 63: 1182–1189.

Fleming I. A., Hindar K., Mjølnerød I. B., Jonsson B., Balstad T., and Lamberg A. 2000. Lifetime success and interactions of farm salmon invading a native population. Proceedings of the Royal Society of London Series B Biological Sciences, 267: 1517–1523.

Fleming I. A., Jonsson B., Gross M. R., and Lamberg A. 1996. An experimental study of the reproductive behaviour and success of farmed and wild Atlantic salmon (*Salmo salar*). Journal of Applied Ecology, 33: 893–905.

Fraser D. J., Weir L. K., Bernatchez L., Hansen M. M., and Taylor E. B. 2011. Extent and scale of local adaptation in salmonid fishes: review and meta-analysis. Heredity, 106: 404–420.

Garcia de Leaniz C., Fleming I. A., Einum S., Verspoor E., Jordan W. C., Consuegra S., Aubin-Horth N., *et al.* 2007. A critical review of adaptive genetic variation in Atlantic salmon: implications for conservation. Biological Reviews, 82: 173–211.

Gargan P. G., Forde G., Hazon N., Russell D. J. F., and Todd C. D. 2012. Evidence for sea lice-induced marine mortality of Atlantic salmon (*Salmo salar*) in western Ireland from experimental releases of ranched smolts treated with emamectin benzoate. Canadian Journal of Fisheries and Aquatic Sciences, 69: 343–353.

Garseth A. H., Biering E., and Aunsmo A. 2013a. Associations between piscine reovirus infection history traits in wild-caught Atlantic salmon *Salmo salar* L. in and life Norway. Preventive Veterinary Medicine, 112: 138–146.

Garseth A. H., Biering E., and Tengs T. 2012. Piscine myocarditis virus (PMCV) in wild Atlantic salmon *Salmo salar*. Diseases of Aquatic Organisms, 102: 157–161.

Garseth Å. H., Ekrem T., and Biering E. 2013c. Phylogenetic evidence of long distance dispersal and transmission of piscine reovirus (PRV) between farmed and wild Atlantic salmon. PLoS ONE, 8: e82202.

Garseth A. H., Fritsvold C., Opheim M., Skjerve E., and Biering E. 2013b. Piscine reovirus (PRV) in wild Atlantic salmon, *Salmo salar* L., and sea-trout, *Salmo trutta* L., in Norway. Journal of Fish Diseases, 36: 483–493.

Gaughan D. J., Mitchell R. W., and Blight S. J. 2000. Impact of mortality, possibly due to herpesvirus, on pilchard *Sardinops sagax* stocks along the south coast of Western Australia in 1998–99. Marine and Freshwater Research, 51: 601–612.

GESAMP. 2008. Assessment and communication of environmental risks in coastal aquaculture. IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP Joint Group of Experts on Scientific Aspects of Marine Environmental Protection. Reports and Studies GESAMP No. 76. FAO, Rome. 198 pp.

Gjedrem T. 2010. The first family-based breeding program in aquaculture. Reviews in Aquaculture, 2: 2–15.

Glover K. A. 2010. Forensic identification of fish farm escapees: the Norwegian experience. Aquaculture Environment Interactions, 1: 1–10.

Glover K. A., Ottera H., Olsen R. E., Slinde E., Taranger G. L., and Skaala Ø. 2009. A comparison of farmed, wild and hybrid Atlantic salmon (*Salmo salar* L.) reared under farming conditions. Aquaculture, 286: 203–210.

Glover K. A., Pertoldi C., Besnier F., Wennevik V., Kent M., and Skaala Ø. 2013a. Atlantic salmon populations invaded by farmed escapees: quantifying genetic introgression with a Bayesian approach and SNPs. BMC Genetics, 14: 4.

Glover K. A., Quintela M., Wennevik V., Besnier F., Sørvik A. G. E., and Skaala Ø. 2012. Three decades of farmed escapees in the wild: A spatio-temporal analysis of population genetic structure throughout Norway. PLoS ONE, 7: e43129.

Glover K. A., Skilbrei O., and Skaala Ø. 2008. Genetic assignment identifies farm of origin for Atlantic salmon *Salmo salar* escapees in a Norwegian fjord. ICES Journal of Marine Science, 65: 912–920.

Glover K. A., Sørvik A. G. E., Karlsbakk E., Zhang Z., and Skaala Ø. 2013b. Molecular genetic analysis of stomach contents reveals wild Atlantic cod feeding on piscine reovirus (PRV) infected Atlantic salmon originating from a commercial fish farm. PLoS ONE, 8: e60924.

Gowen R. J., and Ezzi I. A. 1994. Assessment and prediction of the potential for hypernutrification and eutrophication associated with cageculture of salmonids in Scottish waters. Dunstaffnage Marine Laboratory, Oban, Scotland. 137 pp.

Graham D. A., Staples C., Wilson C. J., Jewhurst H., Cherry K., Gordon A., and Rowley H. M. 2007. Biophysical properties of salmonid alphaviruses: influence of temperature and pH on virus survival. Journal of Fish Diseases, 30: 533–543.

Grimnes A., and Jakobsen P. J. 1996. The physiological effects of salmon lice infection on post-smolt of Atlantic salmon (*Salmo salar*). Journal of Fish Biology, 48: 1179–1194.

Gullestad P., Bjørgo S., Eithun I., Ervik A., Gudding R., Hansen H., Røsvik I. O., *et al.* 2011. Effective and sustainable use of areas for aquaculture – land of desire (in Norwegian). Report from an expert panel appointed by Ministry of Fisheries and Coastal Affairs, Oslo, Norway. 198 pp.

Hall-Spencer J., White N., Gillespie E., Gillham K., and Foggo A. 2006. Impact of fish farms on maerl beds in strongly tidal areas. Marine Ecology Progress Series, 326: 1–9.

Hansen H., Bachmann L., and Bakke T. A. 2003. Mitochondrial DNA variation of *Gyrodactylus* spp. (Monogenea, Gyrodactylidae) populations infecting Atlantic salmon, grayling, and rainbow trout in Norway and Sweden. International Journal for Parasitology, 33: 1471–1478.

Hansen L. P. 2006. Migration and survival of farmed Atlantic salmon (*Salmo salar* L.) released from two Norwegian fish farms. ICES Journal of Marine Science, 63: 1211–1217.

Hansen P. K., Ervik A., Schanning M. T., Johannsen P., Aure J., Jahnsen T., and Stigebrandt A. 2001. Regulating the local environmental impact of intensive marine fish farming. II. The monitoring programme of the MOM system (Modelling – Ongrowing fish farms – Monitoring). Aquaculture, 194: 75–92.

Hargrave B. T., Holmer M., and Newcombe C. P. 2008. Towards a classification of organic enrichment in marine sediments based on biogeochemical indicators. Marine Pollution Bulletin, 56: 810–824.

Haugland O., Mikalsen A. B., Nilsen P., Lindmo K., Thu B. J., Eliassen T. M., Roos N., *et al.* 2011. Cardiomyopathy syndrome of Atlantic salmon (*Salmo salar* L.) is caused by a double-stranded RNA virus of the totiviridae family. Journal of Virology, 85: 5275–5286.

Hazon N., Todd C. D., Whelan B., Gargan P. G., Finstad B., Bjørn P. A., and Wendelar Bonga S. E., *et al.* 2006. Sustainable management of interactions between aquaculture and wild salmonid fish. Final report for the SUMBAWS EU project. pp. 1–293.

Helland I. P., Finstad B., Uglem I., Diserud O. H., Foldvik A., Hanssen F., Bjørn P. A., *et al.* 2012. What determines salmon lice infections on wild salmonids? Statistical calculations of data from the national salmon lice surveillance program 2004–2010 (in Norwegian). NINA Rapport, 891. 51 pp.

Heuch P. A., Bjørn P. A., Finstad B., Holst J. C., Asplin L., and Nilsen F. 2005. A review of the Norwegian "National Action Plan Against Salmon Lice on Salmonids": The effect on wild salmonids. Aquaculture, 246: 79–92.

Hindar K., Fleming I. A., McGinnity P., and Diserud A. 2006. Genetic and ecological effects of salmon farming on wild salmon: modelling from experimental results. ICES Journal of Marine Science, 63: 1234–1247.

Hjeltnes B. (Ed.) 2014. Report on fish health—2013 (in Norwegian). Veterinærinstituttet, Oslo, Norway. 44 pp.

Holm M., Holst J. C., and Hansen L. P. 2000. Spatial and temporal distribution of post-smolts of Atlantic salmon (*Salmo salar* L.) in the Norwegian Sea and adjacent areas. ICES Journal of Marine Science, 57: 955–964.

Holmer M., Argyrou M., Dalsgaard T., Danovaro R., Diaz-Almela E., Carlos M. D. E., Frederiksen M., *et al.* 2008. Effects of fish farm waste on *Posidonia oceanica* meadows: Synthesis and provision of monitoring and management tools. Marine Pollution Bulletin, 56: 1618–1629.

Holmer M., Duarte C. M., Heilskov A., Olesen B., and Terrados J. 2003. Biogeochemical conditions in sediments enriched by organic matter from net-pen fish farms in the Bolinao area, Philippines. Marine Pollution Bulletin, 46: 1470–1479.

Holmer M., and Frederiksen M. S. 2007. Stimulation of sulfate reduction rates in Mediterranean fish farm sediments inhabited by the seagrass *Posidonia oceanica*. Biogeochemistry, 85: 169–184.

Holmer M., and Kristensen E. 1992. Impact of marine fish cage farming on metabolism and sulfate reduction of underlying sediments. Marine Ecology Progress Series, 80: 191–201.

Holmer M., Wildish D., and Hargrave B. 2005. Environmental effects of marine finfish aquaculture. *In* Handbook of environmental chemistry, pp. 181–206. Ed. by B. Hargrave. Springer-Verlag, Berlin.

Holst J. C., Jakobsen P. J., Nilsen F., Holm M., Asplin L., and Aure J. 2007. Mortality of seaward-migrating post-smolts of Atlantic salmon due to salmon lice infection in Norwegian salmon stocks. *In* Salmon at the edge, pp. 136–137. Blackwell Science Ltd, Bodmin, Cornwall.

Husa V., Kutti T., Ervik A., Sjøtun K., Hansen P. K., and Aure J. 2014a. Regional impact from fin-fish farming in an intensive production area (Hardangerfjord, Norway). Marine Biology Research, 10: 241–252.

Husa V., Steen H., and Sjøtun K. 2014b. Historical changes in macroalgal communities in Hardangerfjord (Norway). Marine Biology Research, 10: 226–240.

Hvidsten N. A., Finstad B., Kroglund F., Johnsen B. O., Strand R., Arnekleiv J. V., and Bjørn P. A. 2007. Does increased abundance of sea lice influence survival of wild Atlantic salmon post-smolt? Journal of Fish Biology, 71: 1639–1648.

Hyatt A. D., Hine P. M., Jones J. B., Whittington R. J., Kearns C., Wise T. G., and Crane M. S., *et al.* 1997. Epizootic mortality in the pilchard *Sardinops sagax neopilchardus* in Australia and New Zealand in 1995. II. Identification of a herpesvirus within the gill epithelium. Diseases of Aquatic Organisms, 28: 17–29.

Hytterød S., Lie Linaker M., Hansen H., Mo T. A., and Tavornpanich S. 2014. The surveillance programme for *Gyrodactylus salaris* in Atlantic salmon and rainbow trout in Norway. Surveillance programmes for terrestrial and aquatic animals in Norway. Annual report 2013. Norwegian Veterinary Institute, Oslo. 6 pp.

Jackson D., Cotter D., Newell J., McEvoy S., O'Donohoe P., Kane F., McDermott T., *et al.* 2013. Impact of *Lepeophtheirus salmonis* infestations on migrating Atlantic salmon, *Salmo salar* L., smolts at eight locations in Ireland with an analysis of lice-induced marine mortality. Journal of Fish Diseases, 36: 273–281.

Jackson D., Cotter D., ÓMaoiléidigh N., O'Donohoe P., White J., Kane F., Kelly S., *et al.* 2011. An evaluation of the impact of early infestation with the salmon louse *Lepeophtheirus salmonis* on the subsequent survival of outwardly migrating Atlantic salmon, *Salmo salar* L., smolts. Aquaculture, 320: 159–163.

Jansen P. A., Kristoffersen A. B., Viljugrein H., Jimenez D., Aldrin M., and Stien A. 2012. Sea lice as a density-dependent constraint to salmonid farming. Proceedings of the Royal Society B: Biological Sciences, 279: 2330–2338.

Johansen L. H., Jensen I., Mikkelsen H., Bjørn P. A., Jansen P. A., and Bergh Ø. 2011. Disease interaction and pathogens exchange between wild and farmed fish populations with special reference to Norway. Aquaculture, 315: 167–186.

Johansen R. (Ed.) 2013. Report on fish health—2012 (in Norwegian). Veterinærinstituttet, Oslo, Norway. 44 pp.

Johnsen B. O., and Jensen A. J. 1994. The spread of furunculosis in salmonids in Norwegian rivers. Journal of Fish Biology, 45: 47–55.

Johnsen B. O., Møkkelgjerd P. I., and Jensen A. J. 1999. The parasite *Gyrodactylus salaris* on salmon in Norwegian rivers, status report at start of the year 2000 (in Norwegian). NINA Oppdragsmelding, pp. 1–129.

Jones S., and Beamish R. 2011. Salmon lice: an integrated approach to understanding parasite abundance and distribution. Wiley-Blackwell, Oxford, UK.

Jonsson B., and Jonsson N. 2006. Cultured Atlantic salmon in nature: a review of their ecology and interaction with wild fish. ICES Journal of Marine Science, 63: 1162–1181.

Karlsson S., Moen T., Lien S., Glover K. A., and Hindar K. 2011. Generic genetic differences between farmed and wild Atlantic salmon identified from a 7 K SNP-chip. Molecular Ecology Resources, 11: 247–253.

Kileng Ø., Johansen L-H., and Jensen I. 2011. ILA-virus in wild and farmed fish in Troms (in Norwegian). Nofima report 28/2011, Tromsø, Norway.

Kongtorp R. T., Kjerstad A., Taksdal T., Guttvik A., and Falk K. 2004. Heart and skeletal muscle inflammation in Atlantic salmon, *Salmo salar* L.: a new infectious disease. Journal of Fish Diseases, 27: 351–358.

Krause-Jensen D., Middleboe A. L., Carstensen J., and Dahl K. 2007. Spatial patterns of macroalgal abundance in relation to eutrophication. Marine Biology, 152: 25–36.

Kristoffersen A. B., Bang Jensen B., and Jansen P. A. 2013. Risk mapping of heart and skeletal muscle inflammation in salmon farming. Preventive Veterinary Medicine, 109: 136–143.

Kristoffersen A. B., Viljugrein H., Kongtorp R. T., Brun E., and Jansen P. A. 2009. Risk factors for pancreas disease (PD) outbreaks in farmed Atlantic salmon and rainbow trout in Norway during 2003–2007. Preventive Veterinary Medicine, 90: 127–136.

Krkošek M., Lewis M. A., and Volpe J. P. 2005. Transmission dynamics of parasitic sea lice from farm to wild salmon. Proceedings of the Royal Society B-Biological Sciences, 272: 689–696.

Krkošek M., Revie C., Gargan P. G., Finstad B., and Todd C. D. 2013a. Comment on Jackson et al. "Impact of *Lepeophtheirus salmonis* infestations on migrating Atlantic salmon, *Salmo salar* L., smolts at eight locations in Ireland with an analysis of lice-induced marine mortality". Journal of Fish Diseases, 37: 415–417.

Krkošek M., Revie C. W., Gargan P. G., Skilbrei O. T., Finstad B., and Todd C. D. 2013b. Impact of parasites on salmon recruitment in the Northeast Atlantic Ocean. Proceedings of the Royal Society B: Biological Sciences, 280: 20122359.

Kutti T. 2008. Regional impact of organic loading from a salomonid farm – dispersal, sedimentation rates and benthic fauna response. PhD thesis, University of Bergen.

Kutti T., Ervik A., and Hansen P. K. 2007a. Effects of organic effluents from a salmon farm on a fjord system. I. Vertical export and dispersal processes. Aquaculture, 262: 367–381.

Kutti T., Ervik A., and Høisæther T. 2008. Effects of organic effluents from a salmon farm on a fjord system. III. Linking deposition rates of organic matter and benthic productivity. Aquaculture, 282: 47–53.

Kutti T., Hansen P. K., Ervik A., Høisæther T., and Johannessen P. J. 2007b. Effects of organic effluents from a salmon farm on a fjord system. II. Temporal and spatial patterns in infauna community composition. Aquaculture, 262: 355–366.

Løvoll M., Alarcon M., Jensen B. B., Taksdal T., Kristoffersen A. B., and Tengs T. 2012. Quantification of piscine reovirus (PRV) at different stages of Atlantic salmon *Salmo salar* production. Diseases of Aquatic Organisms, 99: 7–12.

Løvoll M., Wiik-Nielsen J., Grove S., Wiik-Nielsen C. R., Kristoffersen A. B., Faller R., Poppe T. *et al.* 2010. A novel totivirus and piscine reovirus (PRV) in Atlantic salmon (*Salmo salar*) with cardiomyopathy syndrome (CMS). Virology Journal, 7: 309.

Madhun A. S., Biering E., Isachsen C. H., Omdal L. M., Einen A. C. B., Wennevik V., Svåsand T., *et al.* 2014a. Annual report on health monitoring of wild anadromous salmonids in Norway. Report from the Norwegian Veterinary Institute and the Institute of Marine Research, Bergen, Norway.

Madhun A. S., Karlsbakk E., Isachsen C. H., Omdal L. M., Eide Sørvik A. G., Skaala Ø., and Barlaup B. T., *et al.* 2014b. Potential disease interaction reinforced: double-virus-infected escaped farmed Atlantic salmon, *Salmo salar* L., recaptured in a nearby river. Journal of Fish Diseases: in press, DOI: 10.1111/jfd.12228.

Martinez-Rubio L., Morais S., Evensen O., Wadsworth S., Ruohonen K., Vecino J. L. G., and Bell J. G., *et al.* 2012. Functional feeds reduce heart inflammation and pathology in Atlantic salmon (*Salmo salar* L.) following experimental challenge with Atlantic salmon reovirus (ASRV). PLoS ONE, 7: e40266.

Martinez-Rubio L., Morais S., Evensen O., Wadsworth S., Vecino J. G., Ruohonen K., and Bell J. G., *et al.* 2013. Effect of functional feeds on fatty acid and eicosanoid metabolism in liver and head kidney of Atlantic salmon (*Salmo salar* L.) with experimentally induced Heart and Skeletal Muscle Inflammation. Fish & Shellfish Immunology, 34: 1533–1545.

May R. M. 1983. Parasitic infections as regulators of animal interactions. American Scientist, 71: 36–45.

May R. M., and Anderson R. M. 1979. Population biology of infectious diseases: Part II. Nature, 280: 455–461.

Mcallister P. E., Newman M. W., Sauber J. H., and Owens W. J. 1984. Isolation of infectious pancreatic necrosis virus (serotype-Ab) from diverse species of estuarine fish. Helgolander Meeresuntersuchungen, 37: 317–328.

McGinnity P., Prodohl P., Ferguson K., Hynes R., O'Maoileidigh N., Baker N., Cotter D., *et al.* 2003. Fitness reduction and potential extinction of wild populations of Atlantic salmon, *Salmo salar*, as a result of interactions with escaped farm salmon. Proceedings of the Royal Society of London Series B Biological Sciences, 270: 2443–2450.

McGinnity P., Stone C., Taggart J. B., Cooke D., Cotter D., Hynes R., McCamley C., *et al.* 1997. Genetic impact of escaped farmed Atlantic salmon (*Salmo salar* L.) on native populations: use of DNA profiling to assess freshwater performance of wild, farmed, and hybrid progeny in a natural river environment. ICES Journal of Marine Science, 54: 998–1008.

McLoughlin M. F., Nelson R. T., Rowley H. M., Cox D. I., and Grant A. N. 1996. Experimental pancreas disease in Atlantic salmon *Salmo salar* post-smolts induced by salmon pancreas disease virus (SPDV). Diseases of Aquatic Organisms, 26: 117–124.

Mente E., Pierce G. J., Santos M. B., and Neofitou C. 2006. Effect of feed and feeding in the culture of salmonids on the marine aquatic environment: a synthesis for European aquaculture. Aquaculture International, 14: 499–522.

Molvær J., Knutzen J., Magnusson J. B., Rugg B., Skei J., and Sørensen J. 1997. Classification of environmental quality in fjords and coastal waters (in Norwegian). Norwegian Environment Agency, 3. 35 pp.

Mork J. 1991. One-generation effects of farmed fish immigration on the genetic differentiation of wild Atlantic salmon in Norway. Aquaculture, 98: 267–276.

Mørk T., and Hellberg H. 2005. The surveillance and control programme for *Gyrodactylus salaris* in Atlantic salmon and rainbow trout in Norway. *In* Surveillance and control programmes for terrestrial and aquatic animals in Norway, pp. 137–139. Annual report 2004.

Mortensen S. H. 1993. Passage of infectious pancreatic necrosis virus (IPNV) through invertebrates in an aquatic food-chain. Diseases of Aquatic Organisms, 16: 41–45.

Munda I. M. 1996. The northern Adriatic Sea. *In* Marine benthic vegetation, pp. 369–402. Ed. by W. Schramm , and P. Nienhuis . Springer, Berlin, Heidelberg.

Nash C. 2007. Guidelines for ecological risk assessment of marine aquaculture. Aquaculture Europe, 32: 5–11.

Nelson R. T., Mcloughlin M. F., Rowley H. M., Platten M. A., and Mccormick J. I. 1995. Isolation of a toga-like virus from farmed Atlantic salmon *Salmo salar* with pancreas disease. Diseases of Aquatic Organisms, 22: 25–32.

Nolan D. T., Reilly P., and Wendelar Bonga S. E. 1999. Infection with low numbers of the sea louse *Lepeophtheirus salmonis* induces stress-related effects in postsmolt Atlantic salmon (*Salmo salar*). Canadian Journal of Fisheries and Aquatic Sciences, 56: 947–959.

Norði G. A., Glud R. N., Gaard E., and Simonsen K. 2011. Environmental impacts of coastal fish farming: carbon and nitrogen budgets for trout farming in Kaldbaksfjorour (Faroe Islands). Marine Ecology Progress Series, 431: 223–241.

Norris A. T., Bradley D. G., and Cunningham E. P. 1999. Microsatellite genetic variation between and within farmed and wild Atlantic salmon (*Salmo salar*) populations. Aquaculture, 180: 247–264.

Nylund A. 2007. Salmonid Alphavirus (SAV). *In* Review of disease interactions and pathogen exchange between farmed and wild finfish and shellfish in Europe, pp. 41–45. Ed. by R. Raynard , T. Wahli , I. Vatsos , and S. Mortensen . Veterinærmedisinsk Oppdragssenter AS.

Olsen S. A., Ervik A., and Grahl-Nielsen O. 2012. Tracing fish farm waste in the northern shrimp *Pandalus borealis* (Krøyer, 1838) using lipid biomarkers. Aquaculture Environment Interactions, 2: 133–144.

OSPAR. 2010. OSPAR commission 2010. Quality status report 2010. http://qsr2010.ospar.org/en/index.html.

Palacios G., Lovoll M., Tengs T., Hornig M., Hutchison S., Hui J., Kongtorp R. T., *et al.* 2010. Heart and skeletal muscle inflammation of farmed salmon is associated with infection with a novel reovirus. PLoS ONE, 5: e11487.

Parrish D. L., Behnke R. J., Gephard S. R., McCormick S. D., and Reeves G. H. 1998. Why aren't there more Atlantic salmon (*Salmo salar*)? Canadian Journal of Fisheries and Aquatic Sciences, 55: 281–287.

Pihl L., Svenson A., Moksnes P. O., and Wennehage H. 1999. Distribution of green algal mats throughout shallow soft bottoms of the Swedish Skagerrak archipelago in relation to nutrient sources and wave exposure. Journal of Sea Research, 41: 281–295.

Plarre H., Devold M., Snow M., and Nylund A. 2005. Prevalence of infectious salmon anaemia virus (ISAV) in wild salmonids in western Norway. Diseases of Aquatic Organisms, 66: 71–79.

Poppe T. T., and Seierstad S. L. 2003. First description of cardiomyopathy syndrome (CMS)-related lesions in wild Atlantic salmon *Salmo salar* in Norway. Diseases of Aquatic Organisms, 56: 87–88.

Raynard R., Wahli T., Vatsos I., and Mortensen S. 2007. DIPNET – review of disease interactions and pathogen exchange between farmed and wild finfish and shellfish in Europe. p. 452. European Commission/Veterinærmedisinsk Oppdragssenter.

Reno P. W. 1999. Infectious pancreatic necrosis virus and its prevalence. *In* Fish diseases and disorders. Vol 3: viral, bacterial and fungal infections, pp. 1–55. Ed. by P. T. K. Woo , and D. W. Bruno . CABI Publishing, Walingford, UK.

Roberge C., Einum S., Guderley H., and Bernatchez L. 2006. Rapid parallel evolutionary changes of gene transcription profiles in farmed Atlantic salmon. Molecular Ecology, 15: 9–20.

Rueness J., and Fredriksen S. 1991. An assessment of possible pollution effects on the benthic algae of the outer Oslofjord, Norway. Oebalia, 17: 223–235.

Samuel-Fitwi B., Wuertz S., Schroeder J. P., and Schulz C. 2012. Sustainability assessment tools to support aquaculture development. Journal of Cleaner Production, 32: 183–192.

Sanderson J. C., Cromey C. J., Dring M. J., and Kelly M. S. 2008. Distribution of nutrients for seaweed cultivation around salmon cages at farm sites in north–west Scotland. Aquaculture, 278: 60–68.

Sanz-Lazaro C., Belando M. D., Marin-Guirao L., Navarrete-Mier F., and Marin A. 2011. Relationship between sedimentation rates and benthic impact on Maerl beds derived from fish farming in the Mediterranean. Marine Environmental Research, 71: 22–30.

Serra-Llinares R. M., Bjørn P. A., Finstad B., Nilsen R., Harbitz A., Berg M., and Asplin L. 2014. Salmon lice infection on wild salmonids in marine protected areas: an evaluation of the Norwegian "National Salmon Fjords". Aquaculture Environment Interactions, 5: 1–16.

Skaala Ø., Glover K. A., Barlaup B. T., Svåsand T., Besnier F., Hansen M. M., and Borgstrøm R. 2012. Performance of farmed, hybrid, and wild Atlantic salmon (*Salmo salar*) families in a natural river environment. Canadian Journal of Fisheries and Aquatic Sciences, 69: 1994–2006.

Skaala Ø., Høyheim B., Glover K. A., and Dahle G. 2004. Microsatellite analysis in domesticated and wild Atlantic salmon (*Salmo salar* L.): allelic diversity and identification of individuals. Aquaculture, 240: 131–143.

Skaala Ø., Wennevik V., and Glover K. A. 2006. Evidence of temporal genetic change in wild Atlantic salmon, *Salmo salar* L., populations affected by farm escapees. ICES Journal of Marine Science, 63: 1224–1233.

Skilbrei O. T. 2010a. Adult recaptures of farmed Atlantic salmon post-smolts allowed to escape during summer. Aquaculture Environment Interactions, 1: 147–153.

Skilbrei O. T. 2010b. Reduced migratory performance of farmed Atlantic salmon post-smolts from a simulated escape during autumn. Aquaculture Environment Interactions, 1: 117–125.

Skilbrei O. T. 2013. Migratory behaviour and ocean survival of escaped out-of-season smolts of farmed Atlantic salmon *Salmo salar*. Aquaculture Environment Interactions, 3: 213–221.

Skilbrei O. T., Finstad B., Urdal K., Bakke G., Kroglund F., and Strand R. 2013. Impact of early salmon louse, *Lepeophtheirus salmonis*, infestation and differences in survival and marine growth of sea-ranched Atlantic salmon, *Salmo salar* L., smolts 1997–2009. Journal of Fish Diseases, 36: 249–260.

Skilbrei O. T., Glover K. A., Samuelsen O. B., and Lunestad B. T. 2008. A laboratory study to evaluate the use of emamectin benzoate in the control of sea lice in sea-ranched Atlantic salmon (*Salmo salar* L.). Aquaculture, 285: 2–7.

Skilbrei O. T., Heino M., and Svåsand T. 2015. Using simulated escape events to assess the annual numbers and destinies of escaped farmed Atlantic salmon of different life stages from farms sites in Norway. ICES Journal of Marine Science, 72: 670–685.

Skilbrei O. T., and Wennevik V. 2006. Survival and growth of sea-ranched Atlantic salmon, *Salmo salar* L., treated against sea lice before release. ICES Journal of Marine Science: Journal du Conseil, 63: 1317–1325.

Skogen M. D., Eknes M., Asplin L. C., and Sandvik A. D. 2009. Modelling the environmental effects of fish farming in a Norwegian fjord. Aquaculture, 298: 70–75.

Snow M., Black J., Matejusova I., McIntosh R., Baretto E., Wallace I. S., and Bruno D. W. 2010. Detection of salmonid alphavirus RNA in wild marine fish: implications for the origins of salmon pancreas disease in aquaculture. Diseases of Aquatic Organisms, 91: 177–188.

Solberg M. F., Glover K. A., Nilsen F., and Skaala Ø. 2013a. Does domestication cause changes in growth reaction norms? A study of farmed, wild and hybrid Atlantic salmon families exposed to environmental stress. PLoS ONE, 8: e54469.

Solberg M. F., Zhang Z. W., Nilsen F., and Glover K. A. 2013b. Growth reaction norms of domesticated, wild and hybrid Atlantic salmon families in response to differing social and physical environments. BMC Evolutionary Biology, 13: 234.

Sommerset I., Krossøy B., Biering E., and Frost P. 2005. Vaccines for fish in aquaculture. Expert Review of Vaccines, 4: 89–101.

Stabell O. 1984. Homing and olfaction in salmonids: a critical review with special reference to the Atlantic salmon. Biological Reviews of the Cambridge Philosophical Society, 59: 333–388.

Ståhl G. 1987. Genetic population structure of Atlantic salmon. *In* Population genetics and fishery management, pp. 121–140. Ed. by N. Ryman , and F. Utter . University of Washington Press, Seattle.

Stene A. 2013. Transmission of pancreas disease in marine salmon farming in Norway. PhD thesis, Norwegian School of Veterinary Sciences, Oslo, Norway. 117 pp.

Stene A., Viljugrein H., Yndestad H., Tavornpanich S., and Skjerve E. 2014. Transmission dynamics of pancreas disease (PD) in a Norwegian fjord: aspects of water transport, contact networks and infection pressure among salmon farms. Journal of Fish Diseases, 37: 123–134.

Stephens E. B., Newman M. W., Zachary A. L., and Hetrick F. M. 1980. A viral etiology for the annual spring epizootics of Atlantic menhaden *Brevoortia tyrannus* (Latrobe) in Chesapeake Bay. Journal of Fish Diseases, 3: 387–398.

Sterud E., Forseth T., Ugedal O., Poppe T. T., Jørgensen A., Bruheim T., and Fjeldstad H., *et al.* 2007. Severe mortality in wild Atlantic salmon *Salmo salar* due to proliferative kidney disease (PKD) caused by *Tetracapsuloides bryosalmonae* (Myxozoa). Diseases of Aquatic Organisms, 77: 191–198.

Strain P., and Hargrave B. 2005. Salmon aquaculture, nutrient fluxes and ecosystem processes in Southwestern New Brunswick. *In* Handbook of environmental chemistry, pp. 29–57. Ed. by B. Hargrave . Springer.

Taggart J. B., Verspoor E., Galvin P. T., Moran P., and Ferguson A. 1995. A minisatellite DNA marker for discriminating between European and North American Atlantic salmon (*Salmo salar*). Canadian Journal of Fisheries and Aquatic Sciences, 52: 2305–2311.

Taranger G. L., Boxaspen K. K., Madhusn A. S., and Svåsand T. (Eds) 2012b. Suggested first generation method for environmental impact indicators with respect to genetic influences from farmed salmon to wild salmon and the impact of sea lice from farmed fish on wild salmon populations (in Norwegian). Fisken og Havet 13-2012.

Taranger G. L., Svåsand T., Kvamme B. O., Kristiansen T. S., and Boxaspen K. (Eds) 2014. Risk assessment of Norwegian aquaculture 2013 (in Norwegian). Fisken og Havet, Særnummer 2-2014.

Taranger G. L., Svåsand T., Kvamme B. O., Kristiansen T. S., and Boxaspen K. K. 2012a. Risk assessment of Norwegian aquaculture [Risikovurdering norsk fiskeoppdrett] (In Norwegian). Fisken og havet, særnummer 2-2012. 131 pp.

Taranger G. L., Svåsand T., Kvamme B. O., Kristiansen T. S., and Boxaspen K. K. 2013. Risk assessment of Norwegian aquaculture 2012 (In Norwegian). Fisken og Havet, særnummer 2-2013. 164 pp.

Taranger G. L., Svåsand T., Madhun A. S., and Boxaspen K. K. 2011a. Risk assessment of environmental impact of Norwegian aquaculture 2010 [Risikovurdering miljøvirkninger av norsk fiskeoppdrett 2010] (in Norwegian). Fisken og Havet, særnr. 3-2010. 93 pp.

Taranger G. L., Svåsand T., Madhun A. S., and Boxaspen K. K. 2011b. Risk assessment – environmental impact of Norwegian aquaculture. Extracts from Risk assessment of environmental impact of Norwegian aquaculture 2010 Fisken og Havet no. 3-2011. 52 pp.

Taylor E. B. 1991. A review of local adaptation in salmonidae, with particular reference to Pacific and Atlantic salmon. Aquaculture, 98: 185–207.

Torrissen O., Jones S., Asche F., Guttormsen A., Skilbrei O. T., Nilsen F., and Horsberg T. E., *et al.* 2013. Salmon lice – impact on wild salmonids and salmon aquaculture. Journal of Fish Diseases, 36: 171–194.

Tuya F., Sanchez-Jerez P., Dempster T., Boyra A., and Haroun R. J. 2006. Changes in demersal wild fish aggregations beneath a sea-cage fish farm after the cessation of farming. Journal of Fish Biology, 69: 682–697.

Tveiten H., Bjørn P. A., Johnsen H. K., Finstad B., and McKinley R. S. 2010. Effects of the sea louse *Lepeophtheirus salmonis* on temporal changes in cortisol, sex steroids, growth and reproductive investment in Arctic charr *Salvelinus alpinus*. Journal of Fish Biology, 76: 2318–2341.

Valdemarsen T., Bannister R. J., Hansen P. K., Holmer M., and Ervik A. 2012. Biogeochemical malfunctioning in sediments beneath a deep-water fish farm. Environmental Pollution, 170: 15–25.

Valdemarsen T., Kristensen E., and Holmer M. 2009. Metabolic threshold and sulfide-buffering in diffusion controlled marine sediments impacted by continuous organic enrichment. Biogeochemistry, 95: 335–353.

Valdemarsen T., Kristensen E., and Holmer M. 2010. Sulfur, carbon, and nitrogen cycling in faunated marine sediments impacted by repeated organic enrichment. Marine Ecology Progress Series, 400: 37–53.

Vassdal T., Heggøy E., and Johansen P-O. 2012. Marine monitoring Rogaland (in Norwegian). Status report May 2012. UNI Miljø, SAM Marin. E-rapport, 26-2012.

Verspoor E., Beardmore J. A., Consuegra S., De Leaniz C. G., Hindar K., Jordan W. C., Koljonen M. L. *et al.* 2005. Population structure in the Atlantic salmon: insights from 40 years of research into genetic protein variation. Journal of Fish Biology, 67: 3–54.

Villanueva R. D., Yap H. T., and Montano M. N. E. 2006. Intensive fish farming in the Philippines is detrimental to the reef-building coral *Pocillopora damicornis*. Marine Ecology Progress Series, 316: 165–174.

Vollset K. W., Skoglund H., Barlaup B. T., Pulg U., Gabrielsen S-E., Wiers T., and Skår B., *et al.* 2014. Can the river location within a fjord explain the density of Atlantic salmon and sea trout? Marine Biology Research, 10: 268–278.

Wagner G. N., Fast M. D., and Johnson S. C. 2008. Physiology and immunology of *Lepeophtheirus salmonis* infections of salmonids. Trends in Parasitology, 24: 176–183.

Wagner G. N., McKinley R. S., Bjørn P. A., and Finstad B. 2003. Physiological impact of sea lice on swimming performance of Atlantic salmon. Journal of Fish Biology, 62: 1000–1009.

Wagner G. N., McKinley R. S., Bjørn P. A., and Finstad B. 2004. Short-term freshwater exposure benefits sea lice-infected Atlantic salmon. Journal of Fish Biology, 64: 1593–1604.

Wallace I. S., Gregory A., Murray A. G., Munro E. S., and Raynard R. S. 2008. Distribution of infectious pancreatic necrosis virus (IPNV) in wild marine fish from Scottish waters with respect to clinically infected aquaculture sites producing Atlantic salmon, *Salmo salar* L. Journal of Fish Diseases, 31: 177–186.

Wallace I. S., Murray A. G., Gregory A., and Raynard R. S. 2005. The prevalence of infectious pancreatic necrosis virus in wild marine fish from Scotland with reference to clinically infected salmon farms. Abstract in EAFP 12th international Convergence, 12.

Wells A., Grierson C. E., MacKenzie M., Russon I., Reinardy H., Middlemiss C., Bjørn P. A., *et al.* 2006. Physiological effects of simultaneous, abrupt seawater entry and sea lice (*Lepeophtheirus salmonis*) infestation of wild, sea-run brown trout (*Salmo trutta*) smolts. Canadian Journal of Fisheries and Aquatic Sciences, 64: 1360–1369.

Wells A., Grierson C. E., Marshall L., MacKenzie M., Russon I. J., Reinardy H., Sivertsgård R., *et al.* 2007. Physiological consequences of premature freshwater return; for wild sea-run brown trout (Salmo trutta) postsmolts infested with sea lice (*Lepeophtheirus salmonis*). Canadian Journal of Fisheries and Aquatic Sciences, 64: 1360–1369.

Wiik-Nielsen C. R., Lovoll M., Sandlund N., Faller R., Wiik-Nielsen J., and Jensen B. B. 2012. First detection of piscine reovirus (PRV) in marine fish species. Diseases of Aquatic Organisms, 97: 255–258.

Wilding T. A., Cromey C. J., Nickell T. D., and Hughes D. J. 2012. Salmon farm impacts on muddy-sediment megabenthic assemblages on the west coast of Scotland. Aquaculture Environment Interactions, 2: 145–156.

Worm B., and Sommer U. 2000. Rapid direct and indirect effects of a single nutrient pulse in a seaweed-epiphyte-grazer system. Marine Ecology Progress Series, 202: 283–288.

Zhang Z., Glover K. A., Wennevik V., Svåsand T., Sørvik A. G. E., Fiske P., and Karlsson S., *et al.* 2013. Genetic analysis of Atlantic salmon captured in a netting station reveals multiple escapement events from commercial fish farms. Fisheries Management and Ecology, 20: 42–51.

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Contribution to the Themed Section: 'Risk Assessment'

Review

Quantitative environmental risk assessments in the context of marine spatial management: current approaches and some perspectives

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Marine spatial planning (MSP) requires spatially explicit environmental risk assessment (ERA) frameworks with quantitative or probabilistic measures of risk, enabling an evaluation of spatial management scenarios. ERAs comprise the steps of risk identification, risk analysis, and risk evaluation. A review of ERAs in in the context of spatial management revealed a synonymous use of the concepts of risk, vulnerability and impact, a need to account for uncertainty and a lack of a clear link between risk analysis and risk evaluation. In a case study, we addressed some of the identified gaps and predicted the risk of changing the current state of benthic disturbance by bottom trawling due to future MSP measures in the German EEZ of the North Sea. We used a quantitative, dynamic, and spatially explicit approach where we combined a Bayesian belief network with GIS to showcase the steps of risk characterization, risk analysis, and risk evaluation. We distinguished 10 benthic communities and 6 international fishing fleets. The risk analysis produced spatially explicit estimates of benthic disturbance, which was computed as a ratio between relative local mortality by benthic trawling and the recovery potential after a trawl event. Results showed great differences in spatial patterns of benthic disturbance when accounting for different environmental impacts of the respective fleets. To illustrate a risk evaluation process, we simulated a spatial shift of the international effort of two beam trawl fleets, which are affected the most by future offshore wind development. The Bayesian belief network (BN) model was able to predict the proportion of the area where benthic disturbance likely increases. In conclusion, MSP processes should embed ERA frameworks which allow for the integration of multiple risk assessments and the quantification of related risks as well as uncertainties at a common spatial scale.

Keywords: Bayesian belief network, fishing frequency, GIS, marine spatial planning, review.

Introduction

Place-based management tools such as marine spatial planning (MSP) are advocated worldwide to support the implementation of an ecosystem approach to marine management (Katsanevakis *et al.*, 2011). In Europe, MSP is regarded as a means to solve inter-sectoral and cross-border conflicts over maritime space (Douvere and Ehler, 2010) and is promoted by the upcoming EU MSP Directive (Commission, 2014). The latter encourages blue growth and the sustainable use of marine resources (Qiu and Jones, 2013; Brennan *et al.*, 2014). One of the future challenges for European regional Seas is the alignment of the sustainable use of the marine resources with the maintenance of ecosystem health and functioning, as demanded by the EU Marine Strategy Framework Directive (MSFD) (Commission, 2008). Hence, an ecosystem-based MSP process

should seek to manage human activities while balancing multiple ecological, economic, and social objectives (Foley *et al.*, 2013).

As a consequence, an ecosystem-based MSP approach requires robust estimates of the risks of adverse effects of cumulative human pressures on the marine environment at meaningful ecological scales (Eastwood *et al.*, 2007; Halpern *et al.*, 2008a; Stelzenmüller *et al.*, 2010; Fock *et al.*, 2011). Environmental risk assessments (ERAs) (Hope, 2006) that link spatially explicit information on the vulnerability of ecosystem components with the occurrence and magnitude of pressures are basic for the successful implementation of an ecosystem-based MSP approach. The fast growing number of MSP initiatives (Carneiro, 2013; Collie *et al.*, 2013) highlights the increasing importance of spatially explicit ERAs and underpins the need for quantitative or probabilistic measures of risk.

In general, quantitative risk assessments rely on mathematical models to predict the response of the ecosystem component to changing pressures. Qualitative approaches, however, use ecosystem attributes combined with ecological receptors and stressors (Astles *et al.*, 2006). As for today, empirical studies on ERAs that provide, for example, spatially explicit quantifications of risk in relation to management options appear at a slower pace and take various risk assessment approaches (Stelzenmüller *et al.*, 2010; Fock *et al.*, 2011; Gimpel *et al.*, 2013; Redfern *et al.*, 2013). In the light of existing EU policies, in particular the MSFD and new MSP Directive, there is a growing need to align various spatially explicit ERAs to on-going spatial management processes.

To account for this, we adopted the risk assessment framework described in Cormier *et al.* (2013) to first, assess current ERA approaches and second, structure a case study on the risk of benthic disturbance in the German EEZ of the North Sea. The risk assessment framework comprises three steps. First, the risk identification specifies the pressure(s) of concern and the significant ecosystem components. Second, the risk analysis accounts for both, the probability and the magnitude of the pressure, its impacts on ecosystem components, and the degree of uncertainty involved. Third, the risk evaluation assesses the likely impacts on ecosystem components under alternative management measures.

We first reviewed empirical studies of spatially explicit and quantitative ERAs in the context of spatial management and assessed in detail the methods used for the risk identification, risk analysis, and risk evaluation. To address some identified methodological gaps, we defined a case study which describes the stepwise assessment of the risk when changing the current state of benthic disturbance by trawling due to future MSP measures in the German EEZ. Thus, in the risk identification step, we defined the offshore wind development (OWD) and the related displacement of fishing effort as pressures. We identified 10 benthic communities as described by Rachor and Nehmer (2003) as an example of significant ecosystem components since the good environmental status of seabed integrity reflects one of the goals of the MSFD. In the risk analysis step, we computed spatial estimates of a benthic disturbance indicator (Fock, 2011a), which was defined as a ratio between relative local mortality by dermeral trawling fleets and recovery potential of benthic communities (see Hiddink *et al.*, 2006a). For the risk evaluation, we used a spatially explicit probabilistic approach that allows a dynamic assessment of possible trade-offs of alternative spatial management scenarios. We coupled a BN with GIS and predicted occurrence probabilities of different states of benthic disturbance and per cent changes of the study area in relation to simulated spatial management objectives. BNs are acyclic graphs that represent causal dependencies among a set of random variables by directed links between them (McCann *et al.*, 2006). Recently, they have been used in combination with GIS to conduct a spatially explicit assessment of the risk involved with spatial management options (Stelzenmüller *et al.*, 2011; Johnson *et al.*, 2012; Grêt-Regamey *et al.*, 2013a, b). In summary, here we identified some shortcomings of current spatially explicit ERA approaches, and showed some perspectives for assessing trade-offs of MSP scenarios in the German EEZ of the North Sea. Finally, we reflected on the challenges ahead when it comes to the integration of many assessment outputs in a multiple objectives spatial management context.

Material and methods

Risk assessment framework and review of current approaches

We adopted the standardized risk assessment framework defined by Cormier *et al.* (2013) to frame the steps of risk identification, risk analysis and risk evaluation in a spatial management context (Figure 1). We then analysed recent empirical studies of (semi-) quantitative ERAs in the context of marine spatial management with regard to these key steps. Here spatial management was rather broadly defined and encompassed studies concerned with MSP, sectoral management, or marine conservation. With the help of multiple combinations of the keywords such as ERA, risk analysis, quantitative, vulnerability, spatial management, MSP, and map(ping), we selected a total of 32 peer-reviewed papers. In the following, we describe the three risk assessment steps in more detail and specify what information has been extracted from the reviewed literature.

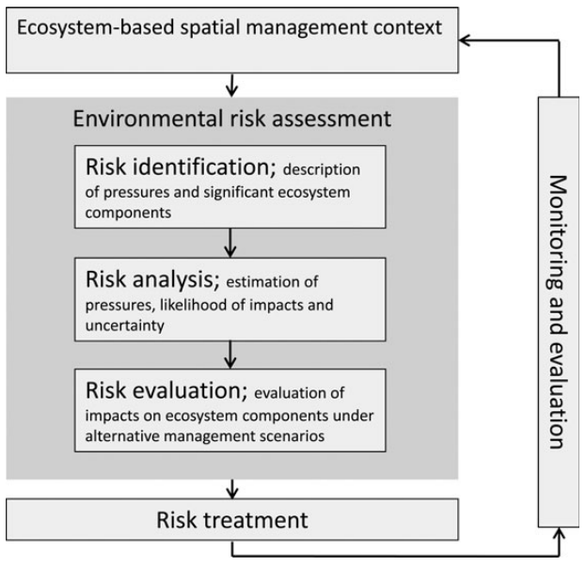

Figure 1. Simplified risk management process redrawn from Cormier *et al.* (2013) in the context of marine spatial management such as MSP. Spatial management goals and operational objectives (Stelzenmüller *et al.*, 2013) determine the contents of the ERA. Risk assessment results enter the risk treatment phase which produces management options, based on cost – benefit analysis of implementation. Suggested management options will in turn feedback in to the spatial management process (development, implementation, or evaluation process).

Risk identification

The risk identification comprises the definition of significant ecosystem components, stressors, or pressures as well as the related environmental cause–effect pathways defined by the operational management objectives for a given area. Operational objectives have specific, measureable, achievable, realistic and time-limited targets, such that management measures can be fitted and performance can be evaluated (Stelzenmüller *et al.*, 2013). Stressors are single or multiple human pressures while cumulative impacts are described as the combined impact of multiple pressures over space and time (MacDonald, 2000). Here risk identification comprises also an estimate of the occurrence probability and magnitude of the pressure and the spatial quantification of the identified ecosystem components or state indicator. According to this definition, the assessed pressures and ecosystem components or state indicators together with the methods used to quantify their occurrence in the respective area were extracted from the reviewed empirical studies.

Risk analysis

This step addresses the quantification of impacts on ecosystem components that accounts for existing mitigation or management measures as well as the risk acceptance in society. The latter should be reflected in the operational management objectives. The impact is generally defined as a function of the vulnerability of ecosystem components and the occurrence likelihood and magnitude of a pressure (Stelzenmüller *et al.*, 2010). De Lange *et al.* (2010) proposed to define vulnerability of an ecosystem component by exposure and sensitivity to a pressure as well as its recovery potential. The sensitivity to a pressure is due to structural properties, functions, or trophic relations of the ecosystem component while recovery depends on population recovery, resilience, positive feedback loops, and adaption (Tyler-Walters *et al.*, 2001; Hope, 2006; Halpern *et al.*, 2008b). We classified each case study according to the type of sensitivity measure used (expert knowledge, model output, empirical data) and the vulnerability assessment approach applied. Uncertainty should be recognized and constructively handled for any integrated risk assessment or models-based decision support (Rotmans and van Asselt, 2001). For instance, a recent review by Ferdous *et al.* (2013) assessed methods which allow recognizing and evaluating the implications of uncertainty in a risk analysis. Thus, we reported further if any form of uncertainty analysis was undertaken and which methods have been used.

Risk evaluation

The result of a risk evaluation indicates whether or not new management actions need to be taken. Technically, this requires the evaluation of management scenarios, including the "the business as usual" scenario. More precisely, it entails a comprehensive assessment of the proposed management measures and scenarios with respect to the potential risks for relevant ecosystem components. Thus, we investigated what kind of management scenarios, if at all, have been tested in the empirical studies.

Case study area and context

The here described risk assessment framework has been hardly applied to marine ecosystems in all aspects. We thus designed a case study assessing future MSP measures in the German EEZ and their likely implications for benthic communities using a quantitative, dynamic, and spatially explicit approach. Since 2008, the maritime spatial plan is legally binding in the German EEZ and comprises designated preference areas for a number of sectors except fishing, including special areas of conservation designated under the Habitat Directive (92/43/EEC, 1992) (BMVBS, 2009; Fock, 2011b; Stelzenmüller *et al.*, 2011; Gimpel *et al.*, 2013). Further environmental objectives with potential spatial management measures are defined by the MSFD and require implementation by 2020. For illustration purposes, we simplified this rather complex spatial management context and focused only on seabed integrity and defined the hypothetical operational management objective "The relative benthic disturbance by trawling should not deteriorate with respect to current levels". This operational objective defines the impact of trawling on benthic communities as the measure or indicator of concern and specifies the current level as the reference point. Therefore, future MSP measures, which comprise the designation of OWD sites within ~35% of the study area, will be assessed against the here defined management objective. Future OWD sites in the German EEZ show a clear spatial overlap with prevailing patterns of fishing (Stelzenmüller *et al.*, 2011). Thus, the potential area loss for fishing will most likely result in an effort displacement with as yet unknown environmental and economic consequences. In the following, we describe the risk assessment steps for the current case study.

Risk identification: *OWD, fisheries, and benthic communities*

We considered the currently designated OWD sites as MSP measures as well as the submitted application areas. The development of this sector triggers a number of conflicts with other human uses through the competition for the same space (Gimpel *et al.*, 2013). The highest conflict potential can be expected between the (international) fishing sector and the OWD, since, for example, roughly 15% of the total international large beam trawl effort takes place in areas where OWD is envisaged. Thus, we defined the average spatial and temporal activity of six different fishing fleets as pressures following Fock (2011) and Stelzenmüller *et al.* (2011) regarding to seabed integrity (as specified above). For this we combined German, Dutch, and Danish VMS (vessel monitoring system) and logbook data from 2005 to 2008 to calculate the average bottom trawling effort (total hours fishing per year) per 3 × 3 nm grid cell (31 km^2). We distinguished six different fleets, which are beam trawlers operating with 80 mm mesh size and an engine power >221 kW (Beam80lrg) and <221 kW (Beam80sml), beam trawler with 16–31 mm mesh size and an engine power >221 kW (Beam1631lrg), and <221 kW (Beam1631sml), and otter trawlers with 80 mm mesh size and an engine power >221 kW (Otter80lrg) and <221 kW (Otter1631sml). For each grid cell, we computed the frequency with which the seabed surface has been swept by the respective fleet (Ffr_{ik}) using the formula and parameters also presented in Fock (2011a) ($Ff\,r_{ik} = (T_{ik} \times V_k \times W_k/A_i)$; with T_{ik}, total hours fished (h), V_k, average fishing speed (km h^{-1}), w_k, net spread (km), and A_i, surface area in km^2). The ecosystem components of concern were 10 benthic communities with a defined spatial distribution (Figure 2) and specific characteristics such as habitat preference or recovery frequency (Table 1) (Rachor and Nehmer, 2003; Pesch *et al.*, 2008; Fock, 2011a). Thus with the help of GIS we allocated to each grid cell the most dominant benthic community with respective measures of recovery potential and mortality rates (see below) together with the average fishing frequency per fleet.

Risk analysis: *Measuring benthic disturbance*

The next step required the definition of vulnerability of the ecosystem components to fishing pressures exerted by the different fleets.

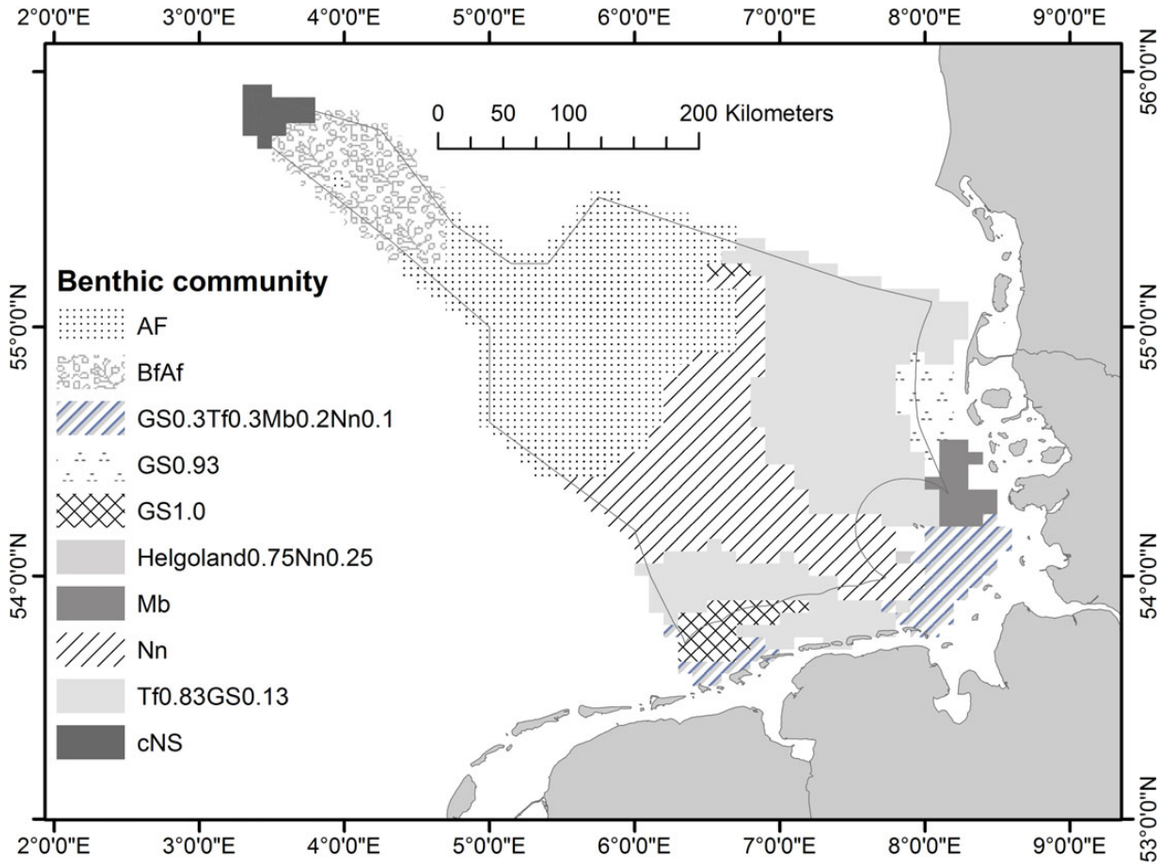

Figure 2. Predicted spatial distribution of the infaunal benthic community in the German EEZ of the North Sea and adjacent waters (redrawn after Pesch *et al.*, 2008).

We built on a previous study (Fock, 2011a and references therein) and computed spatial estimates of the disturbance indicator (DI). DI_i reflects an overall relative local vulnerability of a benthic community to bottom trawling and is defined as the ratio between mortality and recovery (M_i/R_i). DI_i is a unitless relative ratio and $DI_i = 1$ indicates a balance between relative local mortality and recovery. $DI_i > 1$ indicates locally greater mortality rates than recovery potential, whereas $DI_i < 1$ indicates that the recovery potential exceeds local mortality rates by trawling.

The computation of this ratio requires relative estimates of recovery time and recovers frequency for each of the 10 benthic communities (see Table 1). We used the proportion of typical sediment categories (mud, sand, muddy sand, and gravel) favoured by the respective benthic communities (Rachor and Nehmer, 2003) to construct combined relative measures of recovery time (y) ($RT_{\mathrm{BC}} = \sum R_{\mathrm{Sediment}} \cdot$ proportion sediment) and recover frequency (year^{-1}) ($Rfr_{\mathrm{BC}} = \sum Rfr_{\mathrm{Sediment}} \cdot$ proportion sediment), both in relation to one trawling event. With this we computed for each grid cell the relative recovery for each benthic community to 90% of the abundance before trawling as a function of the recovery time and recover frequency $R_i = 1 - (1 - 0.9RT_{\mathrm{BC}})^{RfrBC}$ (Fock, 2011a). Hence, the here applied measure of sensitivity to benthic trawling is derived from model outputs presented in Hiddink *et al.* (2006a) and empirical results by Rachor and Nehmer (2003). In a next step, we computed for each grid cell the local mortality rate for each benthic community. For this, we used the average percentage decline of abundance per sediment type (taken from Fock, 2011a) to construct an average combined measure of mortality per benthic community ($MR_{\mathrm{BC}} = \sum \mathrm{Decline}_{\mathrm{Sediment}} \cdot$ proportion sediment) (see Table 1). Therefore, we computed for each grid cell the fleet specific mortality rate for the benthic community as $M_{ik} = 1 - (1 - MR_{\mathrm{BC}})^{Ffrik}$. The overall local mortality rate is the sum of these mortality rates weighted by a respective impact sore (is); $M_i = \sum_{k=1}^{n} M_{ik} \times is_k$ (modified after Fock, 2011a). This finally allowed us to compute the ratio between relative local mortality and recovery (M_i/R_i), and we refer to this as disturbance indicator (DI_i). We further explored the uncertainty within the estimates of benthic disturbance by accounting for fleet-specific impacts on benthic communities. For that reason, we calculated DI_i based on a local overall mortality rate (M_i) by assuming equal impacts of each fleet (i.e. impact score $is_{\mathrm{k}} = 1$). Alternatively, we computed DI_{iw} with a local overall mortality rate weighted by different impact scores (adapted from Fock, 2011a). Here highest weight is given to the beam trawlers operating with a mesh size of >80 mm, which represent mainly the fishery targeting flatfish, and least weight is given to the small beam trawlers using mesh sizes of 16–31 mm, representing the shrimp fishery ($is_{\mathrm{BEAM80lrg}} = 1$; $is_{\mathrm{BEAM80sml}} = 1$; $is_{\mathrm{BEAM1631lrg}} = 0.1$; $is_{\mathrm{BEAM1631sml}} = 0.1$; $is_{\mathrm{OTTER80lrg}} = 0.15$; $is_{\mathrm{OTTER80sml}} = 0.15$). We compiled for each grid cell the respective measures of recovery, mortality, and

Table 1. Ten benthic communities as defined by Rachor and Nehmer (2003) comprising *Amphiura filiformis* 89% (AF); *Bathyporeia fabulina* 85%, *Amphiura filiformis* 10% (BtAf); central North Sea (cNS); *Tabulina fabula* (Tf) 83%, *Goniadella spisula* (GS) 12,5% (Tf0.83GS0.13); GS30%, Tf30%, *Macoma balthica* (Mb) 20%, *Nucula nitidosa* (Nn) 10% (GS0.3Tf0.3Mb0.2Nn0.1); GS 100% (GS1.0); GS 93% (GS0.93); Helgoland depth 75%, Nn 25% (Helgoland0.75Nn0.25); Mb 100% (Mb); Nn 84% (Nn).

Benthic community	AF	BtAf	cNS	Tf0.83 GS0.13	GS0.3Tf0.3Mb0.2 Nn0.1	GS1.0	GS0.93	Helgoland 0.75Nn0.25	*Mb*	*Nn*
Prop mud[a]					0.11			0.8		0.84
Prop muddy sand[a]	1	0.15	0.5		0.28					0.16
Prop sand[a]		0.85	0.5	0.93	0.44	0.5	0.6	0.15	0.50	
Prop gravel[a]				0.07	0.16	0.5	0.4	0.05	0.50	
R_{Mud} (days)	25	25	25	25	25	25	25	25		
$R_{MuddySand}$ (days)	111	111	111	111	111	111	111	111	111	111
R_{Sand} (days)	193	193	193	193	193	193	193	193	193	193
Rfr_{Mud} (year^{-1})	14	14	14	14	14	14	14	14	14	14
$Rfr_{MuddySand}$ (year^{-1})	3	3	3	3	3	3	3	3	3	3
Rfr_{Sand}(year^{-1})	1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5	1.5
Rfr_{Gravel} (year^{-1})	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
$Decline_{Mud}$ (proportion)	0.345	0.345	0.345	0.345	0.345	0.345	0.345	0.345	0.345	0.345
$Decline_{MuddySand}$ (proportion)	0.675	0.675	0.675	0.675	0.675	0.675	0.675	0.675	0.675	0.675
$Decline_{Sand}$ (proportion)	0.535	0.535	0.535	0.535	0.535	0.535	0.535	0.535	0.535	0.535
$Decline_{Gravel}$ (proportion)	0.74	0.74	0.74	0.74	0.74	0.74	0.74	0.74	0.74	0.74
RT_{BC} (year)	0.3	0.5	0.42	0.49	0.33	0.26	0.32	0.13	0.26	0.11
Rfr_{BC} (year^{-1})	3	1.73	2.25	1.4	3.06	0.8	0.94	11.5	0.8	12.24
R_{BC}	0.62	0.64	0.65	0.56	0.65	0.2	0.27	0.77	0.2	0.71
MR_{BC} (proportion)	0.68	0.56	0.61	0.55	0.58	0.64	0.62	0.39	0.64	0.40

[a]The proportion of sediment per benthic community has been derived from Fock (2011a) based on a study from Rachor and Nehmer (2003).
For each community the relative distribution on four different sediment types, their sediment specific recovery time (*R*), recover frequency (*Rfr*), and decline after one trawling event (Decline) is given (after Fock, 2011a; Hiddink *et al.*, 2006a). Further, the community-specific combined values are listed as relative combined recovery time (RT_{BC}), the relative combined recover frequency (Rfr_{BC}), the relative combined recovery rate (R_{BC}), and the relative combined abundance decline after one trawling event (MR_{BC}).

benthic disturbance in ArcGIS 10.0 using the attribute table of the vector grid for subsequent mapping. Thus, *DI* and DI_w describe spatially disaggregated alternative assumptions of the relative state of benthic disturbance, based on the average bottom trawling effort from 2005 to 2008.

Risk evaluation: *Trade-off analysis of MSP measures*

This final step corresponds to the evaluation of the risk of worsening the current state of benthic disturbance due to future MSP measures in the German EEZ. Our scenario applies to planned OWD sites, where, in case of their realization, extensive areas would be closed for fishery. As a rough estimate 15% of the large beam trawl effort and 3% of the small beam trawl effort would be affected. Effects on the fleets using otter boards are negligible. Thus, we defined the following spatial management scenario: "Current and future offshore wind development cause a spatial shift of 15% of the total fishing frequency of large beam trawlers (Beam80lrg) and 3% of the small beam trawlers (Beam1631sml)". We combined a BN with GIS to predict changing likelihoods of benthic disturbance states due to different trawling effort patterns. We used the Netica software system (www.norsys.com) (see details on the inference algorithm implemented in Netica in Spiegelhalter and Dawid, 1993) to develop the BN model and used the attribute table compiled in the GIS to both built the prior probabilities for each variable (referred to as BN node) and to populate the conditional probability tables (see Table 2). The BN model contains the deterministic relationships described above and reflects the causal links of all parameters required to calculate the unweighted and weighted disturbance indicator (Figure 3). Benthic communities and the fishing frequencies of the six fleets are parent nodes and are considered to be independent from each other. Each parent node has discrete states (e.g. type of benthic community, category of fishing frequency) with an associated probability of occurrence. Fleet-specific mortality rates are represented as functions of the respective fishing frequencies and the estimated decline rates for each benthic community. The overall mortality rate and weighted mortality rate are child nodes of the fleet-specific mortality rates and are defined by their deterministic relationships with their parent nodes. Recover frequency, recovery time, and abundance decline are child nodes of the benthic communities. The likelihoods of the states of the disturbance indicator nodes are predicted as a function of the likelihood of the overall relative mortality rates (unweighted and weighted) and the predicted recovery by the benthic community.

We also assessed the sensitivity of the disturbance indictor node (*DI*) to the influence of the parent nodes by calculating the variance reduction. The performance or "goodness of fit" of the BN model was tested by computing the spherical payoff index (see Marcot *et al.*, 2006). The latter describes how well the predictions of the BN match the actual cases and is defined as the mean probability value of a given state averaged over all cases.

Subsequently, we explored the effects of the planned OWD sites on the two measures of benthic disturbance (*DI* and DI_w) with the help of the trained BN. We assumed that in 15% of the area the likelihood of experiencing the lowest level of fishing pressures by large beam trawlers will increase (since 15% of the area will be closed for this fisheries). Assuming that the fishing effort will relocate in areas with already high fishing intensity, the probability of a unit area experiencing the highest level of fishing pressures (or being in state 3) must increase. Thus we changed in the BN model the prior distribution for the Beam80lrg node, with now 47% of the area having a value from 0 to 0.0025 and in 53% of the area values range between 0.06 and 1.16. We inferred subsequently the changes of the probability distributions of the *DI* and DI_w nodes.

Table 2. Description of BN model nodes, discretization method, and states.

BN node	States	Description
Recover_frequency_BC	0 – 1.4; >1.4 – 3; >3 – 12.24	Relative combined recover frequency for each benthic community ($year^{-1}$) (Table 1; $Rfr_{BC} = \sum Rfr_{Sediment} \cdot$ proportion sediment) from benthic trawling.
Recovery_time_BC	0 – 0.26; >0.26 – 0.33; >0.33 – 0.5	Relative combined recovery time for each benthic community (year) (Table 1; $RT_{BC} = \sum R_{Sediment} \cdot$ proportion sediment) from benthic trawling.
Abundance_decline_BC	0 – 0.5; >0.5 – 0.58; >0.58 – 0.68	Relative combined abundance decline after one trawling event for each benthic community (Table 1; $MR_{BC} = \sum Decline_{Sediment} \cdot$ proportion sediment)
Recovery	0 – 0.56; >0.56 – 0.62; >0.62 – 0.78	Relative local recovery rate for each benthic community (Table 1; $R_i = 1 - (1 - 0.9 \cdot RT_{BC})^{RfrBC}$)
FrBeam80LR FrBeam80SM FrBeam1631LR FrBeam1631SM FrOtter80LR FrOtter80SM	0 – 0.0025; >0.0025 – 0.06; >0.06 – 1.16 0; >0 – 0.0004; >0.00004 – 0.076 0; >0 – 0.00019; >0.000019 – 0.000347 0; >0 – 0.07; >0.07 – 1.17 0; >0 – 0.000279; >0.000279 – 0.335 0 – 0.0007; >0.0007 – 0.012; >0.012 – 0.524	Fleet-specific mean (2005 – 2008) fishing frequency ($Ff\,r_{ik} = (T_{ik} \times V_k \times w_k/A_i)$; with T_{ik}, total hours fished (h), V_k, average fishing speed (km h^{-1}), w_k, net spread (km), and A_i, surface area in km^2) with which the surface area has been swept (Beam = beam trawlers, Otter = otter board trawlers, 80 = 80 mm mesh size, 1631 = 16 – 31 mm mesh size, *LR* = engine power >221 kW, SM = engine power <221 kW).
M_Beam80LR M_Beam80SM M_Beam1631LR M_Beam1631SM M_Otter80LR M_Otter80SM	0 – 0.0021; >0.0021 – 0.05; >0.05 – 0.45 0; >0 – 0.0007; >0.0007 – 0.058 0; >0 – 0.000134; >0.000134 – 0.00039 0; >0 – 0.06; >0.06 – 0.64 0; >0 – 0.000313; >0.000313 – 0.31 0; >0 – 0.000313; >0.000313 – 0.31	Fleet-specific relative mean mortality rates of the prevailing benthic community as a function of the mean frequency of the respective fleet and the combined average abundance decline rate ($M_{ik} = 1 - (1 - MR_{BC})_{ik}^{Ffr}$ (Beam = beam trawlers, Otter = otter board trawlers, 80 = 80 mm mesh size, 1631 = 16 – 31 mm mesh size, *LR* = engine power >221 kW, SM = engine power <221 kW) (see Table 1).
Mortality_rate	0 – 0.032; >0.032 – 0.14; >0.14 – 0.84	Overall mean local mortality rate expressed as the sum of the mean local mortality rates per fleet (from 2005 to 2008) weighted by equal impact scores (*is*): $M_i = \sum_{k=1}^{n} M_{iK} \times is_k$; $is_k = 1$.
Mortality_rate_W	0 – 0.032; >0.032 – 0.14; >0.14 – 0.84	Overall mean local mortality rate weighted by different impact scores (*is*): $is_{BEAM80lrg} = 1$; $is_{BEAM80sml} = 1$; $is_{BEAM1631lrg} = 0.1$; $is_{BEAM1631sml} = 0.1$; $is_{OTTER80lrg} = 0.15$; $is_{OTTER80sml} = 0.15$.
Disturbance_indicator	0 – 0.3; >0.3 – 0.5; >0.5 – 1; >1 – 3	Estimated disturbance indicator (DI_i) as the ratio between mortality rate and recovery.
Disturbance_indicator_W	0 – 0.3; >0.3 – 0.5; >0.5 – 1; >1 – 3	Estimated disturbance indicator (DI_{iW}) as the ratio between the weighted mortality rate and recovery.
Benthic_communities	AF; BtAf; cNS; Tf0.83GS0.13; GS0.3Tf0.3Mb0.2Nn0.1; GS1.0; GS0.93; Helgoland0.75Nn0.25; Mb; Nn	Ten categories of benthic communities as defined by Rachor and Nehmer, (2003) comprising *Amphiura filiformis* 89% (AF); *Bathyporeia fabulina* (85%), *Amphiura filiformis* (10%) (BfAf); central North Sea (cNS); *Tabulina fabula* (83%), *Goniadella spisula* (12.5%) (Tf0.83GS0.13); *Goniadella spisula* (30%), *Tabulina fabula* (30%), *Macoma balthica* (20%), *Nucula nitidosa* (10%) (GS0.3Tf0.3Mb0.2Nn0.1); *Goniadella spisula* (100%) (GS1.0); *Goniadella spisula* (93%) (GS0.93); *Helgoland Depth* 75%, *Nucula nitidosa* (25%) (Helgoland0.75Nn0.25); *Macoma balthica* (100%) (Mb); *Nucula nitidosa* (84%) (Nn).

All model nodes reflect attributes from the 3 × 3 nm vector grid.

Based on the same rational, we have changed the prior distribution for the Beam1631sml node assuming that in 66% the area no fishing is carried out by this fleet, while in 12% of the area values range between >0 and 0.07, and in 22% of the area values range between <0.07 and 1.17. It is worth mentioning that the here defined spatial shift in fishing effort reflects one out of many possible changes to the prior distributions of the parent nodes reflecting the fishing frequencies of the six fleets.

Results

Review of current approaches

The results of the structured literature review of 32 papers are summarized in Table 3. Most studies focused on one or two stressors with a clear emphasis on fisheries; other activities included aggregate mining and marine traffic. Cumulative pressures were analysed in a quarter of all examined studies, mostly assuming additive effects. We observed that the measure of sensitivity of ecosystem components or indicators was mostly related to a metric derived from a model output which based either on empirical data or expert knowledge. In contrast, a quarter of the reviewed studies were based on expert knowledge and three studies being based exclusively on empirical data. Another important result was that the terminology of risk, vulnerability, and impact varied greatly across the studies and has been used synonymously. Despite this variation in terminology, the components to calculate a measure of vulnerability or impact have been similar across all cases. All studies defined vulnerability or impact as a function of a measure of ecosystem sensitivity and

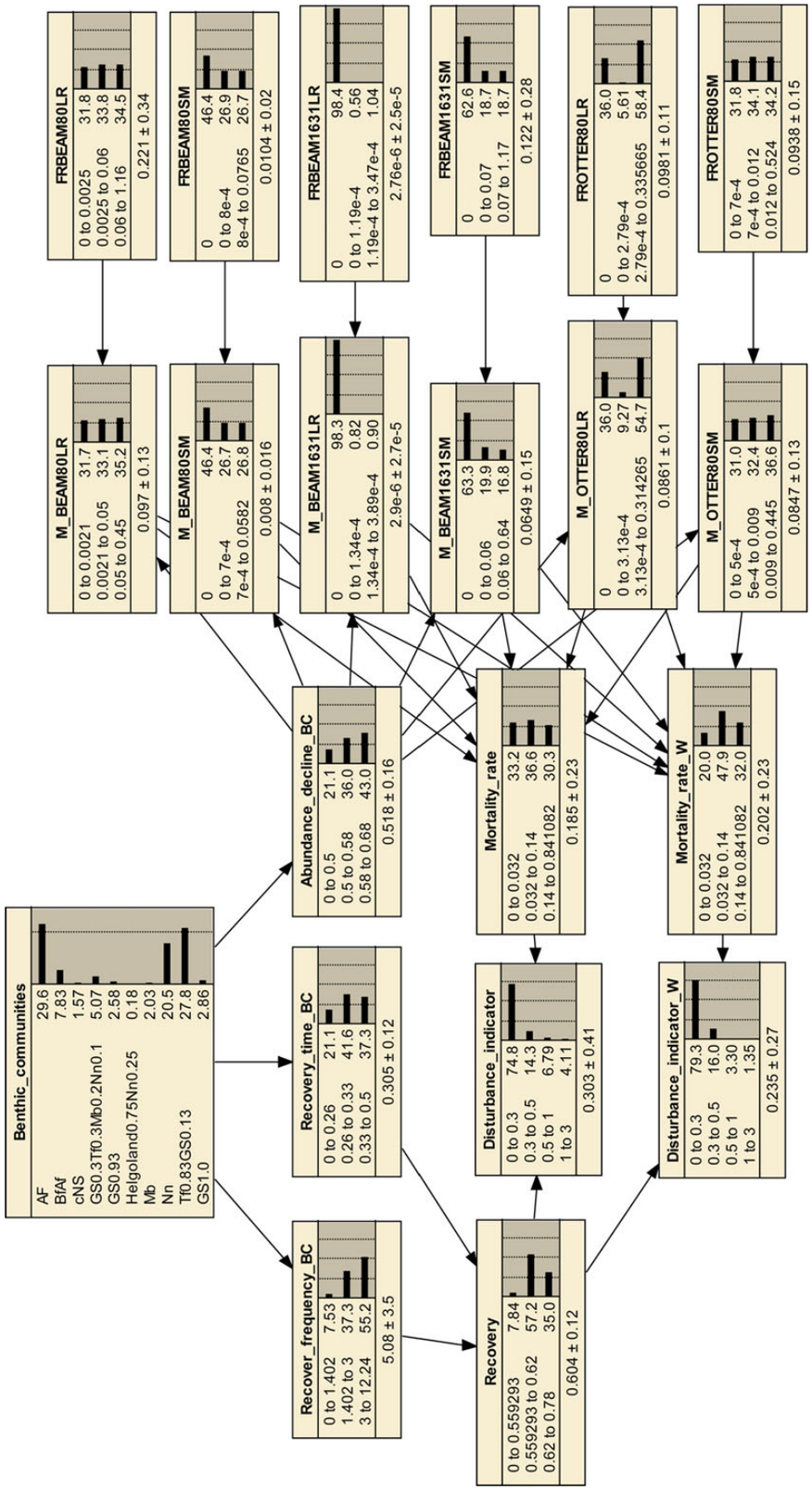

Figure 3. Structure of the BN for assessing future MSP measures in the German EEZ and their likely implications for benthic communities. Values for categorical probabilities (%) of each node are given for the baseline scenario (referred to as "business as usual scenario") (node definitions in Table 2).

Table 3. List of 32 recent empirical studies of (semi-) quantitative ERAs in the context of the development, implementation or evaluation of marine spatial management.

Scale and location small <500 000 km^2 meso ≥500 000 – 106 km^2, large≥106 km^2	Risk identification and characterization		Risk analysis			Risk evaluation	References
	Stressor(driver)/ pressure indicators used	Ecosystem components/ state indicators	Measure of sensitivity of state indicators	Measure and approach used of vulnerability/ risk/impact of ecosystem/ area	Assessment output type	Management scenario analysis (assessed: yes/no)	
Small (ca. 270 000 km^2); Great Barrier Reef MPAs, Australia	Pollution	Multiple habitats (coral reefs and seagrass beds)	Model output	Frequency of plume occurrence with spatially distributed loads; final maps of exposure (E) = annual frequency of plume occurrence grid (F) × sum of spatially distributed TSS and DIN loads grid [for all rivers (P)]	Quantitative measures per unit area; mapping out approach of frequencies	No	Alvarez-Romero *et al.*, (2013)
Meso (ca. 500 000 km^2); Canada's EEZ, Pacific coast	Cumulative pressure (additive) from human stressors	Multiple habitats	Expert knowledge	Cumulative impact = $\sum$(intensity × habitat × vulnerability (vulnerability score for activity i and habitat j, by expert judgement), MPA restrictions included	Quantitative measure per unit area (400 m grid); cumulative impact score matrices	Yes, three scenarios were used: (i) include each fishery separately, (ii) summarize fisheries by type of impact, (iii) include only one layer for commercial and one for recreational fisheries	Ban *et al.* (2010)
Small; 3 Italian MPA, Mediterranean Sea	Multiple human and environmental stressors	Multiple habitats	Empirical data, expert knowledge	Environmental diagnostic = $\sum$ scores of individual habitat per cell for degradation and risk [level]; weighted vulnerability [vulnerability of habitat × number of cells where the habitat is present]; environmental quality [$\sum$naturalistic × economic × aesthetic × rarity of the habitat]; susceptibility to human use [number of habitats × importance]	Semi-quantitative measure per unit area; mapping out approach of environmental quality, susceptibility to use, weighted vulnerability	No	Bianchi *et al.* (2012)
Large; Australasia	Cumulative pressure (additive, antagonistic, synergistic) from global (climate change) and local (nutrient input) stressors	Habitat (seagrass)	Empirical data	Additive effects model (effect size × stressor values) to test for interactions between pressures (no, antagonistic and synergistic interactions)	Quantitative measure per unit area (100 km^2); interactive impact maps (local and global stressors)	Yes, the management effect of each pressure has been assessed	Brown *et al.* (2013)

Continued

Table 3. *Continued*

Scale and location small <500 000 km² meso ≥500 000 – 106 km², large≥106 km²	Risk identification and characterization		Risk analysis			Risk evaluation	References
	Stressor(driver)/ pressure indicators used	Ecosystem components/ state indicators	Measure of sensitivity of state indicators	Measure and approach used of vulnerability/ risk/impact of ecosystem/ area	Assessment output type	Management scenario analysis (assessed: yes/no)	
Small (ca. 70 km²); Ebro Delta, NW Mediterranean Sea	Offshore windfarms	Multiple species (seabirds)	Empirical data, model output	Potential risk = spatial overlap between aggregative patterns of seabirds [coupling Taylor's power law (TPL) with linear mixed effect models] and offshore wind farm placement	Semi-quantitative measure per unit area (12.5 km²); mapping out approach risk	Yes, future offshore wind farm areas have been considered	Christel *et al.* (2013)
Small; South Florida coastal ecosystem, Gulf of Mexico	Multiple global (e.g. climate change) and local (e.g. fishing) stressors	Multiple species, multiple habitats	Expert knowledge	Impact = matrix-based analyses of pressures to states and services, scored by expert opinion	Quantitative measures for given management unit; relative impact matrices	No	Cook *et al.* (2013)
Small (28 500 km²); German EEZ, North Sea	Fisheries	Habitat (benthic)	Model output	Risk = proportion of the ecosystem component × ∑(proportion of the cell × gain function per cell (∑recovery potential over mortality potential for all impacts))	Quantitative measure per unit area; distribution of cumulative risk by area and benthic distribution	Yes, four scenarios evaluated against goals from European maritime policies (MSFD, CFP, HD)	Fock *et al.* (2011)
Small (28 500 km²); German EEZ, North Sea	Fisheries, aggregate extraction	Multiple species (benthic, mammals, seabirds)	Model output	Loss and exposure = mortality (M)/recovery (R)	Quantitative measure per unit area (3 × 3 nm/ 6 × 6 nm); risk scores by area and ecosystem function	No	Fock (2011a)
Small (256 500 and 40 km²); UK (English and Welsh) waters	Cumulative pressures (additive, antagonistic, synergistic) from fisheries and aggregate extraction	Multiple habitats (benthic)	Expert knowledge, model output	Cumulative impact = degree of disturbance from type of fishing gear, fishing intensity, habitat sensitivity and recovery rates	Quantitative measure per unit area (1 km²); cumulative impact scenario output	Yes, four cumulative effects scenarios (greatest, additive, antagonistic and synergistic) to estimate overall recovery times	Foden *et al.* (2010)

Small (256 500 km^2); UK (English and Welsh) waters	Cumulative pressures (greatest, additive, antagonistic, synergistic) from human stressors	Multiple habitats (benthic)	Expert knowledge, model output	Cumulative impact = degree of disturbance from type of pressure, pressure intensity, habitat sensitivity, and recovery rates	Quantitative measure per unit area (1 km^2); cumulative impact scenario output	Yes, four cumulative effects scenarios (greatest, additive, antagonistic, and synergistic) to estimate overall recovery times	Foden *et al.* (2011)
Small (ca. 55 500 km^2); Northern-Central Adriatic, Mediterranean Sea	Fisheries	Multiple species (functional groups)	Model output	Biomass and catch changes = amount of total biomass, commercial species biomass, predator species biomass, fish biomass, invertebrates (except plankton) biomass, total catch, demersal catch, pelagic catch) assessed using spatial – temporal foodweb model Ecospace	Quantitative measure per unit area (25 km^2); scenario output tables	Yes, scenarios regarding spatial management (MPAs), three temporal simulations of temporary closures and overall reduction of fishing effort (Ecospace)	Fouzai *et al.* (2012)
Small, Scottish waters	Cumulative pressures (additive) from human stressors	Multiple species (seabirds)	Expert knowledge, model output	Disturbance risk = (ship and helicopter traffic, habitat specialization) × conservation importance	Semi-quantitative measure for given management unit; ranked species concern scores	No	Furness and Tasker (2000)
Small (28 500 km^2); German EEZ, North Sea	Cumulative pressure (additive) from human stressors	Single species (fish)	Expert knowledge, model output	Risk = pressure to state vulnerability [severity and duration of (negative) effects (due to human pressure) + the sensitivity of species (resiliency, reversibility, sensitivity, etc.)]	Semi-quantitative measure per unit area (5 km^2); mapping out approach and scenario output	Yes, multiple risk scenarios based on the identification of potential conflict areas between drivers and between pressures and nursery grounds	Gimpel *et al.* (2013)
Small; coastal-zone of the Great Australian Bight, South Australia	Fisheries	Multiple species (mammals)	Expert knowledge, model output	Risk of extinction = population viability analysis based on time and probability of terminal extinction and quasi extinction by subpopulation, region and marine fishing areas with the greatest bycatch risk	Semi-quantitative measure per unit area (10 × 10 km nodes); risk scenario output, bycatch rates	Yes, three scenarios of increasing, stable and decreasing population trajectories	Goldsworthy and Page (2007)
Small (ca. 26 000 km^2); Great Barrier Reef, Australia	Cumulative pressure (additive) from human stressors	Habitat (seagrass)	Expert knowledge	Cumulative impact = vulnerability [frequency, functional impact, resistance, recovery time (years), and certainty]	Semi-quantitative measure per unit area (2 km^2); cumulative impact score mapping	No	Grech *et al.* (2011)
Small; Barcelona Habour, Spain	Pollution	Habitat quality (water)	Model output	Risk index = probability, exposure and vulnerability; branch-decision scheme to evaluate the cost of each decision as a function of vulnerability, proximity and toxicity of potential contaminants	Semi-quantitative measures per unit area; spatial distribution of risk	Yes, decision branch model based on cost/utility	Grifoll *et al.* (2010)

Continued

Table 3. *Continued*

Scale and location small <500 000 km² meso ≥500 000 – 106 km², large≥106 km²	Risk identification and characterization		Risk analysis			Risk evaluation	References
	Stressor(driver)/ pressure indicators used	Ecosystem components/ state indicators	Measure of sensitivity of state indicators	Measure and approach used of vulnerability/ risk/impact of ecosystem/ area	Assessment output type	Management scenario analysis (assessed: yes/no)	
Small, (125 000 km²); North Sea	Fisheries	Multiple species (benthic)	Model output	Relative ecological impacts of disturbance = degree to which production and biomass in habitats respond to trawling disturbance; sensitivity = recovery time	Semi-quantitative measures per unit area (9 km²); impact maps	Yes, five management scenarios based on modelled reduction in biomass and production	Hiddink *et al.* (2007)
Small (ca. 80 000 km²); Baltic Sea	Cumulative pressure (additive) from human stressors	Multiple habitats (benthic)	Expert knowledge	Cumulative impact = weighting of pressures to habitat specific impacts [statistical approach, thresholds based on mean ± SD of cumulative impact within habitat type] using HELCOM weighting factors	Semi-quantitative measure per unit area (71 289 m²); cumulative impact scores	No	Korpinen *et al.* (2013)
Small (1 km²); Spanish coast, local beaches, Mediterranean Sea	Multiple human and environmental stressors	Habitat quality, multiple species, ecosystem function and services	Empirical data, expert knowledge	Risk = ∑hazard intensity × ecosystem service values [habitat, disturbance regulation, water supply, recreational and aesthetic services, spiritual and historic values]	Semi-quantitative measure for given management unit; risk valuation and prioritization	No	Lozoya *et al.* (2011)
Small (20 000 km²); Brazilian coast (continental shelf area), Atlantic	Marine traffic, hydrocarbon exploration	Single species (mammals)	Empirical data, expert knowledge	Risk = humpback whale density category + anthropogenic impact category	Semi-quantitative measure per unit area (ca. 50 km radius); risk mapping and conservation prioritization	No	Martins *et al.* (2013)
Small (30 km²); Archipelago of La Maddalena (Sardinia, Italy), Mediterranean Sea	Pollution	Habitat quality (beaches)	Empirical data, expert knowledge	Risk = hazard index [normalised oil particle concentration derived from models] × vulnerability [geomorphology and environmental protection]	Semi-quantitative measure per unit area (90 km²); mapping out of hazard index	No	Olita *et al.* (2012)
Small (4 km²); Ligurian Sea MPA (Italy), Mediterranean Sea	Multiple human stressors	Multiple habitats	Expert knowledge, model output	Marine territory score (impact) = relationship between pressure intensities and ecosystem status (spatially resolved [distance of habitats from reference/unperturbed conditions (4 habitat indices)] and average over territory)	Semi-quantitative per unit area (250m²): mapping of change in marine territory status (impact)	Yes, management scenarios based on experts judgment of changes in pressure intensities used in the model	Parravicini *et al.* (2012)

Small (10 000 km^2); Bay of BiscaySpanish EEZ at the Basque Coast, Atlantic	Fisheries	Multiple species (trophic levels)	Empirical data, expert knowledge	Total fishing pressure (TFP) = cumulative fishing intensity; fishing pressure per commercially relevant species; fishing pressure by trophic level	Semi-quantitative measure per unit area (1 km^2); TFP maps	No	Pascual *et al.* (2013)
Small (1 km^2); San Foca tourist harbour (Italy), Mediterranean Sea	Pollution	Habitat quality	Expert knowledge, model output	Risk = likelihood of negative environmental changes resulting from human activities (subjective and objective expert opinions)	Semi-quantitative measure for management unit; mapping of spatially explicit risk values	No	Irene *et al.* (2010)
Small (ca. 25 000 km^2); South California, USA	Marine traffic	Multiple species (mammals)	Model output	Ship-strike risk = shipping routes [route-use overlay] in combination with whale distribution model [generalized additive model (GAM)]	Quantitative measures per unit area (4 km^2); risk scores for different shipping scenarios	Yes, spatial scenarios for (alternative) ship traffic and military use, fishing and conservation (MPAs)	Redfern *et al.* (2013)
Small (ca. 10 000 km^2); Pudget Sound, USA	Multiple human stressors	Multiple species (fish)	Empirical data	Risk = direct impacts of pressures [mortality] and resilience [fecundity, behavioural/physiological response, life history traits]; spatial overlaps between pressure and states of various ecosystem components	Semi-quantitative measure for given management units; risk maps and risk scores	No	Samhouri and Levin (2012)
Meso (500 000 km^2); UK southern, eastern and western coastal waters	Aggregate extraction	Multiple species	Empirical data, expert knowledge	Risk = vulnerability [spatial overlap and statistical test]; sensitivity index [recovery potential (e.g. ability to switch diet and reproductive strategy)]	Quantitative measure per unit area (2 × 2 nm); overlay map as vulnerability	Yes, current and future license areas have been considered	Stelzenmüller *et al.* (2010)
Small (28 500 km^2); German EEZ, North Sea	Fisheries	Single species (fish)	Empirical data	Risk = ratio of species abundance [environmental parameters (temperature, salinity and depth)] and catch in commercial fisheries using BN model	Quantitative measures per unit area (3 × 3 degrees); BN model output, vulnerability states	Yes, the impact of no-takes zones due to establishment of wind parks have been considered (changes in fishing effort distribution and temperature)	Stelzenmüller *et al.* (2011)
Small (150 000 km^2); Gulf of Finland	Nutrient loads	Habitat quality (water body)	Model output	Risk = phosphorus loads (t/year), nitrogen loads (t/year)	Quantitative measure for given management unit; mapping out approach of predicted concentrations	Yes, coupled model output using multiple scenarios	Vanhatalo *et al.* (2013)
Large; Western and Central Pacific Ocean	Fisheries	Multiple species (seabirds)	Empirical data, expert knowledge	Risk = productivity (P)/susceptibility (S) [P = Fecundity Factors index; S = product of fishing effort and normalized species distributions weighted with vulnerability of species to longline fishing gear; vulnerability = number of kills reported]; PSA analysis	Semi-quantitative measure per unit area (5 × 5 degrees); mapping out approach, summing up over all species, season and flag	No	Waugh *et al.* (2012)

Continued

Table 3. *Continued*

Scale and location small <500 000 km² meso ≥500 000–106 km², large≥106 km²	Risk identification and characterization		Risk analysis			Risk evaluation	References
	Stressor(driver)/ pressure indicators used	Ecosystem components/ state indicators	Measure of sensitivity of state indicators	Measure and approach used of vulnerability/ risk/impact of ecosystem/ area	Assessment output type	Management scenario analysis (assessed: yes/no)	
Large; Australian waters	Fisheries	Multiple habitats	Empirical data, expert knowledge	Impact = PSA (productivity = level of natural disturbance, regeneration of fauna; susceptibility = availability, encounterability, selectivity)	Semi-quantitative measure for given management unit (30 or 60 nm); risk category per habitat	No	Williams *et al.* (2011)
Small (3800 km²); Rhode Island	Offshore wind farms	Multiple species	Empirical data, expert knowledge	Impact = concern index [sensitivity to displacement, weighting of species by species] to predict areas with high conservation priority in relation to their distribution (surface area)	Quantitative measure per unit area (2 km²); scenario output, mapping of vulnerability	Yes, Zonation software [Moilanen (2013)]	Winiarski *et al.* (2014)

Studies were reviewed according to the spatial scale and the methods used with regard to the three steps of a risk assessment: risk identification, risk analysis, and risk evaluation.

the occurrence probability and magnitude of a stressor or pressure. However, the concepts of resistance and resilience of ecosystem components were only considered in a few studies. The dominating type of assessment outputs (13 studies) have been maps with 'semi-quantitative measures per unit areas' (from 250 m^2 to 90 km^2), followed by 'quantitative measures per unit area' (from 400 m to 100 km^2) in 12 studies, only a small proportion of the assessment outputs related to quantitative (two studies) or semi-quantitative (five studies) measures for given management units (thus one value for a case study area). More than half of the reviewed studies carried out a risk evaluation and tested a broad range of scenarios including simulated pressure–effect scenarios, mostly related to the future license areas of wind farms or fisheries management measures. Cumulative effect scenarios have been tested by weighting for instance the relationship between indicators and pressures. It is relevant to allude to the fact that about one-third of the studies did not account for uncertainty. Some studies assessed uncertainty quantitatively based on model uncertainty. Other studies addressed uncertainty in a qualitative way, mainly by a discussion about the issue of uncertainty and/or proposed methods for further analysis.

Case study

Fleet-specific trawling frequencies show clear spatial patterns, and as an example we illustrated the spatial distribution of the mean trawling frequency of the international beam trawl fleet with 80 mm mesh size and >221 kW overlaid with the current (2013) OWD application areas in Figure 4. The mean overall local mortality rate assuming an equal impact of all fishing fleets is displayed in Figure 5 (top), where high values can be found in the North–East of the study area and along a coastal strip. The relative combined recovery rates of the benthic communities are fishery independent and therefore patterns resembled the benthic communities (Figure 5, bottom). Spatial predictions of *DI* revealed that 5.3% of the total area showed values >1, indicating a higher rate of mortality than recover, whereas 0.74 was the maximum value estimated for the weighted disturbance indicator (DI_w). High values of the unweighted and weighted disturbance indicator were found in different places (Figure 6). This is because for DI_w the beam trawl fleets using nets with >800 mm mesh size (Beam80lrg and Beam80sml) were given by far the highest impact weights. For each BN node

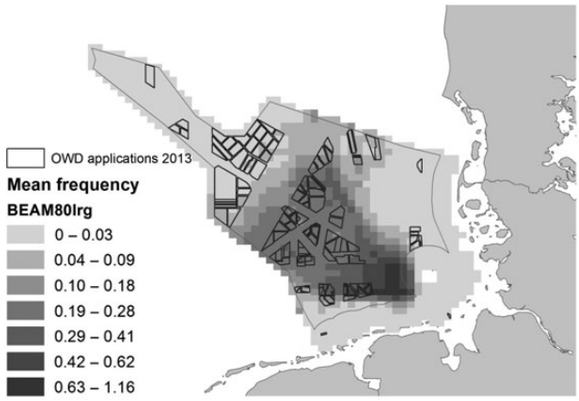

Figure 4. Mean (2005–2008) frequency of a unit area (3 × 3 nm) being reworked by the international beam trawl fleet with a mesh size of 80 mm and >221 kW derived from VMS data (Beam80lrg) and additionally overlaid with the current (2013) OWD application areas.

that represents a continuous variable the weighed mean (the mean value weighted by the probability of occurrence) with its Gaussian standard deviation is shown on the bottom of each node

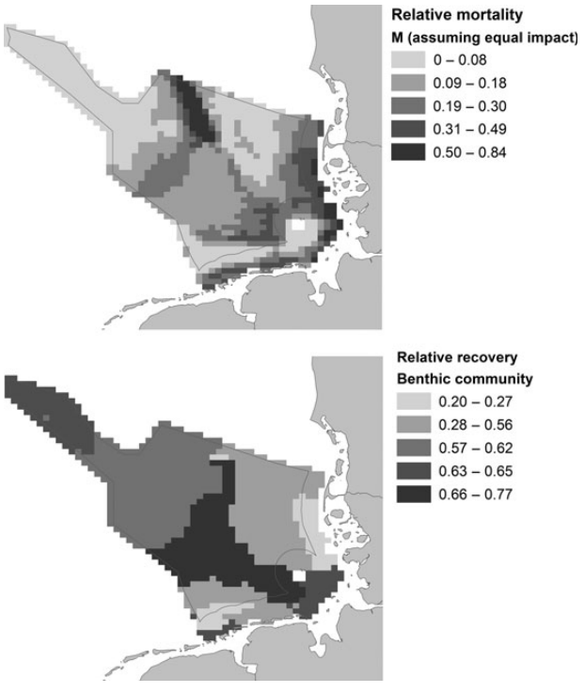

Figure 5. Top: Relative overall local mortality rate (M) ($is_k = 1$) based on the distribution of the mean fishing frequency by the respective fleets; bottom: distribution of the estimated relative recovery rates derived from the combined recovery time and recover frequency of the prevailing benthic communities (see Table 1).

(Figure 3). For instance, the weighted mean state value for large beam trawl frequencies is 0.221 ± 0.34 indicating a high level of variance. The trained BN displays the "business as usual scenario" using the fishing effort patterns from 2005 to 2008, from which it was derived that 34.5% of the total area showed the highest level of trawling frequencies (state 3: 0.06 and 1.16, Figure 3). An alternative interpretation of the probabilities associated with the respective node states is that there is a 34.5% chance to find a value between 0.06 and 1.16 within any given unit area (vector grid cell). The baseline BN showed further that there is a 4.12% chance to find values of $DI > 1$ within any given unit area. In contrast, there is only a 1.35% chance to find values of $DI_w > 1$ within any given unit area. The sensitivity analysis of the disturbance indicator node (DI) showed that the latter was most influenced by the findings for mortality (node M; variance reduction = 22.5%), recovery (node R; variance reduction = 13.8%), combined recover frequency (variance reduction = 10.9%), and type of benthic community (variance reduction = 10.3%), while all other nodes resulted in a variance reduction <2%. The classification success rate (spherical payoff) which ranges from 0 to 1, with 1 being the best model performance, indicated a relative accuracy of the BN model for predicting the disturbance indicator (DI) with a value of 0.87 and a value of 0.95 for predicting DI_w, respectively.

The effects of the planned OWD sites on the two measures of benthic disturbance (DI and DI_w) were explored stepwise (Figure 7a and b). Figure 7a shows that the new prior distribution of the Beam80lrg node (corresponding to the spatial relocation of 15% of the fishing activities) resulted in an average likely value of 0.31 for DI along with a standard error of 0.42. Compared with the "business as usual" scenario the predicted probabilities of the DI states only altered ~1%. In contrast, using the same scenario the average likely value of DI_w increased from 0.235 ($\pm$0.27) to 0.261 ($\pm$0.29). However, this increase was not significant due to the great variance in estimates. The additional modification of

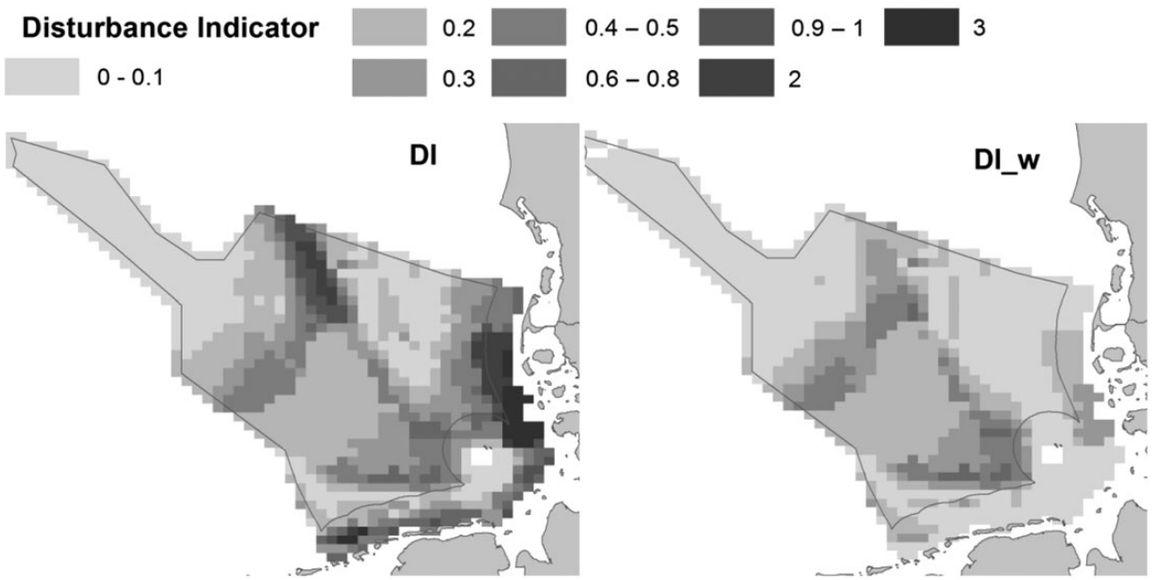

Figure 6. Left: Estimated values of the disturbance indicator (DI) based on an overall local mortality rate with equal weight for the impact scores of the six fishing fleets; right: estimated values of the disturbance indicator (DI_w) based on an overall local mortality rate with different weights for the impact scores of the six fishing fleets ($is_{BEAM80lrg} = 1$; $is_{BEAM80sml} = 1$; $is_{BEAM1631lrg} = 0.1$; $is_{BEAM1631sml} = 0.1$; $is_{OTTER80lrg} = 0.15$; $is_{OTTER80sml} = 0.15$).

the prior distribution of the Beam1631sml node and the predicted probabilities of benthic disturbance states are displayed in Figure 7b. The model predicted an average likely value of 0.309 for DI ($\pm$0.42), while the average likely value for DI_W remained the same. However, for this case study, where the BN is populated with spatial data, the likely values of the disturbance indicator averaged over the entire study area of minor importance (as indicated by the high standard error). Here, the predicted likelihood of an area proportion having a certain value is much more relevant to evaluate trade-offs of spatial management scenarios. Whereas the assumed redistribution scenario of both fleets showed no significant effect on the four DI states, overall changes were predicted in relation to the probability distributions of $DI_{w.}$ states. The estimated probabilities of DI_w values >1 ranged between 1.35% (business as usual scenario) and 1.63% (full displacement scenario). This means that 1.63% of the study area (or 1.63% of all vector grid cells) will experience DI_w values >1 using the current fishing effort displacement scenario. More relevant changes to the predicted probabilities were observed for the DI_w States 1 and 2. Compared with the baseline scenario the predicted probabilities of the DI_w State 1 decreased ~8% (from 79.3 to 71.9%), while the probabilities of DI_w State 2 increased ~6% (from 16 to 22.1%). This means that 8% of the area (8% of the vector grid cells) will likely face a worsening of DI_w values compared with the current state. This is consequently related with an increased probability (by 6%) for any given unit area to have a DI_w value ranging from 0.3 to 0.5. Thus, the here considered MSP measures and the related fishing effort displacement scenario would not fulfil the defined overall operational management objective ("The average relative vulnerability of benthic communities to fishing should not deteriorate with respect to current levels"), since the predicted probability distributions of the DI_w values showed deteriorating values compared with the current state.

Discussion

Current ERA approaches and gaps in a spatial management context

We used the steps of a risk assessment framework described by Cormier *et al.* (2013) to frame the assessment of a fair number of spatially explicit and quantitative ERAs concerned with spatial management questions. There are, of course, other established risk assessment frameworks such as a productivity–susceptibility analysis (PSA) a semi-quantitative ERA methodology (Waugh *et al.*, 2012) or the conceptual driver-pressure-state-impact-response framework which illustrates cause–effect pathways (Elliott, 2002). Further bow tie diagrams describe and analyse risk events by visualizing relevant pathways from causes to consequences (Ferdous *et al.*, 2013). The bow tie diagram focuses on so-called barriers representing existing control or mitigation measures that are placed between the causes and the risk, and the risk and consequences. These diagrams can also be adapted to the DIPSR framework. Recently, BNs have been used in combination with bow tie diagrams to overcome their purely depictive capabilities by adding probabilities and conditional dependencies between components (Badreddine and Amor, 2013; Khakzad *et al.*, 2013).

The here identified methodological shortcomings were based on a structured, but not exhaustive selection of studies. Nevertheless, this selection was a result of a literature database search (Scopus) using defined keywords, context, and expected type of output. Review results showed that independently from the investigated ecosystem components, computing quantitative measures of sensitivity is still challenging and could hardly be derived from empirical data alone. Often a combination of model outputs and expert knowledge seemed to deliver the preferred metric (e.g. Foden *et al.*, 2011). Thus, our findings emphasized the lack of empirical studies to support extrapolation of measures of sensitivity to system scale questions (see discussion in Crain *et al.*, 2008). Another identified weakness was the lack of an explicit assessment of uncertainty, especially in cases where expert judgements were used. Uncertainty cannot be eliminated from any integrated assessment or model-based decision support; however, it should be recognized and constructively handled (Rotmans and van Asselt, 2001; Astles *et al.*, 2006). Thus, the assessment of uncertainty is an important prerequisite of the herein described steps of risk analysis and subsequent risk evaluation. For instance, fuzzy sets and advice theory allow for characterization of uncertainty associated with expert knowledge (Ferdous *et al.*, 2013). Also Walker-type and pedigree matrices were utilized to assess both the sources and respective relative levels of uncertainty related to an assessment process which integrates many sources of information and data qualities (Stelzenmüller *et al.*, 2015).

Despite the great variation of terminology across studies the minimum measure of vulnerability involved always was a combination of a measure of sensitivity of an ecosystem component and the probability and magnitude of a stressor occurring. However, only a few studies computed vulnerability according to the best practices defined in De Lange *et al.* (2010), which require the consideration of resistance and resilience when defining sensitivity and vulnerability, respectively. This depicts a future need to root spatially explicit quantitative ERAs more in ecological theory with regard to system function and processes (e.g. Fock *et al.*, 2011).

Scenario evaluation is deemed as an important step in the risk assessment framework and which has been carried out in roughly half of the reviewed studies. Those who did simulate management scenarios generally used spatially explicit tools and approaches such as Ecospace (Fouzai *et al.*, 2012), Zonation (Moilanen, 2013; Winiarski *et al.*, 2014), or a combination of GIS and BN models (Stelzenmüller *et al.*, 2011) to allow for a non-static assessment of cause–effect pathways.

Surprisingly, only one of the studies, included in this review, exploited a process-based numerical model to predict ecosystem responses to natural or human pressures (Vanhatalo *et al.*, 2013). Process-based models represent physical processes and typically include forcing by waves and/or currents, a response in terms of sediment transport and a morphology-updating module. Routinely used for reconstructions of past conditions or to forecast possible future trends, such models are useful in the context of risk assessments (Weisse *et al.*, 2009), in particularly, when the simulations cover a wide range of natural variability. Building on hydrodynamic drift simulations, Chrastansky and Callies (2011) have demonstrated how such model data can be turned into spatially explicit information on the risk posed by hypothetical oil spills in the North Sea. Their approach based on a BN, which makes the essential information of the model available without the need to access the memory-intensive, original datasets. In that way, detailed information on key natural drivers and their causal relationships with existing pressures can easily be considered in a wider GIS-coupled risk assessment framework. Until now, this is rarely the case in ERAs making it difficult (if not impossible) to separate the effects of natural disturbance, for example by waves, from that caused by human activities such as bottom trawling (Diesing *et al.*, 2013).

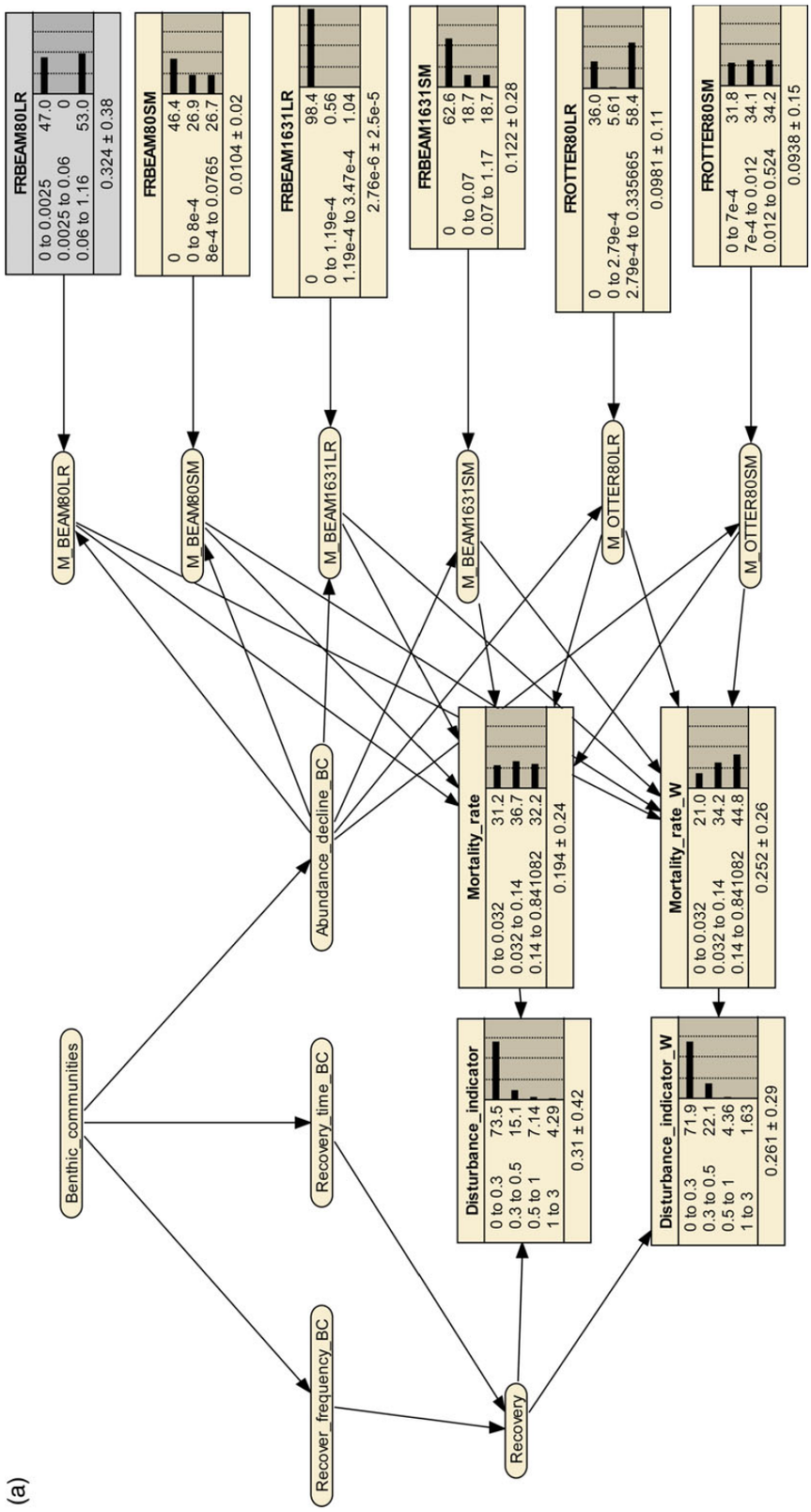

Figure 7. (a, b) Results of the inference the Bayesian belief network model applying the spatial management scenario "What are the likely impacts of spatial shifts of 15% of the total fishing frequency of large beam trawlers (Beam80lrg) and 3% of the small beam trawlers (Beam1631sml) on local disturbance rates (assuming equal and weighted impacts of the different fishing fleets)". Predicted probabilities (%) are shown for all states of the relevant model nodes.

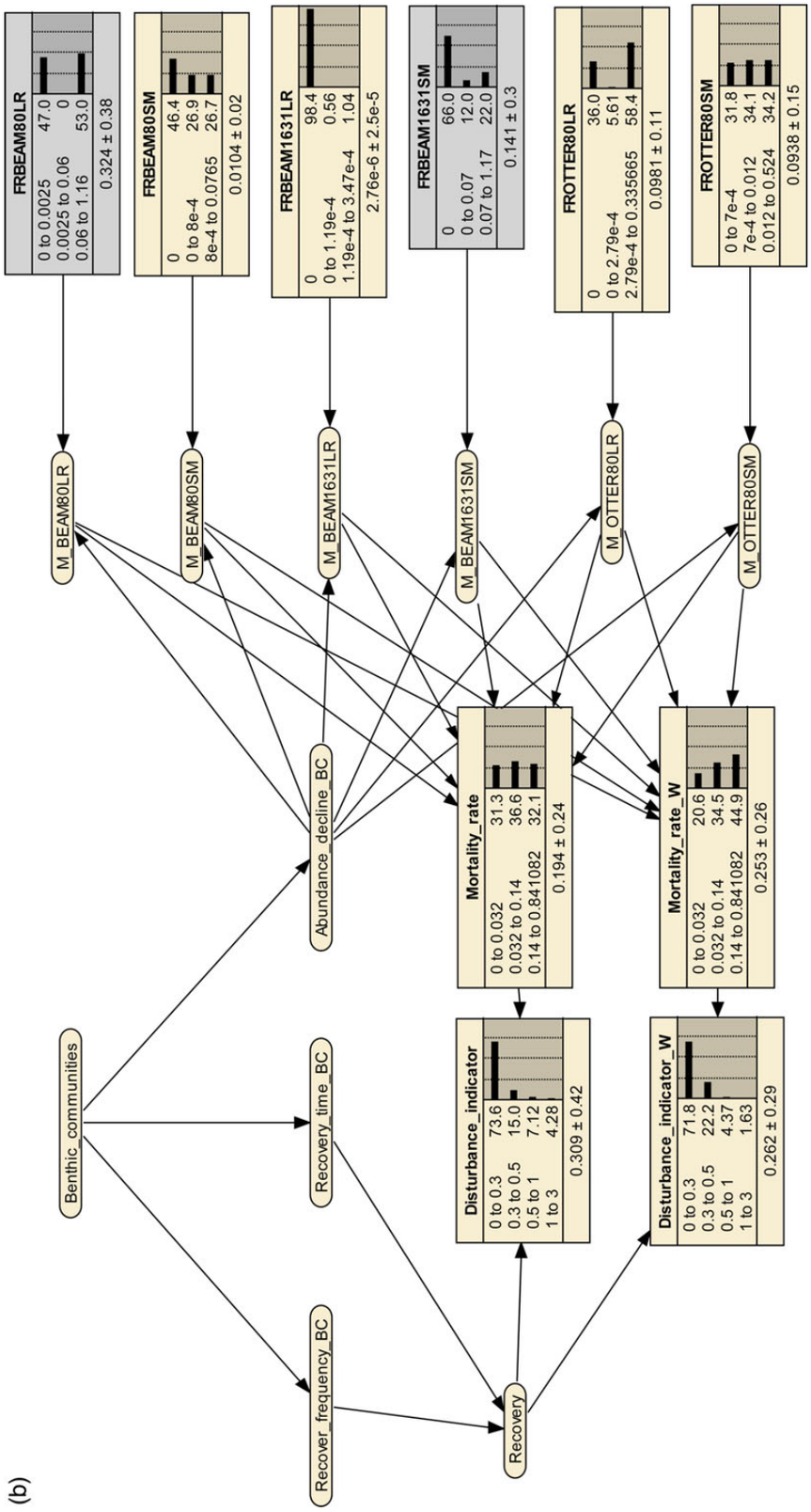

Figure 7. Continued.

According to ecological theory (Pickett and White, 1985), disturbance regime is, however, an important spatial process which should be accounted for when assessing the risks of spatial management scenarios.

Perspectives for assessing the trade-offs of MSP measures in the German EEZ of the North Sea

The aim of the case study was to address some of the methodological shortcomings identified in the current literature on spatially explicit and quantitative ERAs and to provide some perspectives for assessing the trade-offs of on MSP measures in the German EEZ of the North Sea.

We built on a study by Fock (2011a) for calculating measures of fishing frequency, mortality rates and the disturbance indicators. The overall measures of recovery and mortality have been computed for 10 benthic communities (Pesch *et al.*, 2008). For this, we converted existing model outputs on recovery and mortality rates by sediment type to respective rates by benthic community. This has been done by weighting sediment-specific parameters with likely species habitat preferences given in Rachor and Nehmer (2003).

As a consequence, those benthic community-specific estimates on mortality and recovery rates reflect rather rough estimates of those parameters. A promising alternative source for recovery rates (days) by phyla and habitat type provides a meta-analysis of trawl impact studies carried out by Kaiser *et al.* (2006). In future studies, those results could be used to redefine for instance fleet-specific impact scores (is_{fleet}) of the weighted mortality rates. Further, benthic disturbance was only calculated for infaunal benthic communities, while epifaunal species may be more vulnerable to fishing disturbance (Piet *et al.*, 2000). Empirical data for instance revealed longer recovery times of benthic epifaunal communities (7–8 years) compared with infauna communities (2–5 years) in the German Bight (at least after the impact of cold winters) (Neumann and Kröncke, 2011). As a result, future steps to improve mortality and recovery rates of benthic communities would embrace the combination of infaunal and epifaunal recovery and decline rates.

In our case study, we did not explicitly map or consider a measure of natural disturbance; however, we can assume that natural disturbance, for example, by tidal and wave stress as well as daily and seasonal temperature variability, is highest in shallow coastal areas (Becker *et al.*, 1992; Neumann *et al.*, 2013). Here, benthic communities will show greater resilience to fishing disturbance than in zones with larger water depths (e.g. Hiddink *et al.*, 2006b). Further Elliott and Quintino (2007) argued that communities in stressed environments are well adapted to natural stress and will probably never show a recovery to "undisturbed" communities. Thus, taking interactions between fishing and natural disturbances into account would very likely result in different patterns of the disturbance indicator. Nevertheless, Fock *et al.* (2011) suggested that observed recovery rates incorporate indirectly local effects of natural disturbance. Addressing a similar topic Diesing *et al.* (2013) investigated the impact of demersal fishing on seabed integrity in the greater North Sea and proposed a method to incorporate natural and fishing disturbance in a spatially explicit study. They defined trawling impact as significant when it exceeds natural disturbance (by waves and tides). The resulting indicator was expressed as a probability on a 12 × 12 nm grid and could as such be rescaled and incorporated into our risk assessment approach.

The observed differences in spatial pattern of the two disturbance indicators were clearly a result of the weighting of the impact of the different fishing fleets. Hence, DI and DI_w describe a range of likely outcomes of disturbance modelling with DI_w as lower and DI as upper bound. In this sense, it reflects a transparent assessment of uncertainty.

To enable a dynamic link of risk analysis and risk evaluation, hence scenario evaluation, we combined GIS with a BN model to conduct a quantitative spatially explicit risk assessment. For the integration of BNs and GIS, we followed in general the good practice described in Johnson *et al.* (2012). BNs indeed are advantageous, especially when considering the input from various data types (Aguilera *et al.*, 2011), but model construction often is challenging and nontrivial (Kjræulff and Madsen, 2013). BNs represent multidimensional distributions and can conveniently be applied for updating probability distributions of all variables given observations for just a subset of them. Information available will propagate across the whole network regardless of the orientation of edges (see, e.g. Kjræulff and Madsen, 2013). This analysis of joint probabilities based on incomplete observations must be distinguished, however, from predicting the results of external interventions (e.g. scenario assessment). For the latter purpose, a BN must be formulated in line with causal relationships (see Pearl, 2000). According to Kjræulff and Madsen, (2013), a BN is a probabilistic network for reasoning under uncertainty, whereas an influence diagram is a probabilistic network for reasoning about decision-making under uncertainty. Thus, an influence diagram represents parameters actively controlled by rational decision-makers as non-random decision nodes. They rate system configurations that result from management decisions based on value or utility nodes (Pearl, 1988; Bedford and Cooke, 2001). In our example, we did not construct an influence diagram with decision nodes. Further multistage decision networks allow even for considering a sequence of decisions at future points in time when certain types of information will become available. Such repeated decision-making is an essential part of an adaptive management process (Vugteveen *et al.*, 2014). A representation of such practically relevant concepts in a probabilistic framework such as the one illustrated here, however, is scientifically challenging and requires future development.

Our spatial management scenario simulated a general spatial shift of fishing effort from medium fished areas to low and highly fished areas due to the development of offshore renewables in areas where 15 and 3% of the total average beam trawl effort took place. This was based on the assumption that vessels conducting demersal mixed or crustacean fishery reallocate their effort in areas of potential large catch or previous knowledge and experience (Bastardie *et al.*, 2013a). Results showed that the assumed shift in fishing frequencies did not result in significant changes of the average likely value of the disturbance indicator. However, disturbance indicators (assuming unequal impact) still worsen in ~8% of the study area. This information is much more meaningful when evaluating the trade-offs of spatial management options. Once, more realistic fishing effort displacement scenarios become available, the combined GIS and BN approach can be used to predict likely local values of, for example, the disturbance indicator. For instance, individual-based models, predicting fishing fleet behaviour under changing economic or ecological conditions (Bastardie *et al.*, 2013b), would allow entering specific findings for prior distributions of fishing frequencies of specific fleets.

Conclusion

Currently, quantitative ERA studies in a spatial management context reflect a wide range of assessment approaches, with

varying interpretations of the terms risk, vulnerability, or impact. Especially, the different definitions of vulnerability suggest that future spatially explicit quantitative ERAs should be more rooted in ecological theory with regard to system function and processes. Spatially explicit risk assessments yet to come should also consider the inclusion of numerical models for instance describing natural disturbance, since this is an important component in ecological disturbance theory. We identified a transparent assessment of uncertainty as clear shortcoming of many current approaches and conclude that the application of BNs are a promising approach to address this. Also future research is needed on how to build meaningful influence diagrams, with parameters actively controlled by rational decision-makers (decision nodes), in the course of quantitative ERAs. Independently from the concepts and methods applied to predict a measure of risk, we strongly recommend putting caution on the type of output produced and its potential uptake in an actual spatial management process. The latter often refers to complex multiple objectives settings, where the impacts of many human activities need to be jointly assessed. In conclusion, marine spatial management or MSP processes should embed ERA frameworks which allow for the integration of multiple risk assessments and the quantification of related uncertainties at a common spatial scale.

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References

92/43/EEC, C. D. 1992. Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora.

Aguilera, P. A., Fernández, A., Fernández, R., Rumí, R., and Salmerón, A. 2011. Bayesian networks in environmental modelling. Environmental Modelling and Software, 26: 1376–1388.

Alvarez-Romero, J. G., Devlin, M., Teixeira da Silva, E., Petus, C., Ban, N. C., Pressey, R. L., Kool, J., *et al.* 2013. A novel approach to model exposure of coastal-marine ecosystems to riverine flood plumes based on remote sensing techniques. Journal of Environmental Management, 119: 194–207.

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.

Badreddine, A., and Amor, N. B. 2013. A Bayesian approach to construct bow tie diagrams for risk evaluation. Process Safety and Environmental Protection, 91: 159–171.

Ban, N. C., Alidina, H. M., and Ardron, J. A. 2010. Cumulative impact mapping: advances, relevance and limitations to marine management and conservation, using Canada's Pacific waters as a case study. Marine Policy, 34: 876–886.

Bastardie, F., Nielsen, J. R., Andersen, B. S., and Eigaard, O. R. 2013a. Integrating individual trip planning in energy efficiency – Building decision tree models for Danish fisheries. Fisheries Research, 143: 119–130.

Bastardie, F., Nielsen, J. R., and Miethe, T. 2013b. DISPLACE: a dynamic, individual-based model for spatial fishing planning and effort displacement—integrating underlying fish population models. Canadian Journal of Fisheries and Aquatic Sciences, 71: 366–386.

Becker, G. A., Dick, S., and Dippner, J. W. 1992. Hydrography of the German bight. Marine Ecology Progress Series, 91: 9–18.

Bedford, T., and Cooke, R. 2001. Probabilistic Risk Analysis: Foundations and Methods. Cambridge University Press, Cambridge, UK.

Bianchi, C. N., Parravicini, V., Montefalcone, M., Rovere, A., and Morri, C. 2012. The challenge of managing marine biodiversity: a practical toolkit for a cartographic, territorial approach. Diversity, 4: 419–452.

BMVBS. 2009. Spatial Plan for the German Exclusive Economic Zone in the North Sea. http://www.bsh.de/en/Marine_uses/Spatial_Planning_in_the_German_EEZ/index.jsp (last accessed 12 November 2014).

Brennan, J., Fitzsimmons, C., Gray, T., and Raggatt, L. 2014. EU marine strategy framework directive (MSFD) and marine spatial planning (MSP): Which is the more dominant and practicable contributor to maritime policy in the UK? Marine Policy, 43: 359–366.

Brown, C. J., Saunders, M. I., Possingham, H. P., and Richardson, A. J. 2013. Interactions between global and local stressors of ecosystems determine management effectiveness in cumulative impact mapping. Diversity and Distributions, 20: 538–546.

Carneiro, G. 2013. Evaluation of marine spatial planning. Marine Policy, 37: 214–229.

Chrastansky, A., and Callies, U. 2011. Using a Bayesian network to summarize variability in numerical long-term simulations of a meteorological-marine system: drift climatology of assumed oil spills in the North Sea. Environmental Modeling and Assessment, 16: 1–14.

Christel, I., Certain, G., Cama, A., Vieites, D. R., and Ferrer, X. 2013. Seabird aggregative patterns: A new tool for offshore wind energy risk assessment. Marine Pollution Bulletin, 66: 84–91.

Collie, J. S., Vic Adamowicz, W. L., Beck, M. W., Craig, B., Essington, T. E., Fluharty, D., Rice, J., *et al.* 2013. Marine spatial planning in practice. Estuarine, Coastal and Shelf Science, 117: 1–11.

Commission, E. 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, L164: 19–40.

Commission, E. 2014. Directive 2014/89/EU of the European Parliament and of the Council of 23 July 2014 establishing a framework for maritime spatial planning. In /89/EU, pp. 135–145. Ed. by E. P. a. t. C. o. t. E. Union.

Cook, G. S., Fletcher, P. J., and Kelble, C. R. 2013. Towards marine ecosystem based management in South Florida: investigating the connections among ecosystem pressures, states, and services in a complex coastal system. Ecological Indicators, 44: 26–39.

Cormier, R., *et al.* 2013. Marine and Coastal Ecosystem-Based Risk Management Handbook. ICES Cooperative Research Report No. 317. 60 pp.

Crain, C. M., Kroeker, K., and Halpern, B. S. 2008. Interactive and cumulative effects of multiple human stressors in marine systems. Ecology Letters, 11: 1304–1315.

De Lange, H. J., Sala, S., Vighi, M., and Faber, J. H. 2010. Ecological vulnerability in risk assessment – a review and perspectives. Science of the Total Environment, 408: 3871–3879.

Diesing, M., Stephens, D., and Aldridge, J. 2013. A proposed method for assessing the extent of the seabed significantly affected by demersal fishing in the Greater North Sea. ICES Journal of Marine Science, 70: 1085–1096.

Douvere, F., and Ehler, C. N. 2010. The importance of monitoring and evaluation in adaptive maritime spatial planning. Journal of Coast Conservation, 15: 305–311.

Eastwood, P. D., Mills, C. M., Aldridge, J. N., Houghton, C. A., and Rogers, S. I. 2007. Human activities in UK offshore waters: an assessment of direct, physical pressure on the seabed. ICES Journal of Marine Science, 64: 453–463.

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.

Elliott, M., and Quintino, V. 2007. The estuarine quality paradox, environmental homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. Marine Pollution Bulletin, 54: 640–645.

Ferdous, R., Khan, F., Sadiq, R., Amyotte, P., and Veitch, B. 2013. Analyzing system safety and risks under uncertainty using a bow-tie diagram: An innovative approach. Process Safety and Environmental Protection, 91: 1–18.

Fock, H. 2011a. Integrating multiple pressures at different spatial and temporal scales: a concept for relative ecological risk assessment in the European marine environment. Human and Ecological Risk Assessment, 17: 187–211.

Fock, H. O. 2011b. Natura 2000 and the European Common Fisheries Policy. Marine Policy, 35: 181–188.

Fock, H. O., Kloppmann, M., and Stelzenmüller, V. 2011. Linking marine fisheries to environmental objectives: a case study on seafloor integrity under European maritime policies. Environmental Science and Policy, 14: 289–300.

Foden, J., Rogers, S. I., and Jones, A. P. 2010. Recovery of UK seabed habitats from benthic fishing and aggregate extraction-Towards a cumulative impact assessment. Marine Ecology Progress Series, 411: 259–270.

Foden, J., Rogers, S. I., and Jones, A. P. 2011. Human pressures on UK seabed habitats: a cumulative impact assessment. Marine Ecology Progress Series, 428: 33–47.

Foley, M. M., Armsby, M. H., Prahler, E. E., Caldwell, M. R., Erickson, A. L., Kittinger, J. N., Crowder, L. B., *et al.* 2013. Improving ocean management through the use of ecological principles and integrated ecosystem assessments. BioScience, 63: 619–631.

Fouzai, N., Coll, M., Palomera, I., Santojanni, A., Arneri, E., and Christensen, V. 2012. Fishing management scenarios to rebuild exploited resources and ecosystems of the Northern-Central Adriatic (Mediterranean Sea). Journal of Marine Systems, 102–104: 39–51.

Furness, R. W., and Tasker, M. L. 2000. Seabird-fishery interactions: Quantifying the sensitivity of seabirds to reductions in sandeel abundance, and identification of key areas for sensitive seabirds in the North Sea. Marine Ecology Progress Series, 202: 253–264.

Gimpel, A., Stelzenmüller, V., Cormier, R., Floeter, J., and Temming, A. 2013. A spatially explicit risk approach to support marine spatial planning in the German EEZ. Marine Environmental Research, 86: 56–69.

Goldsworthy, S. D., and Page, B. 2007. A risk-assessment approach to evaluating the significance of seal bycatch in two Australian fisheries. Biological Conservation, 139: 269–285.

Grech, A., Coles, R., and Marsh, H. 2011. A broad-scale assessment of the risk to coastal seagrasses from cumulative threats. Marine Policy, 35: 560–567.

Grêt-Regamey, A., Brunner, S. H., Altwegg, J., and Bebi, P. 2013a. Facing uncertainty in ecosystem services-based resource management. Journal of Environmental Management, 127: S145–S154.

Grêt-Regamey, A., Brunner, S. H., Altwegg, J., Christen, M., and Bebi, P. 2013b. Integrating expert knowledge into mapping ecosystem services tradeoffs for sustainable forest management. Ecology and Society, 18: art. no. 34.

Grifoll, M., Jordà, G., Borja, Á., and Espino, M. 2010. A new risk assessment method for water quality degradation in harbour domains, using hydrodynamic models. Marine Pollution Bulletin, 60: 69–78.

Halpern, B. S., McLeod, K. L., Rosenberg, A. A., and Crowder, L. B. 2008a. Managing for cumulative impacts in ecosystem-based management through ocean zoning. Ocean and Coastal Management, 51: 203–211.

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

Hiddink, J. G., Jennings, S., and Kaiser, M. J. 2006a. Indicators of the ecological impact of bottom-trawl disturbance on seabed communities. Ecosystems, 9: 1190–1199.

Hiddink, J. G., Jennings, S., and Kaiser, M. J. 2007. Assessing and predicting the relative ecological impacts of disturbance on habitats with different sensitivities. Journal of Applied Ecology, 44: 405–413.

Hiddink, J. G., Jennings, S., Kaiser, M. J., Queirós, A. M., Duplisea, D. E., and Piet, G. J. 2006b. Cumulative impacts of seabed trawl disturbance on benthic biomass, production, and species richness in different habitats. Canadian Journal of Fisheries and Aquatic Sciences, 63: 721–736.

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

Irene, P., Paolo, V., Donatella, V., Alberto, M. J., Mauro, F., and Giovanni, Z. 2010. Mapping the environmental risk of a tourist harbor in order to foster environmental security: Objective vs. subjective assessments. Marine Pollution Bulletin, 60: 1051–1058.

Johnson, S., Low-Choy, S., and Mengersen, K. 2012. Integrating bayesian networks and geographic information systems: Good practice examples. Integrated Environmental Assessment and Management, 8: 473–479.

Kaiser, M. J., Clarke, K. R., Hinz, H., Austen, M. C. V., Somerfield, P. J., and Karakassis, I. 2006. Global analysis of response and recovery of benthic biota to fishing. Marine Ecology Progress Series, 311: 1–14.

Katsanevakis, S., Stelzenmüller, V., South, A., Sørensen, T. K., Jones, P. J. S., Kerr, S., Badalamenti, F., *et al.* 2011. Ecosystem-based marine spatial management: review of concepts, policies, tools, and critical issues. Ocean and Coastal Management, 54: 807–820.

Khakzad, N., Khan, F., and Amyotte, P. 2013. Dynamic safety analysis of process systems by mapping bow-tie into Bayesian network. Process Safety and Environmental Protection, 91: 46–53.

Kjræulff, U. B., and Madsen, A. L. 2013. Bayesian Networks and Influence Diagrams: A Guide to Construction and Analysis: A Guide to Construction and Analysis. Springer, New York.

Korpinen, S., Meidinger, M., and Laamanen, M. 2013. Cumulative impacts on seabed habitats: An indicator for assessments of good environmental status. Marine Pollution Bulletin, 74: 311–319.

Lozoya, J. P., Sardá, R., and Jiménez, J. A. 2011. A methodological framework for multi-hazard risk assessment in beaches. Environmental Science & Policy, 14: 685–696.

MacDonald, L. 2000. Evaluating and managing cumulative effects: process and constraints. Environmental Management, 26: 299–315.

Marcot, B. G., Steventon, J. D., Sutherland, G. D., and McCann, R. K. 2006. Guidelines for developing and updating Bayesian belief networks applied to ecological modeling and conservation. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere, 36: 3063–3074.

Martins, C. C. A., Andriolo, A., Engel, M. H., Kinas, P. G., and Saito, C. H. 2013. Identifying priority areas for humpback whale conservation at Eastern Brazilian Coast. Ocean and Coastal Management, 75: 63–71.

McCann, R. K., Marcot, B. G., and Ellis, R. 2006. Bayesian belief networks: Applications in ecology and natural resource management. Canadian Journal of Forest Research, 36: 3053–3062.

Moilanen, A. 2013. Planning impact avoidance and biodiversity offsetting using software for spatial conservation prioritisation. Wildlife Research, 40: 153–162.

Neumann, H., and Kröncke, I. 2011. The effect of temperature variability on ecological functioning of epifauna in the German Bight. Marine Ecology, 32: 49–57.

Neumann, H., Reiss, H., Ehrich, S., Sell, A., Panten, K., Kloppmann, M., Wilhelms, I., *et al.* 2013. Benthos and demersal fish habitats in the German Exclusive Economic Zone (EEZ) of the North Sea. Helgoland Marine Research, 67: 445–459.

Olita, A., Cucco, A., Simeone, S., Ribotti, A., Fazioli, L., Sorgente, B., and Sorgente, R. 2012. Oil spill hazard and risk assessment for the shorelines of a Mediterranean coastal archipelago. Ocean & Coastal Management, 57: 44–52.

Parravicini, V., Rovere, A., Vassallo, P., Micheli, F., Montefalcone, M., Morri, C., Paoli, C., *et al.* 2012. Understanding relationships between conflicting human uses and coastal ecosystems status: a geospatial modeling approach. Ecological Indicators, 19: 253–263.

Pascual, M., Borja, A., Galparsoro, I., Ruiz, J., Mugerza, E., Quincoces, I., Murillas, A., *et al.* 2013. Total fishing pressure produced by artisanal fisheries, from a Marine Spatial Planning perspective: a case study from the Basque Country (Bay of Biscay). Fisheries Research, 147: 240–252.

Pearl, J. 1988. Probabilistic Reasoning in Intelligent Systems: Networks of Plausible Inference. Morgan Kaufmann, Publishers Inc. San Francisco, USA.

Pearl, J. 2000. Causality: Models, Reasoning and Inference. Cambridge University Press New York, NY, USA.

Pesch, R., Pehlke, H., Jerosch, K., Schröder, W., and Schlüter, M. 2008. Using decision trees to predict benthic communities within and near the German Exclusive Economic Zone (EEZ) of the North Sea. Environmental Monitoring and Assessment, 136: 313–325.

Pickett, S. T. A., White, P. S. 1985. Patch dynamics: a synthesis. *In* The Ecology of Natural Disturbance and Patch Dynamics, pp. 371–384. Ed. by S. T. A. P. a. P. S. White. Academic Press, New York.

Piet, G. J., Rijnsdorp, A. D., Bergman, M. J. N., Van Santbrink, J. W., Craeymeersch, J., and Buijs, J. 2000. A quantitative evaluation of the impact of beam trawling on benthic fauna in the southern North Sea. ICES Journal of Marine Science, 57: 1332–1339.

Qiu, W., and Jones, P. J. S. 2013. The emerging policy landscape for marine spatial planning in Europe. Marine Policy, 39: 182–190.

Rachor, E., and Nehmer, P. 2003. Erfassung und Bewertung ökologisch wertvoller Lebensräume in der Nordsee. Alfred Wegener Institute for Polar and Marine Research, Bremerhaven.

Redfern, J. V., McKenna, M. F., Moore, T. J., Calambokidis, J., Deangelis, M. L., Becker, E. A., Barlow, J., *et al.* 2013. Assessing the risk of ships striking large whales in marine spatial planning. Conservation Biology, 27: 292–302.

Rotmans, J., and van Asselt, M. B. 2001. Uncertainty management in integrated assessment modeling: towards a pluralistic approach. Environmental Monitoring and Assessment, 69: 101–130.

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

Spiegelhalter, D. J., and Dawid, P. 1993. Bayesian analysis in expert systems. Statistical Science, 8: 219–283.

Stelzenmüller, V., Breen, P., Stamford, T., Thomsen, F., Badalamenti, F., Borja, T., Buhl-Mortensen, L., *et al.* 2013. Monitoring and evaluation of spatially managed areas: a generic framework for implementation of ecosystem based marine management and its application. Marine Policy, 37: 149–164.

Stelzenmüller, V., Ellis, J. R., and Rogers, S. I. 2010. Towards a spatially explicit risk assessment for marine management: Assessing the vulnerability of fish to aggregate extraction. Biological Conservation, 143: 230–238.

Stelzenmüller, V., Schulze, T., Fock, H. O., and Berkenhagen, J. 2011. Integrated modelling tools to support risk-based decision-making in marine spatial management. Marine Ecology Progress Series, 441: 197–212.

Stelzenmüller, V., Vega Fernández, T., Cronin, K., Röckmann, C., Pantazi, M., Vanaverbeke, J., Stamford, T., *et al.* 2015. Assessing uncertainty associated with the monitoring and evaluation of spatially managed areas. Marine Policy, 51: 151–162.

Tyler-Walters, H., Hiscock, K., Lear, D., and Jackson, A. 2001. Identifying species and ecosystem sensitivities. Final report to the Department for Environment, Food and Rural Affairs. The Marine Biological Association, MarLIN, Plymouth. 84 pp. + Annexes.

Vanhatalo, J. P., Tuomi, L. M., Inkala, A. T., Helle, S. I., and Pitkänen, J. H. 2013. Probabilistic ecosystem model for predicting the nutrient concentrations in the gulf of Finland under diverse management actions. Environmental Science and Technology, 47: 334–341.

Vugteveen, P., van Katwijk, M. M., Rouwette, E., and Hanssen, L. 2014. How to structure and prioritize information needs in support of monitoring design for Integrated Coastal Management. Journal of Sea Research, 86: 23–33.

Waugh, S. M., Filippi, D. P., Kirby, D. S., Abraham, E., and Walker, N. 2012. Ecological risk assessment for seabird interactions in Western and Central Pacific longline fisheries. Marine Policy, 36: 933–946.

Weisse, R., von Storch, H., Callies, U., Chrastansky, A., Feser, F., Grabemann, I., Günther, H., *et al.* 2009. Regional meteorological – marine reanalyses and climate change projections. Bulletin of the American Meteorological Society, 90: 849–860.

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.

Winiarski, K. J., Miller, D. L., Paton, P. W. C., and McWilliams, S. R. 2014. A spatial conservation prioritization approach for protecting marine birds given proposed offshore wind energy development. Biological Conservation, 169: 79–88.

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Contribution to the Themed Section: 'Risk Assessment'

Original Article

Review and refinement of an existing qualitative risk assessment method for application within an ecosystem-based management framework

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The adoption of risk-based methodologies is considered essential for the successful implementation of an ecosystem approach to fisheries and broader aquatic management. To assist with these initiatives, one of the qualitative risk assessment methods adapted for fisheries management over a decade ago has been reviewed. This method was updated to ensure compliance with the revised international standards for risk management (ISO 31000) and to enable consideration of ecological, economic, social, and governance risks. The review also addressed the difficulties that have been encountered in stakeholder understanding of the underlying concepts and to increase the discipline in its application. The updates include simplifying the number of consequence and likelihood levels, adopting graphical techniques to represent different consequence levels, and discussing how changes in uncertainty can affect risk scores. Adopting an explicit "weight of evidence" approach has also assisted with determining which consequence scenarios are considered plausible and, where relevant, their specific likelihoods. The revised methods therefore incorporate the conceptual elements from a number of qualitative and quantitative approaches increasing their reliability and enabling a more seamless transition along this spectrum as more lines of evidence are collected. It is expected that with continued application of these methods, further refinements will be identified.

Keywords: consequence, ecosystem approach, fisheries, likelihood, qualitative assessments, risk analysis, risk assessment, risk management, stock assessment, weight of evidence.

Introduction

Taking an "ecosystem" approach for the management of natural resources is increasingly recognized as most appropriate because it considers all relevant ecological, economic, social, and governance issues to deliver holistic community outcomes (FAO, 2002, 2003, 2012; Bianchi and Skjoldal, 2008; Bianchi *et al.*, 2008). With such a wide scope, an extremely large and diverse set of issues can be identified which often generates concern among managers, especially those with limited resources (FAO, 2009, 2012; Link, 2010; Fletcher and Bianchi, 2014). The use of some form of risk assessment to at least filter the different types of ecological issues has therefore increased substantially over the past decade (e.g. Fletcher *et al.*, 2002; Fletcher, 2005; Patrick *et al.*, 2009; Hobday *et al.*, 2011; Zhou *et al.*, 2011; MSC, 2014). This trend is consistent with growing recognition that fisheries and aquatic management are just specific forms of risk management (Francis and Shotton, 1997; Fletcher, 2005, 2008).

Risk management involves the explicit consideration of risks in all decision-making processes with risk assessment core to this by providing evidence-based information and analyses to help make informed decisions of the adequacy of current controls in achieving objectives (IEC, 2009; ISO, 2009; SA, 2012). The lack of available information for many issues is often seen as an impediment to completing formal risk analyses, including the completion of basic stock assessments for data-poor species. However, with the ISO definition of risk updated to "*the effect of uncertainty on objectives*" (ISO, 2009), examining risk now includes the clear articulation of objectives and the level of uncertainty generated from having incomplete information (IEC, 2009; SA, 2012). Uncertainty can be

explicitly incorporated within the analysis of risk by utilizing methods capable of using all available quantitative and qualitative data (IEC, 2009; Linkov *et al.*, 2009; SA, 2012).

Risk analysis, which is a critical part of the risk management process, involves consideration of the causes and sources of risk to achieving the objectives of an "organization" (which, in an aquatic resource management context, would include stakeholders and the relevant management agency). It also includes an examination of the magnitude of the potential consequences and the probability (likelihood) that those consequences will occur given current management controls (ISO, 2009; SA, 2012). One of the many qualitative risk analysis methods that conforms to these requirements is the consequence–likelihood (probability) matrix (IEC, 2009; SA, 2012). This C × L method is widely used as a screening tool in many fields, especially when a large number of potential risks may be identified (IEC, 2009; SA, 2012). This makes it highly suitable to cope with the large number of ecological, social, economic, and governance issues identified using an ecosystem approach. This method was first adapted for use in fisheries management within Australia over a decade ago (Fletcher *et al.*, 2002; Fletcher, 2005; Fletcher *et al.*, 2005) and has subsequently been applied in many other locations (e.g. Cochrane *et al.*, 2008; Fletcher, 2008; FAO, 2012). It has even been considered one of the ten "must be read" methods supporting the implementation of the ecosystem approach to fisheries (Cochrane, 2013).

Since its initial adaptation, this C × L method has been continually amended to better enable its use with ecosystem-based approaches for developing fisheries (e.g. Fletcher, 2008; FAO, 2012), regional-level, management-planning frameworks (e.g. AFMF, 2010; Fletcher *et al.*, 2010; MEMA, 2013), and for whole-of-agency risk-management systems (Fletcher *et al.*, 2012). Successive guidelines have included refinements that deal with the differing scopes of these frameworks and also address the difficulties often encountered with its implementation. This iterative process of improvement has resulted in many major enhancements being identified compared with the original published versions.

This review outlines the most significant updates made to each of the steps in the qualitative risk assessment process originally outlined in Fletcher *et al.* (2002) and Fletcher (2005). The key updates include (i) incorporating changes in the terminology and techniques now contained within the updated versions of the international standards for risk assessment and risk management; (ii) a summary of the main difficulties encountered when applying this methodology and descriptions of the refinements designed to improve clarity and consistency in terminology usage leading to an increased level of discipline and rigor when completing the analyses and evaluations; (iii) an expansion in the scope of the assessments to cover ecological, economic, social, and institutional components and their associated objectives to meet the requirements for full implementation of the ecosystem approach; (iv) an outline of how to integrate this methodology with the outputs generated from other assessment methods frequently used in fisheries management.

The outlined refinements, based on experiences gained in a wide variety of situations over the past decade (see references above), especially when embedded within a whole of agency risk management system, are expected to increase the efficiency, comprehensiveness, and robustness of the outcomes generated by the risk assessment process. This should improve both the timeliness and acceptance of any resultant management decisions, but most importantly, lead to better outcomes for aquatic natural resource managers and their respective communities.

Methods

The main activities undertaken in this review were to (i) examine the terminology that is used within the risk assessment documentation and compare this with the updated ISO standards, (ii) identify the key improvements that facilitate undertaking this form of risk analysis, and (iii) expand the scope of the methods to enable the assessment of the additional objectives covered by the ecosystem approach.

(i) *Terminology*: The qualitative risk assessment methodologies originally outlined in Fletcher *et al.* (2002) and Fletcher (2005) were based on risk management standard AS/NZ 4360 (SA, 2000, 2004). These international standards for risk management, risk assessment, and communicating and consulting about risk have subsequently been updated to ISO 31000 and ISO 31010 (IEC, 2009; ISO, 2009; SA, 2010, 2012). The specific methods and operational principles presented within fisheries and aquatic management risk assessment guidelines or presentations were therefore reviewed to ensure that the terminology, definitions, and techniques were fully compliant with these new standards (Table 1). Where appropriate, text from the various standards has been directly incorporated into the amended descriptions for each step in the risk assessment process. It should be noted, however, that alternative risk management frameworks and their definitions are available (e.g. ICES, 2013).

(ii) *Risk assessment techniques:* Based on considerable experience gained over the past decade from completing or facilitating assessments, undertaking training exercises, answering many queries, and developing a series of guidelines for different situations, the descriptions for each step in the risk assessment process have been updated. Areas where problems in the application of methods or interpretation of outcomes have most consistently been encountered were selected for specific examination. For each of these, the underlying basis for the errors or confusion was identified and descriptions of the refinements, which were developed to overcome these issues, were presented, together with examples.

(iii) *Objectives and scope of assessments:* The consequence tables were revised to ensure that they accommodated the broad range of objectives covered by the ecosystem approach (FAO, 2012). In addition to the set of ecological tables presented previously (Fletcher, 2005), an expanded set of consequence tables was compiled based on the common types of issues and high-level social, economic, and governance objectives frequently encountered across multiple country and fishery situations. The suite now not only allows for the assessment of risks associated with all aspects of the fishery but also extends to cover the factors affecting the internal governance and operations of the management agency and the industry.

Results

Risk assessment vs. risk management

The risk assessment process, which is an essential part of implementing a risk management system (Figure 1), includes three steps; risk identification, risk analysis, and risk evaluation. It is important to note that, while the other steps in the risk management process are not specifically covered in this review, they are all necessary to

Table 1. Definitions of risk management terms and their numbering as presented in the ISO 31000 (2009) plus notes on common issues to improve consistency of use within an ecosystem approach.

Standards, definition (and reference number)	Frequent issues
Risk (2.1) is the effect of uncertainty on objectives. It is often expressed in terms of a combination of the consequences of an "event" or "events" and the associated likelihood of the consequence actually occurring	This definition is much narrower than general public usage. It is commonly used instead of other more appropriate terms—threat, likelihood, vulnerability etc. It must be linked to meeting a specific management objective.
Context (2.9) defining the external and internal parameters to be taken into account when managing risk, and setting the scope and risk criteria	This includes the description of what is to be managed, the stakeholders that may be affected, the high level objectives to be achieved, the levels of acceptable impact (including their attitude to risk), and the timelines to assess risk. These must be established before completing a risk assessment
Risk assessment (2.14) includes the overall process of risk identification, risk analysis, and risk evaluation	
Risk identification (2.15) is the process of finding, recognizing, and describing risks. This may involve the identification of risk sources (2.16; the elements with the potential to give rise to risk), and/or events (2.17; their causes and potential consequences)	This step includes the identification of the issues, threats, impacts, and drivers that may affect the achievement of objectives—and therefore the risk. During this step, some of risk context elements may want to be re-examined
Risk analysis (2.21) is the process used to determine the magnitude or level of risk (2.23) which is expressed in terms of the combination of consequences and their likelihood	This is the most critical step, and it is therefore often thought of and incorrectly described as being the entire risk assessment step
Consequence (2.18) is the outcome of an event (2.17—which can include one or more occurrences of the event or even consist of something not happening) affecting objectives. It can be certain or uncertain, have positive or negative effects on objectives, and be expressed qualitatively or quantitatively	Most consequences will be described as different levels of impact for an asset. The separation points will be determined by what levels of impacts are considered acceptable for meeting the objective
Likelihood (2.19) is the chance of something happening and can be measured objectively or subjectively, qualitatively or quantitatively. It is used with the same broad interpretation as the term "probability" but less mathematical	This term is often misunderstood. It is not the likelihood of an event or activity but the specific likelihood a specific consequence actually occurring within the specified time frame.
Risk evaluation (2.24) is the process of comparing the results of a risk analysis with risk criteria (2.22; the reference levels against which the significance of a risk is evaluated) to determine whether the risk and its magnitude is acceptable or tolerable	Based on the risk score or level, this determines whether the current set of management actions needs to be change or increase, decrease, or remain the same

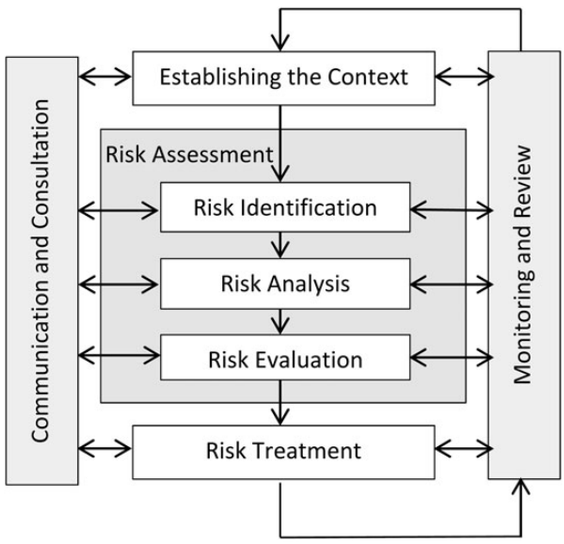

Figure 1. Position of risk assessment within the risk management process (modified from SA, 2012).

the overall success of the risk management process. Critically, unless the risk context, including the scope of management (which defines which activities, stakeholders and geographical extent will be covered), the objectives to be delivered, the time frame for the assessment, and what is considered acceptable performance have all been established, it is not possible to undertake a valid risk assessment. The various methods available to assist with the development of the risk context for a fishery or other aquatic activity plus the development of suitable risk treatments (the other two steps in the risk management process) are covered elsewhere (see FAO, 2012; Fletcher and Bianchi, 2014).

Definitions

Definitions of risk

Issue: The formal ISO definition of risk is now "*the effect of uncertainty on objectives*" (ISO, 2009). When applied to the ecosystem approach, a relatively high level should initially be taken by asking: "*What is the risk to meeting the agreed objectives for each asset (e.g. a fish stock or other ecological unit), outcome (e.g. food security, healthy community), system (e.g. management plan) from all the activities covered within the management system?*" (FAO, 2012).

As was previously identified by Francis and Shotton (1997), the word "*risk*" is used in a number of different ways. Many participants and stakeholders involved in risk assessment processes do not restrict their understanding or usage of the term "risk" to the international standards definition. The four most common alternatives being: (i) "*Threats*" such as too much fishing effort, or coastal pollution are often described as "*risks*". These are more formally described as the "events" or "risk sources" that can potentially

generate a level of risk of not meeting an objective; (ii) rare or long-lived species are often described as being "*at risk*" rather than being more accurately described as "*inherently vulnerable*" to various risk sources; (iii) it is also common to hear that the "*risk*" of a stock collapse occurring is "*x*", rather than the more appropriate phrase that the "*likelihood*" of a stock collapse is "*x*" which generates "*y*" level of risk; and (4) finally, the maximum "*potential consequence*" that could eventuate in a situation can be incorrectly used as the level of risk irrespective of how small the likelihood is for that consequence level actually occurring.

While all these elements form essential parts of the risk assessment process, they should not be used as synonyms for risk. The lack of clarity generated from a high level of incorrect and inconsistent usage of these terms can add considerably to the confusion of participants, increasing the difficulties completing the risk assessment and potentially affecting acceptance of the outcomes.

Refinements: Given the increasing adoption of formal risk-based management and risk assessment methods in fisheries and aquatic management, it is recommended that consistency is increased by adopting the ISO terminology. The international standard definitions for each of the main terms used in risk management (ISO, 2009). The set of common issues for each of these that may be encountered when this method is applied are presented in Table 1.

Definition of likelihood and consequence pairs

Issue: Another common difficulty in terminology has been the incorrect understanding of how the term "likelihood" should be applied within the risk assessment process. It is often incorrectly assumed to refer to (1) the "likelihood" that a particular activity/event (i.e. catching a species, going fishing) will occur; or (2) the "likelihood" that a set of management arrangements is (or will be) adopted; or even (3) the "likelihood" that any level of consequence may occur. In a formal risk analysis context, however, the term likelihood should only refer to the likelihood that a specific consequence will occur (SA, 2012).

Refinements: The relevant guidelines have been modified to more clearly describe likelihood in the risk management context as—the conditional likelihood that a specific level of impact (consequence level) may occur within the defined time frame, given the current or proposed set of management arrangements either from an accumulation of small "events" and/or from a single large "event". This description emphasizes that the selection of likelihood and consequence levels must form a pair and they should not be chosen independently.

Risk identification

Overview description: Risk identification is formally defined as the process of finding, recognizing, and describing risks, which involves the identification of risk sources and events, their causes, and their potential consequences including those managed and not managed by the "organization" (ISO, 2009). The process of identifying risks must involve individuals who have relevant knowledge and this activity should occur within an appropriate environment that enables effective stakeholder participation (SA, 2010). To facilitate this outcome, a wide range of tools that assist with effective risk identification for an ecosystem approach are now available from the FAO EAF toolbox (FAO, 2012; Fletcher and Bianchi, 2014).

Issue: The high level of stakeholder engagement that occurs during risk identification for an ecosystem approach often results in a wide variety of matters being raised (de Young *et al.*, 2008; FAO, 2012). These can include stakeholders opinions of the desired state for the ecological assets (e.g. target stock and ecosystem health) the types of social and economic outcomes (e.g. food security, economic rent, safe working environments) stakeholders want the management system to deliver; and the effectiveness and efficiency of the governance system (e.g. administration, compliance, monitoring, research, etc.). In risk management terminology, these are the goals and objectives of the risk management activities (ISO, 2009) and they are part of establishing the risk context. It is common, however, that the risk identification workshops are the first occasions when the various components of the risk context are presented or openly discussed. If most stakeholders present do not agree with the management objectives or levels of acceptable impact that are presented, the risk analysis process will be problematic and the outcomes unlikely to be definitive. This may require additional consultative processes for their resolution.

Participants will also identify what they consider to be the threats, impacts, and drivers (e.g. too much fishing effort, the price of fuel, illegal fishing, unsafe working conditions) that may be affecting the assets to be managed and the outcomes they provide. These risk sources or events may be generating potential consequences for one or more objectives and therefore affecting the level of risk (ISO, 2009). Both of these types of matters are important, but to complete the risk analysis phase, they need to be sorted into their respective categories.

Refinements: The items identified during the stakeholder workshops can be clearly sorted into the two categories. The set of ecological assets and social/economic outcomes (goals and objectives) to be achieved are listed as columns in a table with each of the identified risk sources (impacts/threats/opportunities) to these objectives listed as rows (see FAO, 2012, www.fao.org/fishery/eaf-net, for more details). This approach has the advantage of illustrating that a single risk source/event can affect a large number of objectives and a large number of risk sources/events can often be affecting a single objective.

Risk analysis

Overview description: Risk analysis involves the consideration of the causes and sources of risk, their positive and negative consequences, and the likelihood that those consequences can occur (ISO, 2009). The potential consequences, likelihoods, and resultant levels of risk are all dependent on the effectiveness of the controls that are in place (SA, 2012). Undertaking risk analysis using the consequence–likelihood ($C \times L$) methodology either involves multiplying the scores from qualitative or semi-quantitative ratings of appropriate consequence (levels of impact) and likelihood (levels of probability) of each of these consequences actually occurring from which a risk score and risk rating are calculated, or by directly assigning risk levels to each of the appropriate combinations of consequence and likelihood (IEC, 2009).

Determining the appropriate (plausible) combinations of consequence and likelihood scores should involve the collation and analysis of all information available on an issue. This will include (but is not limited to) the (i) inherent vulnerability of the ecological assets and the relative susceptibility of those assets to the various managed activities and other threats (risk sources/events) that may be affecting them; (ii) the level of uncertainty in the information available about the asset or the risk sources; (iii) the relative comprehensiveness and effectiveness of any current or proposed management systems in mitigating the effects of various threats or events; and (iv) the observed outcomes (lines of evidence) that results from these factors which, for captured species, often include the catch,

size composition, and spatial distribution of effort (see example in Table 3). Based on the available information and the expert opinions from those involved (including stakeholders), the most appropriate combinations of consequence and likelihood levels that fit the situation for a particular objective are selected.

If more than one combination of consequence and likelihood is considered plausible, the combination that generates the highest risk score (or risk level) should be chosen as the final outcome (i.e. consistent with taking a precautionary approach). Given that this is the most critical part of the risk assessment process, a number of procedures have been identified over the past decade that can improve the discipline and effectiveness for completing this step and therefore the robustness of the outcomes. The key elements are listed below.

Structure of the analysis methods

Issue: The consequence and likelihood tables can be user-defined and therefore individually tailored for each particular objective and its associated level of acceptability (IEC, 2009; SA, 2012). The number of different levels can also be varied to suit the level of detail most appropriate for each situation. There is a trade-off in the number of levels used because each of the tables needs to have suitable non-ambiguous descriptions relevant to the specific objective. A larger number of levels can increase the precision of outputs, but it can also increase the level of disputes in choosing between adjacent levels. Using fewer levels will increase the coarseness of the assessment, which can also reduce stakeholder acceptance.

Refinements: The original six by six level tables described in Fletcher (2005) have been considered too complex for use in many situations, but especially with developing fisheries. A four category system was therefore established for use in the Pacific and Africa (Fletcher, 2007, 2008), but this simpler structure (Figure 2) has also been accepted for use with other types of fisheries (FAO, 2012). Other structures can be applied where this is appropriate or required (e.g. most Western Australian Government Agencies use a 5 × 5 system), with between three and five levels being the most common (IEC, 2009).

Using this simpler four-level system, the standard generic descriptions for likelihood and consequence levels are presented in Table 2. These generic consequence descriptions should be individually tailored to become specific for each objective and clearly delineate the maximum acceptable level of impact, which in a four-level system, is normally consequence level 2. It is also common to include a "zero" consequence level because this can assist deal with situations where large numbers of "insignificant" issues are likely to be raised. Having a zero level enables scoring combinations of a high likelihood of a negligible (0) consequence (negligible risk), which is simpler for many participants to comprehend compared with having to choose a very low likelihood of even a minor consequence level actually occurring.

		Likelihood Level			
		Remote	Unlikely	Possible	Likely
Consequence level		1	2	3	4
Minor	1	1	2	3	4
Moderate	2	2	4	6	8
Major	3	3	6	9	12
Extreme	4	4	8	12	16

Figure 2. Consequence × likelihood risk matrix. The generic descriptions of each of the consequence and likelihood levels are presented in Table 2. The numbers in the cells indicate the risk score values and the colours/shades represent the levels of risk as described in Table 8.

Levels of data, uncertainty, and risk scoring

Issue: One of the biggest concerns in implementing the ecosystem approach is calculating the levels of risk for issues where there are minimal quantitative data. While risk is the effect of uncertainty on objectives, the process of undertaking risk analyses in situations where there are inherent uncertainties can cause a high degree of stress for some participants (including scientists).

Risk assessments are designed to make the most informed decision possible using all available information, even if this is limited (SA, 2012). It is important to recognize that not assessing the risk associated with an issue because there is a perceived lack of information essentially means that the current level of action or inaction is, by default, rated as acceptable. Where there are clear uncertainties, the highly disciplined approach outlined below can appropriately incorporate these into the justifications for the final scores that are selected. The justifications should include a suitably detailed narrative that refers to, and to the extent possible, is consistent with all available lines of evidence, including their levels of uncertainty (see Francis and Shotton, 1997, for a list of the different types of uncertainty).

Refinements: Evaluating the levels of risk associated with meeting an objective will inherently involve addressing uncertainties and variability that may occur in the future (SA, 2012). The level of current or future uncertainties associated with an issue can be included within the determination of the best combination of likelihood and consequence by incorporating all available lines of evidence and other information. The level of uncertainty can be conceptually depicted using the relative size of the "sphere" or range of plausible C × L combinations (Figure 3). As this sphere of uncertainty increases, this will result in progressively higher overall risk scores being selected.

To illustrate this concept, if the current level of impact on an objective was known with a high degree of certainty and precision to be fully within consequence level 2 (C2) (Figure 3, Sphere A). The appropriate qualitative risk score for this would be that it was "likely" (L4) to be a C2 consequence; which would generate a risk score of 8 which equates to a moderate level of risk (Table 8). If, however, for this same issue, less information had been available and the level of uncertainty increased, the sphere of plausible combinations could also increase potentially until the likelihood profile reached well beyond the boundary of the C2 into C3 (Figure 3, Sphere C). At this level of uncertainty, the more appropriate combination would be that it was possible (L3) for the level of impact to be at C3, which would generate a risk score of 9; which equates to a high risk (Table 8). With this outcome, additional data could be collected that reduced the uncertainty (and the size of the sphere) to an acceptable level (Figure 3, Sphere B). Alternatively, additional restrictions could be imposed that lowered the potential impact such that the "sphere" of plausible outcomes rose sufficiently

Table 2. Generic descriptions of likelihood and consequence using a four-level system modified from Standards Australia (2000), Fletcher *et al.* (2002), and Fletcher (2007).

Level	Likelihood descriptor
Generic likelihood levels	
Likely (4)	A particular consequence level is expected to occur in the time frame (indicative probability of 40 – 100%)
Possible (3)	Evidence to suggest this consequence level may occur in some circumstances within the time frame (indicative probability of 10 – 39%)
Unlikely (2)	The consequence is not expected to occur in the time frame but some evidence that it could occur under special circumstances (indicative probability of 3 – 9%)
Remote (1)	The consequence not heard of in these circumstances, but still plausible within the time frame (indicative probability 1 – 2%)
Level	**Consequence descriptor**
Generic consequence levels	
Negligible (0)	No measurable impact and no effect on meeting objective
Minor (1)	Measurable but minimal "impacts" that are highly acceptable and easily meet objective
Moderate (2)	Maximum acceptable level of "impact" that would still meet the objective
Major (3)	Above acceptable level of impact. Broad and/or long-term negative effects on objective which may no longer be met. Restoration can be achieved within a short to moderate time frame
Extreme (4)	Well above acceptable level of impact. Very serious effects on objective which is clearly not being met and may require a long restoration time or may not be possible

Note that the descriptions for each of the generic consequence levels need to be specifically tailored for each objective (see Table 7 for examples) and that inclusion of a zero level is recommended, but not essential.

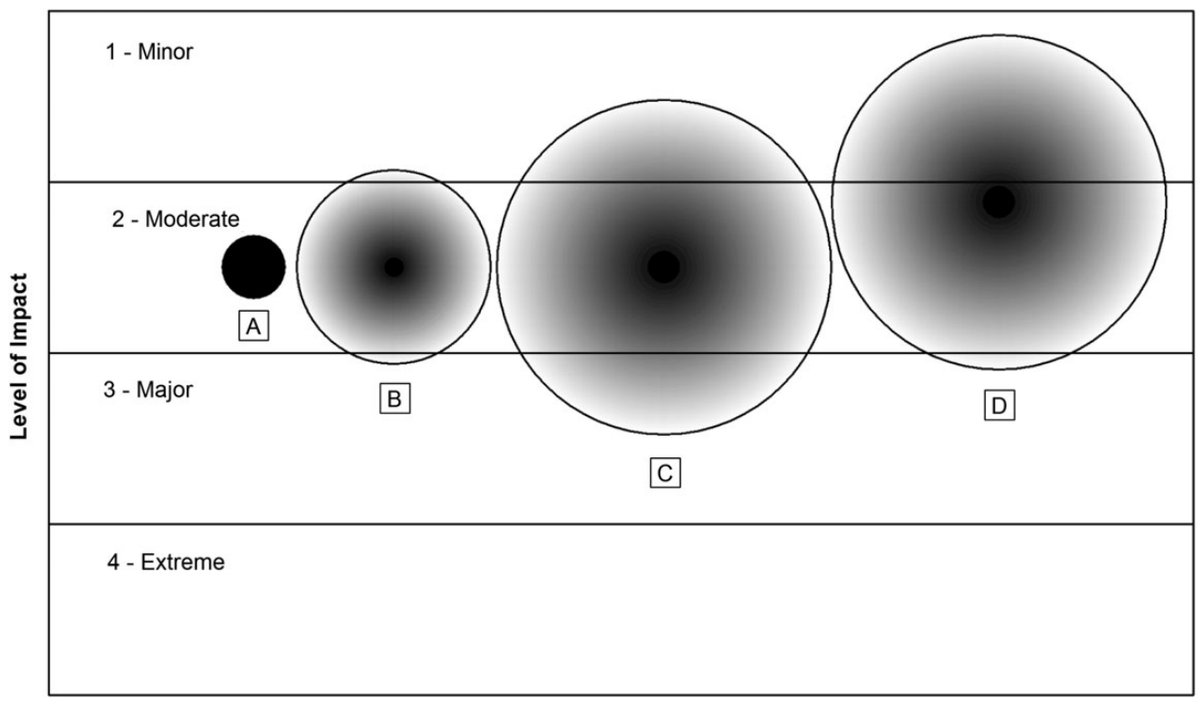

Figure 3. Pictorial representation of how uncertainty affects the sphere or range of plausibility for the same issue depending whether it is known with; high certainty (Sphere A), moderate uncertainty (Sphere B), or high uncertainty (Spheres C and D). The numbered levels on the impact scale represent the different consequence levels. The darker the region within the spheres represents higher likelihood; concentric bands could also be used for the different likelihoods (see text for details on resultant risk scores).

to be largely within acceptable levels (i.e. mostly above Level 3, Figure 3, Sphere D).

The use of this "sphere" or range of plausibility concept has been extremely valuable in getting participants to more clearly understand how qualitative risk analysis can be applied in a similar conceptual manner to more quantitative methods. An extension of this concept has been developed to increase the level of discipline applied when selecting the most appropriate C × L score combinations. This technique makes use of all available lines of evidence for an issue and is effectively a risk-based variation of the "weight of evidence" (WoE) approach that has been adopted for many assessments (e.g. Wise *et al.*, 2007; Linkov *et al.*, 2009).

The consistency or inconsistency for each line of evidence with the level of impact being within each of the consequence level scenarios is explicitly assessed. If all the lines of evidence for an issue are only consistent with a single consequence level (x)

then only a single C × L combination would be plausible. In this situation, the appropriate C × L combination would be that C "*x*" was "likely" (L4), which would essentially be equivalent to Sphere B in Figure 3. The more the different lines of data are consistent with different or multiple consequence levels, the wider the set of plausible combinations that would be generated. This would be equivalent to the larger spheres C or D in Figure 3.

Example**:** The commercial whiting fishery in Shark Bay, WA, provides an example of how WoE can be incorporated into this formal qualitative risk analysis approach. A preliminary assessment of this fishery examined the standard information on the biological characteristics (productivity) of this species and their potential susceptibility to the fishery (distribution vs. fishery boundaries), as used in the Marine Stewardship Council's pre-assessment framework (MSC, 2014), plus recent trends in catch and effort (Table 3). This level of information is common for data-poor fisheries both in WA and in many other areas of the world.

Using just these lines of evidence (Table 4), all four levels of consequence (depletion) would be plausible, but with different levels of likelihoods (Table 5). While the total catch and effort levels have been maintained at similar levels for at least 20 years (Jackson *et al.*, 2012), the boundaries of the fishery cover most of the species distribution in Shark Bay. These patterns could be consistent with stocks fished at light/acceptable levels (C1 or C2) or a stock that has been in a collapsed state for some time (C4). This catch history would not, however, normally be associated with a stock in transition (C3) because, over 20 years, this should either generate a trend of reducing catches for the same effort or an increase in effort would be needed to maintain catch levels (Caddy and Gulland, 1983; Hilborn and Walters, 1992). Hence, the stock is unlikely (L2) to be at a C3 level of depletion.

Adding the catch-history information to the biological/productivity characteristics of whiting, it is likely (L4) that the stock has been fished at moderate levels (C2). However, without any additional information, it is possible (L3) that it could either be lightly fished (C1) or, alternatively, it is possible (L3) that the stock could have already collapsed (C4) before the period examined and is not able to recover due to continued fishing pressure. With this amount of information and the corresponding level of uncertainties, there was a large set of plausible risk score combinations with the highest (C4 L3) equating (based on Table 8) to a high risk level (Table 5).

A more comprehensive assessment was completed for this fishery by including all the known lines of evidence (Table 4). This resulted in a revision to the plausibility and likelihood profiles associated with the four consequence scenarios.

The more detailed examination of the management arrangements recognized that there were only 12 commercial licences and only 5 active operators fishing across the whole of the 10 000 km^2 Shark Bay (Jackson *et al.*, 2012). Furthermore, these operators can only use beach-seine nets with a restricted length and mesh size

Table 3. Summary table of the information used to complete the two risk analyses of Shark Bay Whiting.

	Lines of evidence		
	Biology/productivity	**Susceptibility to the fishery**	**Outcomes**
Initial level of information	*Max age:* moderate—8 years *Age at maturity:* early—2 years *Reproductive strategy:* simple with high fecundity *Distribution:* around Shark Bay	*Distributional overlap:* 70% of total area within boundaries of the commercial fishery	
Additional level of information	*Distribution:* stock known to also be present in deeper waters of Shark Bay	*Management restrictions:* strong—only 5 licence holders currently operate, limitations on gear restricted to one beach-seine per crew with length and mesh restrictions. *Overlap in effective effort:* small proportion of beach area can actually be fished each year. Deeper waters not accessed. *Management effectiveness and compliance:* high compliance *Processor imposed:* catch limits per day for last 10 years	*Catch history:* stable catch levels for over 20 years *Effort history:* slight decline in effort over past 10 years *Market:* focus on high-quality product caps on daily catch levels *Catch composition:* most of catch is well above size at maturity

See Jackson *et al.* (2012) and Smallwood *et al.* (2013) for more information.

Table 4. The degree of consistency with the four levels of consequence for each of the different lines of evidence for Shark Bay whiting.

	Initial lines of evidence		**Additional lines of evidence**		
Consequence level scenario	**Potential overlap**	**Catch/effort**	**Management restrictions (effective overlap)**	**Catch size composition**	**Offshore distribution**
1	o	√	√	√	√
2	√	√	√	√	√
3	√	o	o	x	x
4	√	√	x	X	X

The biology/productivity information presented in Table 3 affected the interpretation of each of these different lines of evidence (see text for details). Legend: √, consistent; o, partially consistent; x, not consistent; X, inconsistent.

Table 5. Likelihoods (as indicated by Xs) for each of the consequence levels for the Shark Bay Whiting stock based only on the lines of evidence for biological/productivity, potential overlap/susceptibility, plus simple catch and effort (see Table 4).

Consequence Level	Remote 1	Unlikely 2	Possible 3	Likely 4	Risk score	Final risk level
1	X	X	X		3	
2	X	X	X	X	8	
3	X	X			6	
4	X	X	X		12	High

The final risk level is the combination that generates the highest risk score which, in this case (C4 × L3 = 12), equated to a high risk (Table 8).

Table 6. Likelihoods (as indicated by Xs) for each of the consequence levels for the Shark Bay Whiting stock that included the additional lines of evidence for total catch and effort plus management restrictions, effective effort levels, markets, and catch composition (see Table 4).

Consequence Level	Remote 1	Unlikely 2	Possible 3	Likely 4	Risk score	Final risk level
1	X	X	X		3	
2	X	X	X	X	8	MOD
3	X				3	
4					n/a	

The highest risk score combination was C2 × L4 = 8 which equates to a moderate risk (see Table 8).

with no access to deeper waters of Shark Bay where this stock is also known to occur (Kangas *et al.*, 2007). These formal restrictions result in the effective level of annual effort that can be applied by this fishery being very small (<5% of the shore line) if the extent of the distribution of fishing effort is compared with that of whiting across the entire Shark Bay region.

In addition to these formal restrictions, this fishery has been subject to processor restrictions for over a decade. To meet the market requirements of high-quality fish reaching the factory, commercial fishers are subject to a ceiling on the amount of fish they can land per day. Most importantly, the catch composition is dominated by fish well above the age at maturity (Gary Jackson, pers. comm.).

Including these additional lines of evidence in the analysis reduced the uncertainties and therefore the "sphere" of plausible outcomes (Table 6). It is much more certain that the effective level and distribution of annual effort is relatively small compared with the total distribution of whiting across Shark Bay. This reduces the potential level of fishing mortality on this stock. Furthermore, the size composition of the catch that has been maintained during the long history of stable catch and catch rate levels are relatively high, both of these lines of evidence are also consistent with the view that the level of fishing mortality is acceptable. In combination with the life history and catch history outlined above, these additional lines of evidence are all consistent with a stock that is stable and subject to sustainable levels of fishing.

Importantly, none of these additional lines of evidence were consistent with the scenario that the fishery has been operating on a collapsed stock (C4) for decades. For this to have remained plausible, the effective overlap of effort on this stock would have to be high with catch dominated by juveniles, as is true for those stocks known to be in a collapsed state (e.g. eastern gemfish; Flood *et al.*, 2012). The additional data were also inconsistent with this stock being close to being overfished (C3); hence, at most, there is only a remote likelihood of this scenario.

To further reduce uncertainty and discriminate between this stock being lightly fished (C1) or sustainably/"fully" fished (C2) would require a quantitative estimation of fishing mortality. Any decision to collect the additional data needed for this should be based on economic considerations because it should not be needed to meet sustainability objectives.

Stakeholder involvement and risk score selection processes

Issue: Application of this methodology can be undertaken with a high degree of stakeholder involvement with participants able to directly assist when selecting the appropriate C × L score combinations. This approach can increase the acceptance of the outcomes, but it can also lead to large discrepancies in the scores selected among individuals. This can often reflect that some stakeholders (i) are really assessing different objectives, (ii) have different ideas of acceptable impact, (iii) have different knowledge bases on the subject, or (iv) are unwilling to accept alternative risk outcomes to their preconceived positions.

Refinements: It is strongly recommended that workshops that apply this method utilize an experienced facilitator who fully understands both the underlying concepts and terminology of risk management and has direct experience in applying the ISO-based C × L methodology, including its idiosyncrasies. It is also preferable for any participants directly involved in scoring to be given some level of instruction on how these methods operate. This approach is one of the most widely used in the world, which means it is covered within the introductory risk courses available in most countries.

The discussions during these workshops must be undertaken in a language, and within an environment, where the participants feel comfortable and are able to freely and easily express their opinions (de Young *et al.*, 2008; SA, 2010; FAO, 2012). If there are different language or sector groups, it may be necessary to initially run separate sessions and have a separate meeting that synthesizes the outcomes.

Where the number of participants is very large, even with good facilitation, it can be hard to ensure that everyone is willing and able to apply the system in a consistent and objective manner. In such circumstances, it can be more effective to have the final risk score combinations chosen by a smaller "expert" panel which can include non-technical people. The broader audience can provide their input during an open discussion phase and provide subsequent comment on the outcomes. For example, the Western Rock Lobster Fishery in WA has both a Stakeholder Working Group and a Technical Panel that participate in risk assessments (Stoklosa, 2013). The Stakeholder Working Group includes a range of individuals and organizations involved in or interested in the fishery while the Technical Working Group is made up of a range of scientists with specific expertise relevant to the assessment. While both groups discuss all aspects of the risk assessment, only the Technical Panel completes the final risk scores with any discrepancies in scoring noted (Stoklosa, 2013).

Recording and reporting

To ensure that sufficient discipline and intellectual rigor has been applied to the risk analysis, it is essential that there is suitable documentation of the results of the assessment (SA, 2012). The justifications for choosing each of the different combinations of consequence and likelihood must be recorded within a suitably detailed narrative that examines and integrates all the lines of evidence, including their consistency and inconsistency with alternative scenarios.

A defendable case needs to be developed for the choice of each score combination so other parties who were not directly part of the risk assessment process can examine and understand the logic and assumptions used to make the decisions. Such documentation also assists the review of the risk sometime in the future if it is clear why the levels were originally chosen.

Assessing all relevant objectives

Issue: Applying an ecosystem approach involves the examination of a wide spectrum of objectives. If only risks associated with ecological objectives are examined, this will often lead to arguments or dissatisfaction with the outcomes of the risk assessment process because the ecological objective may not always be the highest risk. For example, the most common area where high risks have been identified, especially for developing fisheries, has been in governance, not ecological components as many would expect (Fletcher, 2008). The implications of using different consequence categories to assess the same information are illustrated below.

Example: Albacore Tuna are the primary target species for the tuna longline fishing managed by the Western Central Pacific Fisheries Commission and form the basis for cannery operations in some of the member countries (Williams and Reid, 2006). This species has a relatively robust biology and the stock assessment model at the time suggested it had been relatively resilient to the long history of fishing with the spawning biomass having not been substantially reduced. Under the rates of exploitation at the time, the total stock was likely to fluctuate well above the stock sustainability threshold level of B_{msy} (Figure 4a). Nonetheless, its local density can become reduced through intense fishing within a specific area and its migration routes can be affected by regional oceanographic conditions, both of which can affect the catch rates of member countries (Langley and Hampton, 2006).

Given the estimated biomass trajectories of the stock of Albacore at the time, from a stock sustainability perspective, it was unlikely (L2) the stock would decline to even a moderate level of depletion (C2). This represented only a low risk against this stock sustainability objective. From an economic objective perspective, however, the fishery needed to have the catch rates levels maintained at their historical levels with any material reduction in biomass expected to reduce the catch rate levels generating unacceptable economic outcomes (Figure 4b). Therefore, it was possible (L3) that the stock would decline below its current level (C3) which represents a moderate to high economic risk. This economic risk score explains why there were comprehensive management arrangements in place for this stock within each of these countries when the sustainability risk score was only low. The B_{mey} biomass level was effectively being used as the basis to determine acceptability in the risk/stock analysis.

Refinements: To implement an ecosystem approach, it is essential that the risks associated with all relevant objectives that were identified during the risk context step are assessed. This includes not only the risks associated with objectives for the ecological assets (target species, bycatch species, habitat, ecosystem structure) but also the assessment of objectives associated with: (i) the set of outcomes (economic and social) the community wants generated from the "use" of these assets; (ii) the governance (institutional and legal) systems used to manage the assets to achieve the outcomes; (iii) the set of organizational assets (buildings, people, etc.) and processes that undertake the management, and (iv) the external drivers (outside of direct management control) that may affect the ability to achieve these objectives all need to be assessed.

A starting set of consequence tables has been developed for applying the ecosystem approach (Table 7) which covers the most common fishery and management agency-related objectives. The descriptions for each of these tables has been developed based on experiences gained across many fisheries and situations but should be examined, and where necessary amended, to ensure they suit local circumstances.

Risk evaluation

Overview description: The risk evaluation step uses the risk scores or risk levels calculated from the risk analysis to help make decisions about (i) which risks need treatment, (ii) the degree of treatment required, and (iii) the priority for undertaking these actions. The risk evaluation is completed either by comparing the calculated risk score with those associated with the different levels of risk (e.g. Table 8), alternatively, where the risk scores are not considered sufficiently linear, each specific combination of consequence and likelihood can be directly assigned to a specific risk level (SA, 2012). Importantly, the determination of what risk scores, or what specific C × L combinations correspond to the different levels of risk, must be determined during the risk context step (i.e. before the risk assessment phase). These should be based on what constitutes acceptable performance and the degree of risk aversion of the managers and stakeholders (SA, 2012).

Issue: Following the risk assessment process, there can still be a large number of moderate or higher risk level issues identified that require attention. Determining the appropriate level and type of risk treatment (management actions) that should be applied to each issue will generally involve a number of factors apart from just the level of risk to one objective.

Refinements: A clear separation in the definition between a risk and a priority has now been included. The level of risk is only one of the factors that need to be considered when determining the priority of an issue. Other factors include the relative social, economic, or other benefit for society generated, the level of risk to these benefits, the time frame for failure if actions are delayed, the level of additional political fallout if it "fails", and also the degree to which the risk can be directly controlled. Determining priorities can involve some form of informal or formal multi-criteria analysis or other cost–benefit method which usually involves a high level of political input in the final decision-making process. To assist with these processes, a number of formal methods have been outlined in the EAF toolbox (FAO, 2012).

Discussion

The qualitative risk assessment methodology adapted for fisheries and aquatic management by Fletcher (2005) has been reviewed and updated to ensure full compliance with the revised international standards for risk management. In addition, many refinements have been made to assist with the efficient implementation of more holistic, ecosystem approaches.

The enhancements facilitate a higher level of stakeholder engagement and participation throughout each step of the risk assessment process, which should lead to a greater level of ownership and trust in the outcomes (de Young *et al.*, 2008; SA, 2010). The new tools enable stakeholders to outline their expectations and concerns, including relevant external factors, in a manner that can be assessed in a more consistent and objective manner. The highly transparent and logical nature of the C × L risk analysis method encourages full stakeholder engagement in discussions, scoring, and reviews of risks. These attributes were the principal reasons for the NSW

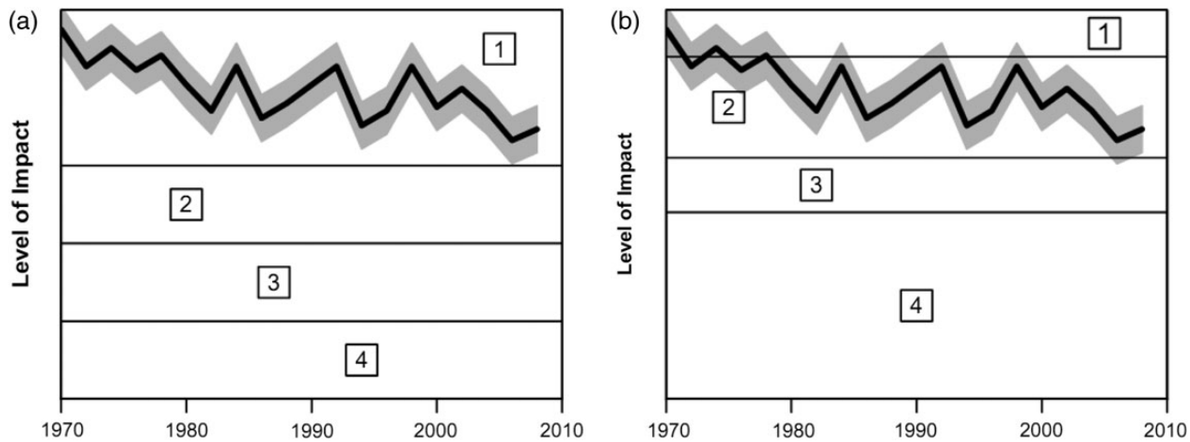

Figure 4. Illustration of how the same levels of impact on Albacore tuna abundance can result in different consequence scores using (a) stock sustainability and (b) economic outcomes as objectives. The horizontal lines in each graph indicate the separation points between the different consequence levels 1–4.

Marine Estate Management Authority deciding to adopt the C × L approach for their threat and risk framework rather than the methods previously applied in NSW (MEMA, 2013).

The method can be applied to the full variety of objectives relevant to the ecosystem approach including ecological, social, economic, political, and occupational safety issues. The different types of consequences and the levels of acceptable impact just need to be defined for each situation. This is often important for assessing objectives related to non-target, iconic species because the acceptable levels of impact can vary greatly among countries and time frames based on social considerations. For example, when the season for the western rock lobster fishery in WA was recently extended, this resulted in increased interactions with whales (i.e. entanglement of whales in ropes attached to lobster pots). A risk assessment completed in 2013 for this fishery as part of its ongoing MSC certification considered both the ecological and the social impacts of these whale entanglements (i.e. public concern for a dead whale on beach, a whale freed of entanglement or an entangled whale). While the impact of entanglements on the stock status of whales was considered C1 L1 (negligible), the social risk was, however, assessed as C2 L3 and therefore required management intervention (Stoklosa, 2013).

The selection of the appropriate risk score combinations for each objective is the most critical element in any risk assessment process and requires an appropriate level of discipline. Some risk analysis methods impose discipline by adopting a highly structured set of data inputs and score calculation procedures (e.g. MSC, 2014). This may result in a high level of consistency in outputs but unless all available information can be included and there are no complex or variable interactions among factors, their accuracy may be affected. Some of these methods have recognized this potential issue and have added an "expert override" step (e.g. Zhou *et al.*, 2011).

The selection of risk scores for the C × L method are made directly by either all or a subset of the participants following input and discussion of all the available information and viewpoints. These selections are all deliberate, "expert opinion" based decisions, with no predefined formulae used to calculate the final risk scores, so outputs from other analysis methods may be included within the deliberations. Most importantly, there are no restrictions on what information can be used or how it must be used to make the final decisions (although the basis for those decisions must be recorded). This high level of flexibility can potentially enable more accurate outcomes to be generated, but it also requires a higher comprehension of the underlying principles of risk assessment and strong discipline to apply the method appropriately and consistently (SA, 2012). The suite of refinements outlined above has been designed to achieve these improved outcomes.

It has been recognized for some time that the implementation of risk management could be assisted by improved consistency in the use of terminology (Francis and Shotton, 1997). The increasing level of adoption of risk-based approaches within aquatic management and its associated scientific literature suggests that it may be timely to better enforce compliance of use with the ISO standard irrespective of which risk analysis method is applied. The word risk, and all other risk-related terms, should, therefore, be restricted to the ISO definitions, similar to how the word "significant" is now largely restricted in scientific publications to a statistical definition.

Improvements to stakeholder understanding of the risk assessment process have also been obtained through the development of the various pictorial representations of how impacts, consequence levels, uncertainties, and likelihoods combine to determine the risk scores. The portrayal of qualitative assessments using two-dimensional graphs with the same conceptual units as would be applied in quantitative assessments effectively bridges the gap between these methodologies. It is consistent with the notion that the same principles should be applied for both qualitative and quantitative assessments except words rather than numbers are used to describe the magnitude of both the potential consequences and the probability (likelihood) that those consequences will occur (SA, 2012). For example, all stock assessments are essentially just specific forms of risk assessment that are completed to assess the risk status of fish stocks (see also Francis and Shotton, 1997).

The improved written and visual descriptions illustrate how the different risk analysis methods can be linked such that, with increasing levels of quantitative information, the precision for the levels of risk increases from (i) a qualitative "sphere" of plausibility; to (ii) a

Table 7. Qualitative levels of consequence for each of the main objectives relevant to the ecosystem approach.

Objective	Minor (1)	Moderate (2)	Major (3)	Severe (4)
Target species	Measureable but minor levels of depletion but no impact on dynamics Abundance range 100–70% unfished levels (B_0)	Stock has been reduced to levels approaching that associated with B_{msy} Abundance range <70% B_0 to >B_{msy}	Stock has been reduced to levels below B_{msy} and close to where future recruitment may be affected Abundance range <B_{msy} to >B_{rec}	Significant stock size or range contraction has occurred with average recruitment levels clearly reduced (i.e. recruitment limited) Abundance range <B_{rec}
Bycatch species	Species assessed elsewhere and/or take is very small and area of capture small compared with known distribution (<20%).	Relative level of susceptibility to capture is <50% and not a vulnerable life history	N/A. Once a consequence reaches this point, it should be examined using target species table	N/A
Protected species	Few individuals directly impacted in most years, no general level of public concern	Catch or impact at the maximum level that is accepted by public	Recovery may be affected and/or some clear public concern	Further declines generated and major ongoing public concerns
Ecosystem structure	Measurable but minor changes to ecosystem structure, but no measurable change to function	Maximum acceptable level of change in the ecosystem structure with no material change in function	Ecosystem function now altered with some function or major components now missing and/or new species are prevalent	Extreme change to structure and function. Complete species shifts in capture or prevalence in system
Habitat	Measurable impacts very localized. Area directly affected well below maximum accepted	Maximum acceptable level of impact to habitat with no long-term impacts on region-wide habitat dynamics	Above acceptable level of loss/impact with region-wide dynamics or related systems may begin to be impacted	Level of habitat loss clearly generating region-wide effects on dynamics and related systems
Economic	Detectable but no real impact on the economic pathways for the industry or the community	Some level of reduction for a major fishery or a large reduction in a small fishery that the community is not dependent upon	Major sector decline and economic generation with clear flow on effects to the community	Permanent and widespread collapse of economic activity for industry and the community including possible debts
Social structures	Impacts may be measurable but minimal concerns	Clear impacts but no local communities threatened or social dislocations	Severe impacts on social structures, at least at a local level	Complete alteration to social structures present within a region
Food security	Food security important but no impacts observed	Direct impacts on food resources but not to the point where these are threatened	Significant and long-term (>weeks) impacts on food for a community. Likely to lead to health problems	Severe ongoing reductions in food resources leading to starvation, abandonment of region, or requiring aid
Social amenity	Temporary or minor additional stakeholder restrictions or loss of expectations	Ongoing restrictions or decrease in expectations	Long-term suspension or restriction of expectations in some key activities	Permanent loss of all key expectations for recreational activities
Reputation and image	Low negative impact, low news profile	Some public embarrassment, moderate news profile, minor ministerial involvement	High public embarrassment, high impact, and news profile, Third party actions, public and significant ministerial involvement	Extreme public embarrassment, prolonged news coverage. Third party actions/enquiry, government censure
OHS	First aid only	Minor medical treatment required, visit to doctor's surgery. Less than a week off work	Hospitalization and/or intensive and extended treatment period required for recovery	Serious or extensive injuries/disease/permanent disability or death
Operational effectiveness	Non-achievement of an entire strategic directive	Minor element of one key deliverable unable to be achieved on time	Significant delay but achievement of key deliverables	Non-achievement of more than one key deliverable or major delay to entire strategic directive

Note the 0 level has not been included as this is generally described in all circumstances as not detectable impact.

range of different likelihoods for each of the plausible consequence levels; (iii) a single consequence level; (iv) a fully quantitative point estimate with error; and (v) a historical and future quantitative trajectory with error. The two-dimensional format has been successfully applied to illustrate other risks where the level of impact can theoretically be measured (see Albacore economic example, above). Where it is not possible to conceptually display the level of impact to the objective in such a manner suggests either that the objective has not been clearly defined or that another risk analysis method may need to be applied.

Table 8. Levels of risk and their associated likely management responses and reporting requirements (modified from Fletcher *et al.*, 2002; Fletcher, 2005).

Risk level	Risk scores (C × L)	Probable management response	Expected reporting requirements
Negligible (0)	0–2	Acceptable with no management actions or regular monitoring	Brief justification
Low (1)	3–4	Acceptable with no direct management actions and monitoring at specified intervals	Full justification and periodic reports
Moderate (2)	6–8	Acceptable with specific, direct management and regular monitoring	Full regular performance report
High (3)	9–16	Unacceptable unless additional management actions are undertaken. This may involve a recovery strategy with increased monitoring or even complete cessation of the activity	Frequent and detailed performance reporting

These concepts have also been incorporated into the risk analysis process through the explicit examination of the degree to which each line of evidence is consistent with each of the consequence level scenarios. Each information source is explicitly considered on its merit within an overall narrative that transparently discusses how these factors are thought to interact to determine which consequence scenarios are considered plausible and, where relevant, their specific likelihoods. The analyses of these various lines of evidence must include explicit consideration of how the current (or proposed) management system interacts with the underlying properties (e.g. productivity/susceptibility/vulnerability) of the asset being managed. The whiting example illustrated that with a more comprehensive examination of the effectiveness of the management restrictions, the calculated level of susceptibility assessed for this stock was substantially reduced compared with that which resulted from a simplistic assessment of susceptibility using fishing boundaries and biological productivity. Moreover, when used in combination with additional information on outcomes generated by management such as the patterns of catch, catch rate, and catch composition, a more precise risk profile was generated for this fishery.

Another advantage of the C × L methodology is that it can often be completed within a very short time frame using whatever data are available. For management agencies, this can be important because risk-based decisions are often required to be made in a matter of hours or days, not months or years. This attribute was recently used to provide timely advice to the Western Australian Government concerning their proposal to station drum lines off selected WA beaches to mitigate the risk of shark attacks (Government of Western Australia, 2013). A number of risk assessments associated with this proposal were completed to assess the potential environmental risks of this proposal and to examine the potential risks to the staff directly involved or indirectly affected by its implementation. Despite the short time lines available, the submitted environmental risk assessment (DoF, 2014) subsequently withstood independent review by the Environmental Protection Authority (EPA, 2014). Furthermore, the OHS-based operational procedures that were developed using this risk approach enabled the timely implementation of this controversial strategy by the Department in a safe and controlled manner.

While there are a number of clear benefits of this methodology, even with the added refinements, a number of inherent difficulties remain. Principally, if the facilitator has minimal experience with these concepts, and/or where the language skills and formal education of participants are not high, the use of this risk analysis method can be difficult to complete efficiently. In these situations, undertaking a simpler "risk category" based analysis method (see FAO, 2012) or other preliminary hazard analysis (IEC, 2009; SA, 2012) could be better options. A simple procedure well done may often provide better results than a more sophisticated procedure poorly done (SA, 2012).

Conclusion

The adoption of risk-based methodologies is now clearly seen as an essential component for the successful implementation of ecosystem management approaches (FAO, 2005, 2012), with qualitative assessments often the most appropriate for this purpose (Cochrane, 2013). The suite of refinements that have been developed for the C × L qualitative method over the past decade has greatly improved both its rigor and accessibility for stakeholders.

The focus of these refinements, which are relevant to all methods, emphasizes that risk assessment should not be viewed as just a technical scoring procedure but as an intellectual process that involves developing a conceptual model for each issue and an illustrated narrative that examines the consistency of all the lines of evidence against this model in a disciplined and auditable manner. These narratives should explicitly consider how the management system and uncertainties have affected the selection of the most appropriate risk score. From a manager's perspective, it is these narratives and the depictions of risk status that provide the basis to determine the most appropriate future "risk treatments" for an objective, not the risk score.

Incorporation of the conceptual elements from a number of qualitative and quantitative approaches in the updated methods have not only increased the reliability of those methods but also have enabled more seamless transition across these methods as more lines of evidence are collected and used to update the assessment. This will also assist agencies in the wider adoption of risk management principles to cover all their activities.

Given the variety of issues and situations that often arise when completing risk assessments, additional nuances are frequently identified that better explain or complete the process. It is expected that further refinements to the various risk assessment guidelines will continue to emerge over time.

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References

AFMF. 2010. Ecosystem Based Fisheries Management Approach: Guidelines for Adoption. Australian Fisheries Managers' Forum, October 2010, 10 pp. *In* FRDC Final Report 2000/145 National Application of Sustainability Indicators for Australian Fisheries. Part 2: Ecosystem based frameworks for aquaculture, multi-fishery and international applications. Fisheries Research Report 235, Department of Fisheries, WA, July 2012, Appendix 2, pp. 50–53.

Bianchi, G., Sandberg, P., Skjoldal, H. R., and Thorarinsson, K. 2008. The Bergen conference on implementing the ecosystem approach to fisheries (Bergen, Norway, 26–28 September 2006): summary and main conclusions. *In* The Ecosystem Approach to Fisheries, pp. 1–20. Ed. by G. Bianchi, and H. R. Skjoldal. FAO-CABI, Rome.

Bianchi, G., and Skjoldal, H. R. (Eds) 2008. The Ecosystem Approach to Fisheries. FAO-CABI, Rome. 363 pp.

Caddy, J. F., and Gulland, J. A. 1983. Historical patterns of fish stocks. Marine Policy, 7: 267–278.

Cochrane, K. 2013. 10 Must-Be-Read Publications on the Ecosystem Approach to Fisheries. Forum for Ocean Planning and Management. http://openchannels.org/literature-library-top-lists/10-must-be-read-publications-ecosystem-approach-fisheries.

Cochrane, K. L., Augustyn, C. J., and O'Toole, M. J. 2008. The implementation of the ecosystem approach to fisheries management in the Benguela Region: experiences, advances and problems. *In* The Ecosystem Approach to Fisheries, pp. 262–292. Ed. by G. Bianchi, and H. R. Skjoldal. FAO-CABI, Rome.

Department of Fisheries (DoF). 2014. Research Advice on the Proposed Shark Mitigation Strategy using drum lines for January to April 2014. Department of Fisheries, Western Australia, January 2014. Published on EPA website www.epa.wa.gov.au, 12 February 2014. 8 pp.

De Young, C., Charles, A., and Hjort, A. 2008. Human dimensions of the ecosystem approach to fisheries. An overview of context, concepts, tools and methods. FAO Fisheries Technical Papers, 489. Food and Agriculture Organisation, Rome. 152 pp.

Environment Protection Authority (EPA). 2014. EPA determines not to assess shark drum-line proposal. Public Advice under Section 39A(7) *Environmental Protection Act 1986*, 12 March 2014. 5 pp. Environment Protection Authority, Perth, Western Australia. http://www.epa.wa.gov.au/News/mediaStmnts/Documents/Media%20Statement%20Shark%20Drum%20Line%20March%202014%20and%20public%20advice.pdf.

Fletcher, W. J. 2005. Application of qualitative risk assessment methodology to prioritise issues for fisheries management. ICES Journal of Marine Research, 62: 1576–1587.

Fletcher, W. J. 2007. A Guide to Implementing an Ecosystem Approach to Fisheries Management (EAFM) within the Western and Central Pacific Region. Forum Fisheries Agency, Honiara, Solomon Islands. 79 pp.

Fletcher, W. J. 2008. Implementing an ecosystem approach to fisheries management: lessons learned from applying a practical EAFM framework in Australia and the Pacific. *In* The Ecosystem Approach to Fisheries, pp. 112–124. Ed. by G. Bianchi, and H. R. Skjoldal. FAO-CABI, Rome.

Fletcher, W. J., and Bianchi, G. 2014. The FAO–EAF toolbox: making the ecosystem approach accessible to all fisheries. Ocean and Coastal Management, 90: 20–26.

Fletcher, W. J., Chesson, J., Fisher, M., Sainsbury, K. J., Hundloe, T., Smith, A. D. M., and Whitworth, B. 2002. National ESD Reporting Framework for Australian Fisheries: The "*How To*" Guide for Wild Capture Fisheries. FRDC Project 2000/145, Canberra, Australia. 120 pp. http://www.fisheries-esd.com/a/pdf/WildCaptureFisheries_V1_01.pdf.

Fletcher, W. J., Chesson, J., Sainsbury, K. J., Fisher, M., and Hundloe, T. 2005. A flexible and practical framework for reporting on ecologically sustainable development for wild capture fisheries. Fisheries Research, 71: 175–183.

Fletcher, W. J., Gaughan, D. J., Metcalfe, S. J., and Shaw, J. 2012. Using a regional level, risk based framework to cost effectively implement Ecosystem Based Fisheries Management (EBFM). *In* Global Progress on Ecosystem-Based Fisheries Management, pp 129–146. Ed. by G. H. Kruse, H. I. Browman, K. I. Cochrane, D. Evans, G. S. Jamieson, P. A. Livingstone, D. Woodby, *et al.* Alaska Sea Grant College Program, doi:10.4027/gpebfm.2012.07.

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.

Flood, M., Stobutzki, I., Andrews, J., Begg, G., Fletcher, W., Gardner, C., Kemp, J., *et al.* (Eds) 2012. Status of key Australian fish stocks reports 2012. Fisheries Research and Development Corporation, Canberra. 420 pp. http://www.daff.gov.au/ABARES/Pages/exit_site.aspx?link=http://www.fish.gov.au/reports/Documents/SAFS_Report.pdf.

Food and Agriculture Organisation (FAO). 2002. Report of the Reykjavik conference on responsible fisheries in the marine ecosystem. Reykjavik, Iceland, October 2001. FAO Fisheries Report, 658. Food and Agriculture Organisation, Rome. 128 pp.

Food and Agriculture Organisation (FAO). 2003. Fisheries management. 2. The ecosystem approach to fisheries. FAO Technical Guidelines for Responsible Fisheries, 4. Food and Agriculture Organisation, Rome. 112 pp.

Food and Agriculture Organisation (FAO). 2005. Putting into practice the Ecosystem Approach to Fisheries management. FAO, Rome. 52 pp.

Food and Agriculture Organisation (FAO). 2009. Report of the Workshop on a Toolbox for Applying the Ecosystem Approach to Fisheries. Rome, 26–29 February 2008. FAO Fisheries Report, 884. Food and Agriculture Organisation, Rome. 52 pp.

Food and Agriculture Organisation (FAO). 2012. The EAF Toolbox: the ecosystem approach to fisheries. Food and Agriculture Organisation, Rome. 172 pp. www.fao.org/fishery/eaf-net.

Francis, R. I. C. C., and Shotton, R. 1997. "*Risk*" in fisheries management: a review. Canadian Journal of Fisheries and Aquatic Sciences, 54: 1699–1715.

Government of Western Australia. 2013. New measures to combat WA shark risks. 12 December 2013. http://www.fish.wa.gov.au/About-Us/Media-releases/Pages/New-measures-to-combat-WA-shark-risks.aspx.

Hilborn, R., and Walters, C. J. 1992. Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty. Chapman and Hall, New York.

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.

International Council Exploration of the Sea (ICES). 2013. Marine and coastal ecosystem-based risk management handbook. ICES Cooperative Research Report, 317, Copenhagen. 60 pp.

International Electrotechno Commission (IEC) 31010. 2009. International Standard. Risk management—risk assessment techniques. International Electrotechno Commission. Geneva, Switzerland. 90 pp.

International Organisation of Standards ISO 31000. 2009. Risk management—principles and guidelines. International Organisation of Standards, Geneva, Switzerland. 24 pp.

Jackson, G., Norriss, J., and Lunow, C. 2012. Inner shark bay scalefish fishery status report. *In* Status Reports of the Fisheries and Aquatic Resources of Western Australia 2011/12: The State of the Fisheries, pp. 49–61. Ed. by W. J. Fletcher, and K. Santoro. Department of Fisheries, Western Australia.

Kangas, M. I., Morrison, S., Unsworth, P., Lai, E., Wright, I., and Thomson, A. 2007. Development of biodiversity and habitat monitoring systems for key trawl fisheries in Western Australia. Final report to Fisheries Research and Development Corporation on Project No. 2002/038. Fisheries Research Report, 160, Department of Fisheries, Western Australia. 334 pp.

Langley, A. D., and Hampton, J. 2006. An update of the stock assessment for South Pacific albacore tuna including an investigation of the sensitivity to key biological parameters included in the model. WCPFC-SC2-2006 WP-4. 12 pp.

Link, J. S. 2010. Ecosystem-Based Fisheries Management: Confronting tradeoffs. Cambridge University Press, New York. 207 pp.

Linkov, I., Loney, D., Cormier, S., Satterstrom, F. K., and Bridges, T. 2009. Weight-of-evidence evaluation in environmental assessment: review of qualitative and quantitative approaches. Science of the Total Environment, 407: 5199–5205.

Marine Estate Management Authority (MEMA). 2013. Managing the NSW Marine Estate: Purpose, Underpinning Principles and Priority Setting. NSW Marine Estate Management Authority, Sydney, NSW. 15 pp. http://www.marine.nsw.gov.au/__data/assets/pdf_file/0004/498604/Managing-Marine-Estate.pdf.

Marine Stewardship Council (MSC). 2014. Guidance to the MSC Certification Requirements v1.4 Marine Stewardship Council. London. http://www.msc.org/documents/scheme-documents/msc-scheme-guidance-documents/guidance-to-the-msc-certification-requirements-v1.4/.

Patrick, W. S., Spencer, P., Ormseth, O., Cope, J., Field, J., Kobayashi, D., Gedamke, T., *et al.* 2009. Use of Productivity and Susceptibility Indices to Determine Stock Vulnerability, with Example Applications to Six U.S. Fisheries. US Department of Commerce NOAA Technical Memorandum. NMFS-F/SPO-101. 90 pp.

Smallwood, C. B., Hesp, S. A., and Beckley, L. 2013. Biology, stock status and management summaries for selected fish species in south-western Australian. Fisheries Research Report 242, Department of Fisheries, Western Australia. 180 pp.

Standards Australia (SA). 2000. Environmental risk management: principles and process. Based on AS/NZS 4360:1999, Risk Management. HB 203-2000. Standards Australia, Homebush, NSW, Australia. 81 pp.

Standards Australia (SA). 2004. Risk Management Guidelines: Companion to AS/NZS 4360: 2004. Standards Australia. Homebush, NSW. 116 pp.

Standards Australia (SA). 2010. Communicating and consulting about risk. HB 327. Standards Australia Limited. Sydney, Australia. 28 pp.

Standards Australia (SA). 2012. Risk management—guidelines on risk assessment techniques. HB 89. Standards Australia Limited, Sydney, Australia. 92 pp.

Stoklosa, R. 2013. West Coast Lobster Fishery Ecological Risk Assessment, 20 February 2013. Fisheries Occasional Paper 118. Department of Fisheries, Western Australia. 99 pp.

Williams, P., and Reid, C. 2006. Overview of tuna fisheries in the western and central Pacific Ocean including economic conditions—2005. WCPFC-SC2 2006/GN WP-1. 42 pp. http://www.wcpfc.int/sc2/pdf/SC2_GN_WP1.pdf.

Wise, B. S., St John, J., and Lenanton, R. C. J. 2007. Spatial Scales of Exploitation among Populations of Demersal Scalefish: Implications for Management. Stock Status of the Key Indicator Species for the Demersal Scalefish Fishery in the West Coast Bioregion. Fisheries Research Report 163, Department of Fisheries, Western Australia. 130 pp.

Zhou, S., Smith, A. D. M., and Fuller, M. 2011. Quantitative ecological risk assessment for fishing effects on diverse data-poor non-target species in a multi-sector and multi-gear fishery. Fisheries Research, 112: 168–178.

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Contribution to the Themed Section: 'Risk Assessment'

Original Article

Risk assessment of cartilaginous fish populations

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We review three broad categories of risk assessment methodology used for cartilaginous fish: productivity-susceptibility analysis (PSA), demographic methods, and quantitative stock assessments. PSA is generally a semi-quantitative approach useful as an exploratory or triage tool that can be used to prioritize research, group species with similar vulnerability or risk, and provide qualitative management advice. Demographic methods are typically used in the conservation arena and provide quantitative population metrics that are used to quantify extinction risk and identify vulnerable life stages. Stock assessments provide quantitative estimates of population status and the associated risk of exceeding biological reference points, such as maximum sustainable yield. We then describe six types of uncertainty (process, observation, model, estimation, implementation, and institutional) that affect the risk assessment process, identify which of the three risk assessment methods can accommodate each type of uncertainty, and provide examples mostly for sharks drawn from our experience in the United States. We also review the spectrum of stock assessment methods used mainly for sharks in the United States, and present a case study where multiple methods were applied to the same species (dusky shark, *Carcharinus obscurus*) to illustrate differing degrees of model complexity and type of uncertainty considered. Finally, we address the common and problematic case of data-poor bycatch species. Our main recommendation for future work is to use Management Strategy Evaluation or similar simulation approaches to explore the effect of different sources of uncertainty, identify the most critical data to satisfy predetermined management objectives, and develop harvest control rules for cartilaginous fish. We also propose to assess the performance of data-poor and -rich methods through stepwise model construction.

Keywords: chondrichthyans, demography, risk assessment, stock assessment, uncertainty.

Introduction

The field of risk assessment of chondrichthyan (sharks, skates, rays, and chimaeras) populations has lagged behind that of other vertebrate groups. This is due in large part to their comparatively low economic value, and as a consequence, their lack of basic life-history and fishery information. However, there is growing interest in this group, particularly sharks, sparked by the recent realization that many species have undergone substantial population declines (Stevens *et al.*, 2000; Baum *et al.*, 2003; Burgess *et al.*, 2005; Myers *et al.*, 2007; Dulvy *et al.*, 2008; Dulvy and Forrest, 2010; Cortés *et al.*, 2012). As a result, risk assessment of chondrichthyan populations, and the research to support it, is now drawing increased attention and resources.

The approaches used to assess the risk of various stressors, notably fishing, on chondrichthyan populations have been heavily influenced by both the quantity and quality of available data. This process takes different forms depending on the discipline and the questions being asked. In a conservation context, the objective is typically the avoidance of large population declines or extinction, whereas in fisheries the goal is to maintain a healthy population while allowing for its sustainable, long-term exploitation. In both cases, a common objective is estimating current status and projecting

future trends of a population subjected to stressors or management intervention (e.g. fishing, habitat degradation, and improved water quality). Both current and future status will depend on the population's life-history characteristics; in addition, future status will depend on the type of management action that is implemented.

We consider the process of estimating vulnerability, population growth rates, or stock status and evaluating potential consequences of management actions to fall broadly under the category of "risk analysis" or "risk assessment". A more narrow distinction could be made between risk assessment and stock assessment; however, in this review, we treat stock assessment as part of the continuum of risk analysis methods, where the appropriate method depends on the amount of data available (Figure 1). Burgman *et al.* (1993) define risk assessment as the process of obtaining qualitative or quantitative measures of risk levels, or the probability of an adverse event. Rosenberg and Restrepo (1994) refer to an *ad hoc* working group that defined risk as the "expected loss of benefits from the resource" and risk analysis as "the analysis of benefit streams under uncertainty". A more comprehensive definition includes both the probability of an event and some measure of the severity of the event (Francis and Shotton, 1997). Furthermore, the International Organization for Standardization defines risk as the effect of uncertainty on objectives (ISO 31000, 2009). By reviewing both the methods to assess risk, and the types of uncertainty each method can account for, our review of risk assessment encompasses all these definitions to some extent.

We review three broad categories of risk assessment methodology that have been used for cartilaginous fish, noting the data required and the types of management products that are generated. We also discuss types of uncertainty, how they can be modelled, and which risk analysis methods can accommodate these uncertainties. Because risk analysis can have different objectives for different contexts, we discuss the approaches that have been traditionally used in the conservation arena and compare them with those followed in the field of fisheries. We then review the different types of stock assessments used mainly for sharks in the United States, showcasing a study where a comprehensive suite of methods were applied to Dusky shark (*Carcharhinus obscurus*). We conclude with considerations for bycatch species, review a framework for simultaneously exploring the effect of different sources of uncertainty, and make recommendations for future work.

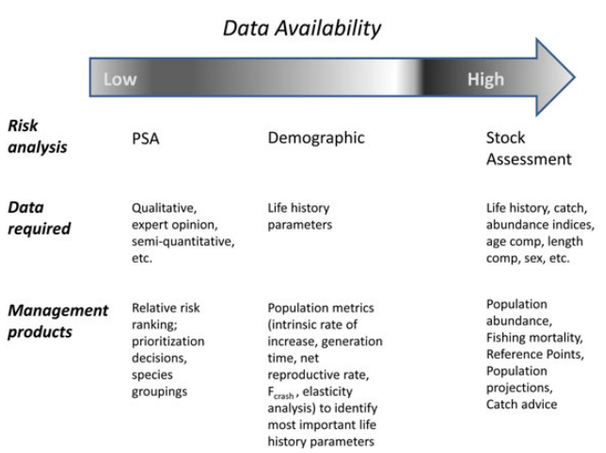

Figure 1. Continuum of risk assessment methods and the types of management products they generate. Although the figure presents the methods as a linear continuum, we recognize that there is overlap between the risk analysis categories.

Risk assessment methods

Productivity and susceptibility analysis

Data-poor situations are generally the norm when assessing risk of chondrichthyan populations. This group of fish is often taken as bycatch in many fisheries around the world and their biology is poorly understood. This situation gave rise to the use of productivity and susceptibility analysis (PSA, also known as ecological risk assessment or ERA), an approach initially designed to provide management advice when faced with cursory exploitation and biological information for a suite of species caught as bycatch (e.g. Stobutzki *et al.*, 2001). This approach ranges from purely qualitative to quantitative, and is designed to provide management advice by assessing the vulnerability to fishing of a species or population. Vulnerability is expressed as a function of productivity, or capacity of the stock to recover after it has been depleted, and of susceptibility, or propensity to be captured by fishing practices and not survive the interaction. In its most widely used application, PSA is a semi-quantitative approach wherein the productivity and susceptibility components are defined by several attributes that are scored based on a predetermined numerical scale. The attribute scores are then averaged for each component and displayed graphically on an x–y (PSA) plot (Figure 2). Although not generally done, the range or a measure of variability of the attribute scores from different experts can also be displayed to convey "inter-expert" uncertainty. From this, vulnerability can be computed, for example, as the Euclidean distance from the origin to the coordinates of the productivity and susceptibility scores on the PSA plot. Examination of these plots provides a quick, practical tool to assess the potential or risk of a stock to become overfished based on its biological characteristics and susceptibility to exploitation. These plots can be used by managers to adjust management measures to suitable levels given the stock's level of vulnerability. PSA can also be used to prioritize research efforts, for example, toward species that are very susceptible to fishing and for which the biology is poorly understood.

A two-step PSA has recently been developed that builds on existing approaches. In the first step, stock vulnerability is evaluated based on the usual life-history parameters to identify high-risk stocks; the second step evaluates the management risk by considering factors such as the existence of a stock assessment, management controls, and monitoring and compliance (Fleming *et al.*, 2012; Sant *et al.*, 2012; Lack *et al.*, 2014). The outcome for this approach is to identify specific management needs for high-risk stocks.

PSA approaches fall short of providing quantitative management advice, such as appropriate levels of fishing mortality, effort, or catch (but see Zhou *et al.*, 2012 for an approach that combines PSA with indices of relative abundance trends). PSAs should thus be viewed as a first step or triage method in data-poor situations within the spectrum of risk analysis techniques that can be applied as more data become available (see Hobday *et al.*, 2011, for example). Nevertheless, it is being used in the United States to distinguish between fishery and ecosystem component stocks, identify and manage stock complexes based on similar vulnerabilities, and establish management (harvest) control rules that take into account scientific and management uncertainty and provide a larger buffer for species with increased vulnerability to overfishing (Patrick *et al.*, 2010). Several Regional Fishery Management Organizations have adopted this approach in recent years with the aim of providing

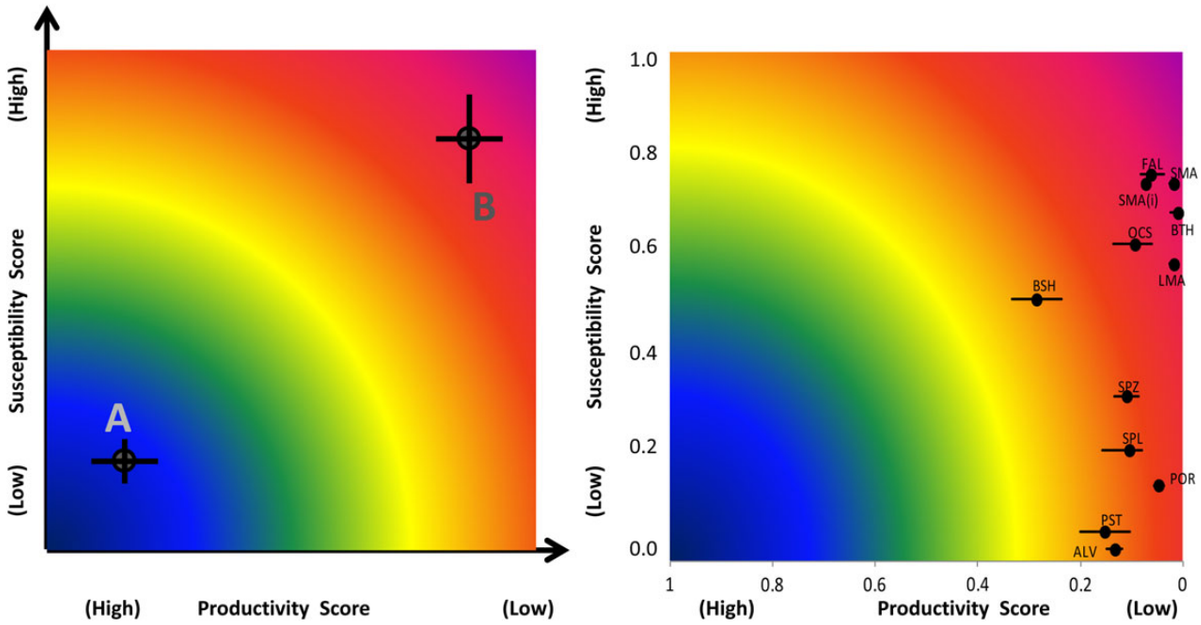

Figure 2. PSA plots. The left panel shows a theoretical example for two species (after Patrick *et al.*, 2009), where species A has high productivity and low susceptibility while species B has low productivity and high susceptibility. Species B would be considered to have higher risk (i.e. greater vulnerability) than species A. Error bars denote the range or a measure of variability of the attribute scores from different experts. The right panel shows a real application to 11 species of Atlantic pelagic elasmobranchs. Note that species greatly differ in their susceptibility score but all have relatively low productivities. Productivity scores incorporated uncertainty in input life-history parameters used to estimate the intrinsic rate of population increase (denoted by the error bars; after Cortés *et al.*, 2010).

management advice for data-poor species for which traditional stock assessments cannot be undertaken. The International Commission for the Conservation of Atlantic Tunas (ICCAT), for example, has recently adopted several management measures for pelagic sharks based on an ERA for the effect of pelagic longline fisheries (Cortés *et al.*, 2010; Figure 2b).

PSAs that compared different groups of fish or vertebrate taxa have consistently found that chondrichthyans were the most vulnerable. For example, Atlantic sharks and North Pacific skates were classified as the most vulnerable in a comparison of Northeast Atlantic groundfish, Atlantic sharks, California nearshore groundfish, California Current coastal-pelagic species, Bering and Aleutian Island skates, and Hawaiian tuna, swordfish, and pelagic sharks. Further, in the Hawaii-based longline fishery, pelagic sharks were more vulnerable than tunas, swordfish, and billfish (Patrick *et al.*, 2010). The same result was found in a comparison of Atlantic sharks, tunas, swordfish, and billfish (Rosenberg *et al.*, 2009). In the Western and Central Pacific Ocean, a PSA of birds, turtles, sharks, tunas, and billfish also found that sharks had the highest vulnerability (Manning *et al.*, 2009). Stobutzki *et al.* (2002) analysed the sustainability of elasmobranchs (sharks and rays) caught as bycatch in a tropical shrimp trawl fishery in Northern Australia and found that pristids (sawfish) and two species of rays have the highest risk. Cortés *et al.* (2008) included large coastal, small coastal, pelagic, and prohibited sharks in a PSA for the effect of fisheries in the Northwestern Atlantic off the United States, and found that coastal sharks were the most vulnerable, particularly larger species that tend to have low productivity and high susceptibility to multiple fishing gears.

In an extension of these more traditional PSAs, Chin *et al.* (2010) developed an integrated risk assessment to examine the vulnerability to climate change of sharks and rays on Australia's Great Barrier Reef. The assessment used three common components to measure vulnerability to climate change: exposure, sensitivity, and adaptive capacity. Freshwater, estuarine, and reef-associated elasmobranchs were found to be most vulnerable to climate change, with vulnerability being driven by species-specific interactions of multiple environmental and ecological factors. Changes in temperature, freshwater input, and ocean circulation tended to have the most widespread effects.

Demographic analysis

Demographic analyses, such as life tables and matrix population models, are another common approach to risk assessment of chondrichthyan species. These methods provide a quantitative estimate of the population intrinsic, or maximum, rate of increase (r_{max}) and other associated population metrics, such as generation time and net reproductive rate. They can be used to assess the level of fishing mortality (F_{crash}) that a stock can sustain before the population growth rate becomes negative and in theory leads to extinction. In some cases, mark-recapture methods have been used to estimate total fishing mortality (Z), from which F can be derived, and thus examine sustainability of shark fisheries (Simpfendorfer, 1999; McAuley *et al.*, 2007; Bradshaw *et al.*, 2013). A more complete accounting of uncertainty in demographic models is done by introducing variability in life-history traits such as fecundity, age at first reproduction, longevity, and natural mortality through Monte Carlo simulation or other resampling methods to generate probabilistic outcomes of the population metrics of interest or to predict extinction risk (also known *sensu lato* as population viability analyses, or PVAs; see, e.g. Fieberg and Ellner, 2001; Cortés, 2002a). The uncertainty introduced in these risk assessments is generally

more epistemic than reflective of our knowledge of natural variability in life-history traits.

A notable shortcoming of demographic methods when applied to chondrichthyan fish is that they do not provide information on stock status. This is because the initial age-structured population abundance is not typically known, although the asymptotic stable age distribution (proportion at age) can be obtained from life tables or as the dominant right eigenvector of a matrix population model (Caswell, 2001). In the interest of exploring transient dynamics, rather than the asymptotic distribution, investigators have simulated an initial population size and age structure, allowed vital rates to vary annually, then compared the results of implementing different harvest levels (Cortés, 1999; Aires-da-Silva and Gallucci, 2007). The output of demographic analyses of shark populations has also been used to generate informative prior distributions of the population growth rate or related parameters, such as steepness (Mace and Doonan, 1988) or the maximum lifetime reproductive rate (Myers *et al.*, 1997), for use in Bayesian stock assessment models (e.g. McAllister *et al.*, 2001, 2008; Cortés, 2002b). It is important to note that productivity derived from demographic methods (expressed as $r_{\max}$) is typically based on density-independent theory, while productivity in fisheries models (e.g. steepness) is predicated on density-dependent premises. In both contexts, the productivity metric is intended to reflect the maximum realizable rate of population growth (Gedamke *et al.*, 2007; Cortés *et al.*, 2012).

Elasticity analysis is a common technique applied to matrix population models that can identify the life-history stages that most influence population growth rate, thereby providing a focus for management action (Benton and Grant, 1999). In the United States, for example, elasticity analysis was the basis for implementing minimum size limits for several shark species in an attempt to protect the vital rate (juvenile survival) that was found to be most important for population growth (Brewster-Geisz and Miller, 2000; Cortés, 2002a).

Stock assessment

In addition to PSA and demographic analyses, traditional stock assessment models have been used to analyse risk of chondrichthyan populations in the fisheries arena. The forms of these models range broadly in their level of complexity (Shertzer *et al.*, 2008; Cortés *et al.*, 2012), and ideally should be dictated by the data available. In general, more complex types of assessment models have greater data requirements. Perhaps most critical are data on catch and indices of abundance (developed from research surveys or catch-per-unit-effort). These data allow for annual estimates of population abundance and fishing mortality, which enables calculation of a population's current status.

Stock assessment models can be used to assess risk by providing probabilities of the stock or fishery exceeding biological reference points. In the United States, for example, these models commonly provide probabilities of the stock being overfished (i.e. biomass being below a threshold derived from B_{MSY}, the biomass level that produces MSY) or of overfishing occurring (i.e. fishing mortality being above F_{MSY}, the fishing rate that yields MSY). Once stock status with respect to these reference points has been established, projections can be performed to explore the likely effects of alternative harvest strategies (e.g. catch quotas) on future stock status (Francis and Shotton, 1997). These alternative projection scenarios can be considered by resource managers when making decisions on harvest levels, i.e. to help guide risk management.

A wide variety of stock assessment models exist from the very simple to the relatively complex. For simple models, one consideration is that the method supported by available data may not adequately reflect important biological processes. At the other extreme, model selection can be difficult when complex models include different dataseries, assumed error distributions, or data-weighting schemes. These issues all relate to uncertainty of one type or another, which we expand on below.

Types of uncertainty

Multiple types of uncertainty affect the stock assessment and fisheries management process. Francis and Shotton (1997) identified six types of uncertainty: process, observation, model, estimation, implementation, and institutions. We address each of these sources of uncertainty in the context of their consideration within risk assessment of chondrichthyan fish.

Process uncertainty

As noted by Francis and Shotton (1997), this type of uncertainty refers to natural variability in biological processes. It is often referred to as "process error" in state-space modelling to distinguish it from observation error (Hilborn and Mangel, 1997). Process error in recruitment is one of the most crucial and widely considered sources of uncertainty in modern stock assessments (Hennemuth et al., 1990; Quinn and Deriso, 1999). Because of their reproductive mode, sharks and chondrichthyan fish in general have a very limited number of offspring or eggs, and thus the spawner–recruit relationship is much more predictable than in teleost fish. This condition has led to reparameterizations of the spawner–recruitment curve into more biologically intuitive metrics, such as steepness, maximum lifetime reproductive rate, and pup survival at low population density (Brooks *et al.*, 2010). Process error can also occur in growth rate, maturation, and natural mortality; however, the range of fluctuation in these processes in chondrichthyan fish remains poorly understood. Process error is routinely incorporated into stock assessments and can also be introduced into demographic approaches.

Observation uncertainty

Measurement error is pervasive and almost impossible to avoid when collecting data. It occurs in scientifically designed surveys and in every source of fishery data, including landings, discards, ages of individual fish, and effort of fishers (Schnute, 1991). Observation error can be accounted for by demographic or stock assessment models to various degrees, from not at all to nearly fully through statistical techniques (e.g. maximum likelihood or Bayesian approaches). Even if the data contain no actual error, sampling itself is uncertain by definition because we are not observing the whole population.

Indices of abundance are particularly important when fitting models of population dynamics to data. Observation error in indices of abundance is now routinely taken into account in shark stock assessments through statistical standardization techniques, such as generalized linear models (GLMs) or analogous methods (Maunder and Punt, 2004). Despite efforts to account for all potential explanatory variables through statistical standardization, one recurring issue in shark stock assessments in the United States is that indices of abundance often show larger interannual variability than seems compatible with the life history of the species. This suggests that the GLMs do not always sufficiently account for all the noise in the data, including observation error.

An added problem when multiple indices are available is that different data sources can provide conflicting trends, leading to tensions among these indices when fitting the model. In such cases, the model might tend toward a compromising solution and not fit any index particularly well. As described by Francis (2011), this outcome is undesirable and probably not informative about the direction of population change. While the degree of reliability of the different indices can be conveyed through a variety of weighting schemes, these approaches still do not ensure that the indices track population abundance. For example, inverse CV weighting gives more weight and thus credibility to the most precise indices (those with lower CVs), but this may be reflective of larger sample size and not necessarily the ability to track relative abundance (e.g. NMFS, 2012). Conn (2010) developed a hierarchical approach that recognizes both process and observation errors in indices of relative abundance. This approach combines multiple indices into one, assuming that each index attempts to estimate the same underlying relative abundance. This approach has become one of several consistently in use for many shark stock assessments in the United States.

Observation error can also be reflected in estimates of life-history parameters such as growth rates, reproductive variables, or natural mortality, and can inform Monte Carlo or other resampling methods. Typically, analysts treat variability in life-history parameters as independent, when it may be that such variation is correlated. Brandon *et al.* (2007) review sampling schemes to obtain joint prior distributions that reflect realistic biological constraints between life-history parameters. This type of uncertainty can be incorporated into stock assessments and demographic analyses.

Model uncertainty

All models are necessarily simplifications of reality. Model uncertainty describes the degree to which the real system is adequately represented by the model. The uncertainty stems from an incomplete knowledge and characterization of the system, and it is introduced in two major forms: (i) model complexity and (ii) model structure.

Choosing the level of complexity requires balancing a trade-off: a simpler model will reduce the amount of data needed (thereby reducing other sources of uncertainty, such as observation error), whereas a more complex model can incorporate more processes important to describing population dynamics, but which may be poorly understood. We believe model choice should reflect a balance between data availability and parsimony—in some cases, compromising biological realism for a simpler model may be warranted, so long as the consequences of simplification are addressed when interpreting the results. As an example, shark stock assessments in the United States were typically conducted with surplus production models (Schaefer, 1954) in the 1990s when data available included only fragmentary catches, a few indices of abundance of relatively short duration, and little biological information. As time series of observed data increased in duration, and the knowledge of biological characteristics improved, age-aggregated production models were replaced by age-structured production models (Punt *et al.*, 1995) that more fully incorporate life history and better reflect the fisheries by accounting for size selectivity of different gear types.

Uncertainty in model structure stems from assuming a certain value and/or distribution for parameters and functional forms for variables (e.g. assuming natural mortality is constant vs. age- or time-dependent, dome-shaped vs. flat-topped selectivity curve, or lognormal vs. gamma error structure for process and observation error). The effect of some of these parameter and distribution choices on results can be explored through sensitivity analysis.

One can take the results of sensitivity analysis further by exploring the risk or consequence of applying alternative model structures on projections of future stock status. For example, conducting a stock assessment with three alternative model structures could produce three different estimates of allowable catch (or other management quantity) for the next year. A consequence analysis would take the advice from one model structure and evaluate the effect of implementing that catch advice in all three model structures (e.g. NEFSC, 2013). The results of a consequence analysis can be described graphically (Figure 3), and provide managers with a summary of the potential effects of basing management action on results from a particular model if the true (but unknown) model had a different structure. This technique differs from model averaging (Draper, 1995; Burnham and Anderson, 2002; Brodziak and Legault, 2005), where the results from different model structures (e.g. the diagonal elements in Figure 3) are weighted to obtain a single outcome.

In US stock assessment of sharks, the effect of using alternative values of parameter inputs that determine productivity (e.g. natural mortality, growth, and reproductive variables) is routinely

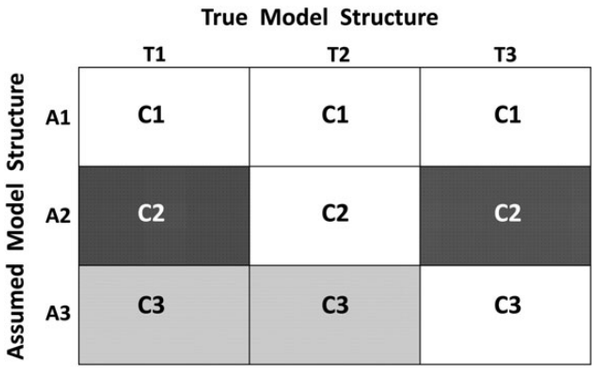

Figure 3. Example of results from a consequence analysis where three different model structures (A1, A2, and A3) are explored. Each model structure is used to perform a stock assessment, and some management quantity (e.g. catch) is estimated for each model (C1, C2, and C3) to achieve a specified goal (e.g. allow spawning biomass to increase). To evaluate the consequence of implementing catch advice from one model if in fact one of the other model structures were more appropriate, the catch from each assumed model structure is implemented in the full suite of models considered. In the above example, results are read across rows (and diagonal elements are self-consistent). The matrix of results is summarized in terms of the specified goal; e.g. if the goal was that spawning biomass would increase, then outcomes where spawning biomass either did not increase or decreased would increase the risk of implementing catch from that model structure. For this hypothetical example, a manager would conclude that the catch estimated for model structure 1 allows spawning biomass to increase regardless of whether or not it reflects the true (or most appropriate) structure. The catch estimated from model 2 only allows spawning biomass to increase if in fact model 2 is the true structure—thus that catch estimate should be considered a risky strategy. The catch estimate from model 3 allows spawning biomass to increase if model 3 is correct, but if not then the spawning stock is expected to remain at its current level (no increase or decrease). The shading of each cell reflects the positive (white), neutral (light grey), or negative (dark grey) outcome.

explored through the use of high and low productivity scenarios, as is the effect of assuming different distributions to describe virgin recruitment (e.g. NMFS, 2012). While performing sensitivity analysis has become routine in stock assessments, taking it further to perform consequence analysis can help managers realize the implications of their choices on future stock status with a more complete picture of model uncertainty. A full consequence analysis has not yet been considered in US shark assessments. This type of uncertainty is usually only considered in stock assessments, but not in PSA or demographic analyses.

Estimation uncertainty

This uncertainty relates to the process of parameter estimation and how well the parameters used for determining stock status represent the state of the stock. In shark stock assessments, uncertainty in parameter estimation is characterized in different ways according to the model used. The sampling-importance resampling (SIR) algorithm (e.g. McAllister *et al.*, 2008) or Markov Chain Monte Carlo (MCMC) (e.g. Cortés, 2002b) is used in Bayesian contexts, whereas bootstrapping (e.g. Simpfendorfer *et al.*, 2000; Hayes *et al.*, 2009) or delta methods (MacCall, 2013) are typically used in frequentist approaches. Accounting for estimation uncertainty results in distributions of model output rather than single point estimates. This type of uncertainty is often considered in stock assessments, much less frequently in demographic-type risk assessments, and not at all in PSAs.

Estimation uncertainty can be incorporated in the formulation of management control rules to help fishery managers establish fishing limits and allowable catches. For example, the estimated distributions from a stock assessment are used to define a distribution of catch that corresponds to F_{MSY}. This catch distribution, and specifically its central tendency, is referred to as the overfishing limit (OFL). A harvest control rule (HCR) can then be used to define the acceptable biological catch (ABC), which is some fraction of OFL that accounts for the degree of uncertainty in the OFL estimation (Figure 4). In US shark stock assessments, the ABC control rule sets a buffer of 30% between the OFL and ABC, i.e. the ABC is the 30th percentile of the OFL distribution, which corresponds to a $\geq$70% probability that overfishing will not occur.

Implementation uncertainty

Implementation uncertainty refers to how successfully management policies will be implemented (Patrick *et al.*, 2013). This is

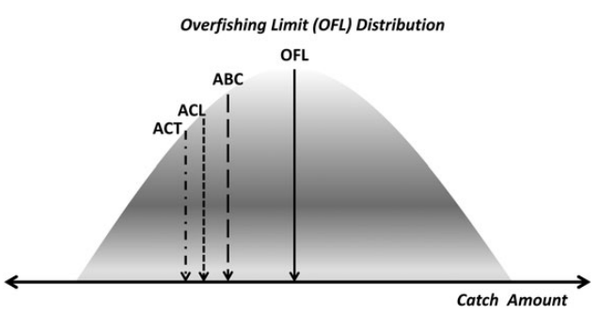

Figure 4. Summary of a type of HCR that determines a catch amount by considering estimation uncertainty (OFL $\rightarrow$ ABC) and management and implementation uncertainty (ABC $\rightarrow$ ACL). An additional buffer can also be accommodated (ACL $\rightarrow$ ACT) to avoid exceeding the ACL in a given year.

particularly problematic in developing nations or in the open oceans where enforcement is practically non-existent. This type of uncertainty could be incorporated into an HCR similar to estimation uncertainty. For instance, in the US example described above, estimation uncertainty defined the buffer between ABC and OFL. Implementation uncertainty could be used to create a second buffer that defines a lower annual catch limit (ACL). Exceeding the ACL can result in penalties, e.g. excess catch is "paid back" by subtracting it from next year's ACL. This can occur when in-season catch monitoring is imprecise or lags due to delays in reporting. To avoid a "payback" penalty, a third buffer can be defined between the ACL and a lower annual catch target (ACT; see Figure 4).

In US shark management, the ACL is set equal to the ABC. When the stock is overfished and rebuilding required, the ACL is defined as the projected catch level that produces $\geq$70% probability of stock biomass being above B_{MSY} by the end of the rebuilding time frame. The ACL is disaggregated into commercial, recreational, and discard components, and the commercial shark fishery can be closed when the quota reaches an ACT of 80% of the quota (NMFS, 2013b).

No formal HCRs have yet been developed in the United States to set ACLs and ACTs for managing lower tier (more data limited) shark stocks. In contrast, the southern and eastern scalefish and shark fishery in Australia developed a three-tier (1, 3, and 4) harvest strategy framework with an associated HCR for each tier that is used to determine a recommended biological catch (RBC) (AFMA, 2009). For tier-1 stocks (those with a well-established quantitative stock assessment), the RBC is calculated by applying F_{target} (the fishing mortality rate corresponding to a spawning biomass of B_{target}) to the current biomass to calculate the total catch in the next year. For tier-3 stocks (those without a quantitative stock assessment but with estimates of F and other biological information), the RBC is obtained from the current catch adjusted by the ratio of the intended and current exploitation rates, where the intended exploitation rate is based on the F for the RBC from the HCR. Tier 4 stocks are those corresponding to the most data-limited situations with no reliable information on current biomass or exploitation rate. For those stocks, the RBC is set based on a catch target derived from a historical period identified as a desirable target in terms of cpue, catches, and status of the fishery, the maximum level of catch that the HCR can set, target and limit cpues, and the average cpue over a given number of recent years. To further account for uncertainty in the lower tier stocks, a discount factor of 5 and 15% is applied to the RBC for the tier-3 and tier-4 stocks, respectively, to set a lower TAC (total allowable catch) with the aim of supporting stock recovery and preventing stocks from becoming overfished in the future.

Institutional uncertainty

Francis and Shotton (1997) further identified institutional uncertainty, arising from a lack of clear objectives for fisheries management and the interaction between different groups (scientists, managers, economists, fishers, and politicians). To some extent, the lack of clear objectives can arise from each group focusing on a different measure or consequence of risk (statistical probabilities, economic forecasts, future catch variability, and legal requirements of rebuilding). Reconciling these diverse considerations requires defining risk tolerance and the relative importance of each of these objectives. These decisions define risk management. In fisheries, the process of risk management is often qualitative and sometimes

only loosely related to the risk assessment from which it stemmed (Francis and Shotton, 1997).

Comparing risk analysis across disciplines

Extinction risk in marine fish has been measured through a variety of methods. Dulvy *et al.* (2004) noted that there is variation in both the definition of extinction risk and the degree of precision and defensibility of the risk assessment methods used in conservation biology, leading them to recommend a two-step approach for defining and assessing extinction risk. First, simple methods would be used to triage a large number of populations, and second, only those populations identified as vulnerable would be subject to more rigorous analysis. This approach is analogous to using some "rapid assessment techniques," such as PSAs, to identify those species or stocks more at risk then for those stocks, to apply stock assessments of different complexity based on data availability.

There has been intense debate over whether to apply methods of assessing extinction risk vs. methods of stock assessment traditionally used in fisheries for highly catchable and productive marine fish species (Matsuda *et al.*, 1998; Punt, 2000; Hutchings, 2001). Dulvy *et al.* (2005) addressed this issue in a study of 76 stocks of exploited marine fish and invertebrate species, in which they applied two criteria defined by the International Union for the Conservation of Nature's Red List of Threatened Species (IUCN, 2004) based on decline rates and population viability, and a criterion defined by the American Fisheries Society (Musick, 1999) based on decline rates and productivity. They compared predictions of extinction risk with those of stock status reported in stock assessments, and found that results from the two approaches were consistent. Davies and Baum (2012) also reported that IUCN conservation metrics and fisheries metrics (whether the stock was above or below reference points) agreed well in assessing the status of marine fish despite basic differences in the methods used in both disciplines; they suggested that the only difference was in the divergent philosophy of how to manage species of mutual concern. This difference between disciplines is exacerbated by the fact that fisheries scientists do not generally consider overfished populations to be at risk of biological extinction and highlights that risk tolerance is not the same because of divergent goals.

The spectrum of stock assessment methods

Biomass dynamic (age-aggregated surplus production) models are the simplest form of model used for assessing marine fish stocks, including chondrichthyans, around the world. Bayesian surplus production (BSP) models have been used for assessing large and small coastal sharks in the United States since 1998 and 2002, respectively (NMFS, 1998; Cortés, 2002b). The BSP model (McAllister and Kirkwood, 1998a, b; McAllister and Babcock, 2006) is a Schaefer biomass dynamic model that considers observation error only and uses the SIR algorithm to draw the estimated parameters from their joint posterior distribution and project the population forward under constant quota- or fishing mortality-based policy options. Probabilistic statements about the condition of the stock with respect to various indices of policy performance are then generated for different projection time intervals thus conveying the uncertainty associated with alternative harvesting strategies. Meyer and Millar (1999) developed a Bayesian state-space model incorporating both process and observation errors, which has been used in several stock assessments of Atlantic sharks (Cortés *et al.*, 2002, 2006). This model is implemented in WinBUGS and uses MCMC for numerical integration (Spiegelhalter *et al.*, 2000). No formal projections of future stock condition were developed with this approach. Jiao *et al.* (2009) illustrated the use of hierarchical BSP models for situations when species-specific data are unavailable in a hammerhead shark complex stock assessment. They found that models incorporating a multilevel prior on the population maximum growth rate ($r_{\max}$) fitted the data better than non-hierarchical models, which tended to produce credible intervals for estimates of stock status that were unrealistically narrow as a result of ignoring variability among species. These narrow intervals could lead to adoption of high-risk management strategies. In a follow-up study, Jiao *et al.* (2011) further explored the use of hierarchical and non-hierarchical BSP models for assessing fish complexes in situations where species-specific data were available, but were of different quality and quantity, concluding that the hierarchical models outperformed the non-hierarchical formulations because the poor-data species could "borrow strength" from the species with better data.

Age-structured production models are a bridge between the simpler production models and the more complex fully age-structured models (ASMs). The underlying dynamics are age-structured, but predicted values are aggregated across ages and compared with observed data that lack age information. The state-space age-structured production model (Porch, 2003a) is one example that can incorporate both observation error in the data variables (catches, cpue, and effort) and process error in state variables (effort, recruitment, and catchability deviations) and has been used to assess shark stocks in the United States since 2002 (e.g. Cortés *et al.*, 2002). Future projections of stock status initially included process error in recruitment only (Porch, 2003b); however, current projection methodology incorporates additional sources of variability in initial abundance, fishing mortality, pup survival at low density, and equilibrium recruitment. This approach also allows one to calculate probabilities of the stock being overfished and overfishing occurring for alternative levels of fixed removals each projection year (NMFS, 2013a). For overfished stocks in the United States, the population is first projected forward at $F = 0$ to determine the year when the stock recovers ($B/B_{MSY} > 1$) with a 70% probability. If that year is >10, then the stock must be rebuilt by the estimated rebuilding time $+1$ generation (Restrepo *et al.*, 1998). Fixed F and catch strategies can then be used to find the level that allows for the stock to be rebuilt with a 70% probability by the target year.

Porch *et al.* (2006) developed a variant of the age-structured production model for situations with no reliable catch history, a condition that is common in shark assessments. The state-space age-structured catch-free production model (ASCFPM) expresses the population dynamics on a relative scale (relative to virgin levels), to account for the lack of catch in the model. Model inputs include the usual age-specific vital rates, indices of abundance, and specification of a form for the stock–recruit curve, which for sharks can be parameterized in terms of maximum lifetime reproductive rate ($\hat{\alpha}$). The model estimates relative biomass trends, fishing mortality rates, predicted values for indices, and MSY-based reference points (abundance-related values are expressed relative to the unexploited level) and has been used for assessing dusky (*Carcharhinus obscurus*) sharks (Cortés *et al.*, 2006; NMFS, 2010), porbeagle (*Lamna nasus*) (ICCAT, 2010), and shortfin mako (*Isurus oxyrinchus*) (ICCAT, 2013).

Statistical catch-at-age models are the most complex form of model used for assessing shark stocks. Through "Integrated Analysis" (Maunder and Punt, 2013), these models attempt to

make use of multiple data sources simultaneously, generally including information on catch and indices of abundance, as well as age and/or length composition. These models can take many different forms (e.g. sex structure in addition to age structure), and their flexibility allows them to accommodate nearly any additional type of data that might be deemed important (e.g. tagging data). Punt and Walker (1998) used a statistical catch-at-age model, along with Bayesian inference and the SIR algorithm, to generate posterior distributions of virgin equilibrium biomass and a parameter determining the magnitude of density dependence in a stock assessment of the school shark (*Galeorhinus galeus*) off southern Australia. They also conducted a risk analysis consisting of probabilistic projections under alternative *F* levels.

Length-based ASMs are also being increasingly used to take advantage of the fact that lengths are often recorded in many fisheries and surveys for chondrichthyan fish. Age information is very scarce, in part because of insufficient sampling of catches, but also because cartilaginous fish are inherently difficult to age. Pribac *et al.* (2005) used a variant of integrated analysis wherein catch, catch rate, length and age compositions, and tagging data were used to assess the status of the gummy shark off the Bass Strait and South Australia within a maximum likelihood estimation framework. Frisk *et al.* (2010) developed an ASM that was fit to catch rate and length composition data to assess trends in winter skate (*Leucoraja ocellata*) abundance, biomass, and exploitation, testing hypotheses to explain the population dynamics of this species in the Georges Bank region.

Stock synthesis (SS), a widely used programme for integrated analysis, is a very flexible assessment framework that accommodates input of many different types of data, including both sex-specific length and age compositions (Methot and Wetzel, 2013). Gertseva (2009) used SS to assess the status of the longnose skate (*Raja rhina*) in the northeast Pacific Ocean, and more recently, Rice and Harley (2012) used SS to assess the status of the oceanic whitetip (*Carcharhinus longimanus*) shark in the western and central Pacific Ocean. As more and better data become available, we expect that shark assessments will rely less on data-poor methods and will transition toward integrated analysis, at least for some species.

Case study: the dusky shark

The dusky shark off the Northwest Atlantic Ocean provides a good example to illustrate the suite of analytic tools that can be used to determine the status of a stock under multiple sources of uncertainty. The dusky shark is a large coastal-pelagic species designated in 1997 as a candidate for listing under the Endangered Species Act in the United States, and classified as vulnerable in the western North Atlantic Ocean under World Conservation Union IUCN criteria in 2004. Capture of dusky sharks off the US East Coast has been prohibited since 2000. Data from a variety of sources and a portfolio of quantitative methods were used to assess the status of the dusky shark population in the western North Atlantic Ocean (Cortés *et al.*, 2006; Table 1). Trends in average size and catch rates (cpues) from five sources standardized through GLM statistical techniques were all found to have declined, many of them significantly. A demographic analysis was conducted in which uncertainty in life-history traits (age, growth, reproduction, and natural mortality) was incorporated through Monte Carlo simulation of life tables, which allowed consideration of a wide range of plausible parameter values. That analysis found dusky sharks to have long generation times (30 years), as well as very low population growth rates ($r_{max} < 0.023$ year^{-1}) and steepness ($h = 0.29$). Some of these estimated population parameters were later used to inform priors in Bayesian stock assessments. Elasticity analysis identified juvenile survival as the main contributor to population growth.

A broad spectrum of stock assessment methods was applied to evaluate stock status. Three complementary approaches of increasing complexity were used: BSP models, the catch-free age-structured production model; and an ASM that incorporated catch. Three Bayesian variants of Schaefer's biomass dynamic model were applied that allowed incorporation of different assumptions about observation and process error and numerical integration techniques: a BSP model with the SIR algorithm (McAllister and Kirkwood, 1998a, b; McAllister and Babcock, 2006), another version of the BSP model with the SIR algorithm but incorporating process error in the projections (Cortés, 2002b), and a state-space BSP model implemented in WinBUGS (Meyer and Millar, 1999).

Table 1. Methods used by Cortés *et al.* (2006) to estimate the status of the dusky shark (*Carcharhinus obscurus*) stock in the western North Atlantic Ocean.

Method	Type of uncertainty	Results	Conclusion
Trends in size		All decreasing (4 of 5, $P = 0.05-0.001$)	Heavily exploited, particularly immature stages
Trends in cpue	Observation	All decreasing (3 of 5, $P = 0.001$)	Declines >50% of virgin likely
Demographic analysis	Observation, model structure	Low productivity ($r < 3\%$ per year); long generation time (30 years)	Can withstand only very low *F*
Elasticity analysis	Observation, model structure	Juvenile (immature) stage most influential to productivity	Should protect immature sharks
Bayesian SPM	Observation, model structure, estimation	$B_{current}/B_{virgin} = 0.03-0.21$; stock overfished; overfishing occurring	Heavily depleted stock in need of rebuilding
Bayesian SSSPM	Observation, process, model structure, estimation	$B_{current}/B_{virgin} = 0.16$; stock overfished; overfishing not occurring[a]	Heavily depleted stock in need of rebuilding
SPMs (combined)	Observation, process, model complexity, model structure, estimation	$B_{current}/B_{virgin} = 0.03-0.21$; stock overfished; overfishing occurring	Heavily depleted stock in need of rebuilding
ASCFPM	Observation, process, model structure, estimation	$B_{current}/B_{virgin} = 0.04-0.13$; stock overfished; overfishing occurring	Heavily depleted stock in need of rebuilding
ASM	Observation, model structure, estimation	$B_{current}/B_{virgin} = 0.21-0.37$; stock overfished; overfishing occurring	Heavily depleted stock in need of rebuilding

The main results and conclusions from application of each method are listed for comparison along with the type of uncertainty that each method addressed. SPM, surplus production model; SSSPM, state-space surplus production model (WinBUGS); ASCFPM, age-structured catch-free production model; ASM, age-structured model.
[a]Only in terminal year.

While the data for production models were certainly available, these models are not able to incorporate important information about age-specific quantities, protracted maturation schedules, or generation time.

Estimates of age-specific vital rates for dusky shark from limited studies were used to derive inputs for ASMs to better capture the biology of the species. The ASCFPM (Porch *et al.*, 2006) and the ASM of Apostolaki *et al.* (2006) were both used. The ASCFPM was a convenient approach because it re-scales the model population dynamics as proportional to unexploited conditions, thereby eliminating dependence of model results on catch levels, which are poorly known. The ASM is sex specific, a feature that is considered important for describing population dynamics of dusky and other sharks.

Use of the three modelling approaches thus addressed several sources of uncertainty: observation, process, model, and estimation uncertainties. Model uncertainty was further addressed directly through model complexity (the type of model used) and model structure (via sensitivity analyses of several parameter input values or distributions). Uncertainty in data inputs was investigated through extensive sensitivity analyses. Estimation uncertainty was addressed through the use of different algorithms for numerical integration (SIR vs. MCMC) or the importance function used in the SIR algorithm (changing it from the priors to a multivariate *t*-distribution).

Despite the diversity of assumptions, required model inputs, and sources of uncertainty considered, the multitude of methods used provided a consistent picture of heavy fishing impact and high vulnerability to exploitation of dusky sharks in the western North Atlantic Ocean (Cortés *et al.*, 2006). All three stock assessment models generally estimated large depletions of at least 80% with respect to virgin levels. Such convergence of results suggests that the data, particularly the biological information and the indices of abundance, were robust and led to conclusions that were largely independent of the method used, despite the acknowledged sources of uncertainty.

Further considerations and recommendations

The case study described for dusky sharks, where multiple methods were applied to the same stock, is not possible for most chondrichthyans. These species tend to be bycatch, thus both the data and the range of applicable methods is limited (Stevens *et al.*, 2000). As a consequence, fisheries impacts on bycatch species are particularly difficult to quantify, and management objectives often lack specific bycatch reduction targets (Moore *et al.*, 2013). In these typically data-poor situations, multiple limit reference points based only on catch and life-history data have been proposed to identify sustainable levels of bycatch for non-target populations of marine megafauna. Moore *et al.* (2013) cite the potential biological removal (PBR) reference point used in the Marine Mammal Protection Act as an example of a precautionary approach to incorporating uncertainty directly into the reference point estimator to ensure relatively high population levels or a high probability of rapid recovery. However, for several elasmobranch species that are relatively abundant but of low economic value, depletion to lower abundance levels or a higher risk tolerance to a given level of bycatch may be a reasonable option (Zhou *et al.*, 2011). Even if a given bycatch or exploitation level in general exceeds the prescribed reference point (e.g. PBR), it could still be sustainable but with a lower degree of certainty (i.e. higher risk tolerance).

Concerns related to the ability of data-poor methods to accurately reflect the complex dynamics and protracted population response times are valid. Furthermore, the inherent uncertainty in data for bycatch species can complicate management decisions about buffer size, rebuilding targets, and how strictly to regulate the fisheries responsible for bycatch. A convenient framework for simultaneously exploring the effect of different sources of uncertainty is management strategy evaluation (MSE; Butterworth and Punt, 1999). In this approach, the entire assessment and management process is evaluated, from data collection to the application of HCRs, using Monte Carlo simulation where parameter or data values are sampled from relevant probability distributions (Little *et al.*, 2011). Typically, an MSE comprises an operating model that describes the "true" population dynamics of the stock, including process error; an observation/estimation model that generates data and estimates reference points considering observation (sampling) error and uncertainty in the operating model; and an assessment/management model that implements HCRs in response to the estimated stock status relative to reference points to define the level of catch each year (e.g. Smith, 1994; Wayte and Klaer, 2010; Moore *et al.*, 2013). MSE thus allows exploration of the likely effect of alternative management strategies and the ability of those strategies to satisfy quantifiable management objectives (Smith, 1994).

Punt *et al.* (2005) used MSE to evaluate the relative benefits of alternative harvest strategies to set annual TACs for school and gummy shark (*Mustelus antarcticus*), finding that the uncertainties that most affected performance measures (related to average catches, catch variability, and resource conservation) were the technical interaction between fishing for school and gummy shark, the productivity of the school shark, and the magnitude of tag loss or shark death immediately after tagging. Little *et al.* (2011) used MSE to evaluate a catch- and cpue-based HCR for the southern and eastern scalefish and shark fishery of Australia for situations with limited data, finding that fishery objectives could be achieved reasonably well when target catch was a function of a predefined historical reference period characterized by relatively stable cpue and catches.

The effort needed to conduct an MSE is incomparably greater than required for a PSA. However, it may be possible to conduct an MSE for a representative species to develop an HCR that incorporates decisions about risk tolerance, then use that HCR for species that scored similarly in a PSA. Such stopgap measures may be a practical management approach until data are sufficient for species-specific applications.

In general, we recommend a stepwise approach wherein the model used to assess risk is determined by the data available. Initially, this can be a simple model that requires few data. As more and better data become available, more complex models can be explored in tandem with identifying the types of data that are most crucial for satisfying predefined management objectives through MSE or similar simulation approaches.

When using simple models like PSA to rank species by risk of overfishing, it would be advisable to explore the use of additional measures of vulnerability and compare them to the more traditionally used Euclidean distance. When using demographic models, it is also important to make sure that the life history inputs (growth, mortality, reproduction) correspond to those that would be expected of a population growing at its maximum rate.Finally, we also recommend testing model performance through stepwise construction. The performance of data-poor methods can be assessed

for their ability to recreate results obtained with more data-rich approaches. A simple model can also be built up to a more complex model by adding data that support the next level of complexity. This sequential model building exercise could identify which steps cause model results to diverge, pointing towards aspects of the data or model structure that are important to refine with targeted future research. In addition, simulation testing can help identify applications where data-limited approaches will not be appropriate.

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References

AFMA (Australian Fisheries Management Authority). 2009. Harvest Strategy Framework for the Southern and Eastern Scalefish and Shark Fishery. Australian Government. 25 pp.

Aires-da-Silva, A. M., and Gallucci, V. F. 2007. Demographic and risk analyses applied to management and conservation of the blue shark (*Prionace glauca*) in the North Atlantic Ocean. Marine and Freshwater Resources, 58: 570–580.

Apostolaki, P., Babcock, E. A., and McAllister, M. K. 2006. Contrasting deterministic and probabilistic ranking of catch quotas and spatially and size-regulated fisheries management. Canadian Journal of Fisheries and Aquatic Sciences, 63: 1777–1792.

Baum, J. K., Myers, R. A., Kehler, D. G., Worm, B., Harley, S. J., and Doherty, P. A. 2003. Collapse and conservation of shark populations in the northwest Atlantic. Science, 299: 389–392.

Benton, T. G., and Grant, A. 1999. Elasticity analysis as an important tool in evolutionary and population ecology. Trends in Ecology and Evolution, 14: 467–471.

Bradshaw, C. J. A., Field, I. C., McMahon, C. R., Johnson, G. J., Meekan, M. G., and Buckworth, R. C. 2013. More analytical bite in estimating targets for shark harvest. Marine Ecology Progress Series, 488: 221–2232.

Brandon, J. R., Breiwick, J. M., Punt, A. E., and Wade, P. R. 2007. Constructing a coherent joint prior while respecting biological realism: Application to marine mammal stock assessments. ICES Journal of Marine Science, 64: 1085–1100.

Brewster-Geisz, K., and Miller, T. J. 2000. Management of the sandbar shark, *Carcharhinus plumbeus*: implications of a stage-based model. Fishery Bulletin, 98: 236–249.

Brodziak, J., and Legault, C. M. 2005. Model averaging to estimate rebuilding targets for overfished stocks. Canadian Journal of Fisheries and Aquatic Sciences, 62: 544–562.

Brooks, E. N., Powers, J. E., and Cortés, E. 2010. Analytical reference points for age-structured models: application to data-poor fisheries. ICES Journal of Marine Science, 67: 165–175.

Burgess, G., Beerkircher, L., Cailliet, G. M., Carlson, J. K., Cortés, E., Goldman, K. J., Grubbs, D., *et al.* 2005. Is the collapse of shark populations in the northwest Atlantic and Gulf of Mexico real? Fisheries, 30: 19–26.

Burgman, M. A., Ferson, S., and Akçakaya, H. R. 1993. Risk Assessment in Conservation Biology. Chapman and Hall, London.

Burnham, K. P., and Anderson, D. 2002. Model Selection and Multimodel Inference. Springer-Verlag, New York.

Butterworth, D. S., and Punt, A. E. 1999. Experiences in the evaluation and implementation of management procedures. ICES Journal of Marine Science, 56: 985–998.

Caswell, H. 2001. Matrix Population Models, 2nd edn. Sinauer Associates, Inc., Sunderland, MA. 722 pp.

Chin, A., Kyne, P. M., Walker, T. I., and McAuley, R. 2010. An integrated risk assessment for climate change: analysing the vulnerability of sharks and rays on Australia's Great Barrier Reef. Global Change Biology, 16: 1936–1953.

Conn, P. B. 2010. Hierarchical analysis of multiple noisy abundance indices. Canadian Journal of Fisheries and Aquatic Sciences, 67: 108–120.

Cortés, E. 1999. A stochastic stage-based population model of the sandbar shark in the western North Atlantic. *In* Life in the Slow Lane: Ecology and Conservation of Long-lived Marine Animals, pp. 115–136. Ed. by J. A. Musick. Symposium 23, American Fisheries Society, Bethesda, MD. 265 pp.

Cortés, E. 2002a. Incorporating uncertainty into demographic modeling: application to shark populations and their conservation. Conservation Biology, 16: 1048–1062.

Cortés, E. 2002b. Stock assessment of small coastal sharks in the U.S. Atlantic and Gulf of Mexico. Sustainable Fisheries Division Contribution SFD-01/02-152. National Oceanographic and Atmospheric Administration, Panama City, FL. 133 pp.

Cortés, E., Arocha, F., Beerkircher, L., Carvalho, F., Domingo, A., Heupel, M., Holtzhausen, H., *et al.* 2010. Ecological risk assessment of pelagic sharks caught in Atlantic pelagic longline fisheries. Aquatic Living Resources, 23: 25–34.

Cortés, E., Brooks, E. N., Apostolaki, P., and Brown, C. A. 2006. Stock assessment of dusky shark in the U.S. Atlantic and Gulf of Mexico. Sustainable Fisheries Division Contribution SFD-2006-014, NOAA Fisheries, Miami, FL. 155 pp.

Cortés, E., Brooks, E. N., and Gedamke, T. 2012. Population dynamics, demography, and stock assessment. *In* Biology of Sharks and Their Relatives, 2nd edn, pp. 449–481. Ed. by J. C. Carrier, J. A. Musick, and M. R. Heithaus. CRC Press, Boca Raton, FL. 633 pp.

Cortés, E., Brooks, E. N., and Scott, G. 2002. Stock assessment of large coastal sharks in the U.S. Atlantic and Gulf of Mexico. Sustainable Fisheries Division Contribution SFD-02/03-177. National Oceanographic and Atmospheric Administration, Panama City, FL. 222 pp.

Cortés, E., Heupel, M., Ribera, M., and Simpfendorfer, C. A. 2008. Productivity and susceptibility analysis (ecological risk assessment) of Atlantic sharks. 88th Annual Meeting of the American Society of Ichthyologists and Herpetologists (ASIH), 24th Annual Meeting of the American Elasmobranch Society (AES), Montreal, Canada, July 23–28.

Davies, T. D., and Baum, J. K. 2012. Extinction risk and overfishing: reconciling conservation and fisheries perspectives on the status of marine fishes. Nature, 2: 561.

Draper, D. 1995. Assessment and propagation of model uncertainty. Journal of the Royal Statistical Society Series B, 57: 45–97.

Dulvy, N. K., Baum, J. K., Clarke, S., Compagno, L. J. V., Cortés, E., Domingo, A., Fordham, S., *et al.* 2008. You can swim but you can't hide: the global status and conservation of oceanic pelagic sharks and rays. Aquatic Conservation, 18: 459–482.

Dulvy, N. K., Ellis, R. J., Goodwin, N. B., Grant, A., Reynolds, J. D., and Jennings, S. 2004. Methods of assessing extinction risk in marine fishes. Fish and Fisheries, 5: 255–276.

Dulvy, N. K., and Forrest, R. E. 2010. Life histories, population dynamics, and extinction risks in chondrichthyans. *In* Biology of Sharks and Their Relatives II, 1st edn, pp. 639–679. Ed. by J. C. Carrier, J. A. Musick, and M. R. Heithaus. CRC Press, Boca Raton, FL. 713 pp.

Dulvy, N. K., Jennings, S., Goodwin, N. B., Grant, A., and Reynolds, J. D. 2005. Comparison of threat and exploitation status in Northeast Atlantic marine populations. Journal of Applied Ecology, 42: 883–891.

Fieberg, J., and Ellner, S. P. 2001. Stochastic matrix models for conservation and management: a comparative review of methods. Ecology Letters, 4: 244–266.

Fleming, L. V., Goodman, G., Williams, J., Crook, V., Littlewood, A., Oldfield, T., and Sant, G. 2012. Addendum to JNCC Report No 453 Fish and Multilateral Environmental Agreements: developing a method to identify high risk commercially-exploited aquatic organisms in trade and an analysis of the potential application of

MEAs. Report of an expert review workshop. Aberdeen, Scotland 26–27 September 2011.

Francis, R. I. C. C. 2011. Data weighting in statistical fisheries stock assessment models. Canadian Journal of Fisheries and Aquatic Sciences, 68: 1124–1138.

Francis, R. I. C. C., and Shotton, R. 1997. "Risk" in fisheries management: a review. Canadian Journal of Fisheries and Aquatic Sciences, 54: 1699–1715.

Frisk, M. G., Martell, S. J. D., Miller, T. J., and Sosebee, K. 2010. Exploring the population dynamics of winter skate (*Leucoraja ocellata*) in the Georges Bank region using a statistical catch-at-age model incorporating length, migration, and recruitment process errors. Canadian Journal of Fisheries and Aquatic Sciences, 67: 774–792.

Gedamke, T., Hoenig, J. M., DuPaul, W. D., Musick, J. A., and Gruber, S. H. 2007. Using demographic models to determine intrinsic rate of increase and sustainable fishing for elasmobranchs: pitfalls, advances and applications. North American Journal of Fisheries Management, 27: 605–618.

Gertseva, V. V. 2009. The population dynamics of the longnose skate, *Raja rhina*, in the northeast Pacific Ocean. Fisheries Research, 95: 146–153.

Hayes, C. G., Jiao, Y., and Cortés, E. 2009. Stock assessment of scalloped hammerheads in the western North Atlantic Ocean and Gulf of Mexico. North American Journal of Fisheries Management, 29: 1406–1417.

Hennemuth, R. C., Palmer, J. E., and Brown, B. E. 1990. A statistical description of recruitment in eighteen selected fish stocks. Journal of Northwest Atlantic Fisheries Science, 1: 101–111.

Hilborn, R., and Mangel, M. 1997. The Ecological Detective. Princeton University Press, Princeton, NJ. 315 pp.

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

Hutchings, J. 2001. Conservation biology of marine fishes: perceptions and caveats regarding assignment of extinction risk. Canadian Journal of Fisheries and Aquatic Sciences, 58: 108–121.

ICCAT. 2010. Report of the 2009 porbeagle stock assessments meeting. Collective Volume of Scientific Papers ICCAT, 65: 1909–2005.

ICCAT. 2013. Report of the 2012 shortfin mako stock assessment and ecological risk assessment meeting. Collective Volume of Scientific Papers ICCAT, 69: 1427–1570.

ISO 31000. 2009. Risk Management – Principles and Guidelines. International Organization for Standardization.

IUCN. 2004. Guidelines for using the IUCN Red List categories and criteria. World Conservation Union, Gland, Switzerland.

Jiao, Y., Cortés, E., Andrews, K., and Guo, F. 2011. Poor-data and data-poor species stock assessment using a Bayesian hierarchical approach. Ecological Applications, 21: 2691–2708.

Jiao, Y., Hayes, C., and Cortés, E. 2009. Hierarchical Bayesian approach for population dynamics modelling of fish complexes without species-specific data. ICES Journal of Marine Science, 66: 367–377.

Lack, M., Sant, G., Burgener, M., and Okes, N. 2014. Development of a Rapid Management-Risk Assessment Method for Fish Species through its Application to Sharks: Framework and Results. Report to the Department of Environment, Food and Rural Affairs. DEFRA Contract No. MB0123. 108 pp.

Little, L. R., Wayte, S. E., Tuck, G. N., Smith, A. D. M., Klaer, N., Haddon, M., Punt, A. E., *et al.* 2011. Development and evaluation of a cpue-based harvest control rule for the southern and eastern scalefish and shark fishery of Australia. ICES Journal of Marine Science, 68: 1699–1705.

MacCall, A. D. 2013. Use of the delta method to evaluate the precision of assessments that fix parameter values. Fisheries Research, 142: 56–60.

Mace, P. M., and Doonan, I. J. 1988. A generalized bioeconomic simulation model for fish population dynamics. New Zealand Fisheries Assessment Research Document 88/4.

Manning, M. J., Bromhead, D. B., Harley, S. J., Hoyle, S. D., and Kirby, D. S. 2009. The feasibility of conducting quantitative stock assessments for key shark species and recommendations for providing preliminary advice on stock status in 2010. Western and Central Pacific Fisheries Commission. WCPFC-SC5-2009/EB-WP-08. 32 pp.

Matsuda, H., Takenaka, Y., Yahara, T., and Uozumi, Y. 1998. Extinction risk assessment of declining wild populations: the case of the southern bluefin tuna. Research in Population Ecology, 40: 271–278.

Maunder, M. N., and Punt, A. E. 2004. Standardizing catch and effort data: a review of recent approaches. Fisheries Research, 70: 141–159.

Maunder, M. N., and Punt, A. E. 2013. A review of integrated analysis in fisheries stock assessment. Fisheries Research, 142: 61–74.

McAllister, M. K., and Babcock, E. A. 2006. Bayesian surplus production model with the Sampling Importance Resampling algorithm (BSP): a user's guide. http://www.iccat.int/en/AssessCatalog.htm (last accessed May 2006).

McAllister, M. K., and Kirkwood, G. P. 1998a. Bayesian stock assessment: a review and example application using the logistic model. ICES Journal of Marine Science, 55: 1031–1060.

McAllister, M. K., and Kirkwood, G. P. 1998b. Using Bayesian decision analysis to help achieve a precautionary approach to managing newly developing fisheries. Canadian Journal of Fisheries and Aquatic Sciences, 55: 2642–2661.

McAllister, M. K., Pikitch, E. K., and Babcock, E. A. 2001. Using demographic methods to construct Bayesian priors for the intrinsic rate of increase in the Schaefer model and implications for stock rebuilding. Canadian Journal of Fisheries and Aquatic Sciences, 58: 1871–1890.

McAllister, M. K., Pikitch, E. K., and Babcock, E. A. 2008. Why are Bayesian methods useful for the stock assessment of sharks? *In* Sharks of the Open Ocean, pp. 351–368. Ed. by M. D. Camhi, E. K. Pikitch, and E. A. Babcock. Blackwell Publishing, Oxford. 502 pp.

McAuley, R. B., Simpfendorfer, C. A., and Hall, N. G. 2007. A method for evaluating the impacts of fishing mortality and stochastic influences on the demography of two long-lived shark stocks. ICES Journal of Marine Science, 64: 1710–1722.

Methot, R. D., and Wetzel, C. R. 2013. Stock synthesis: a biological and statistical framework for fish stock assessment and fishery management. Fisheries Research, 142: 86–99.

Meyer, R., and Millar, R. B. 1999. BUGS in Bayesian stock assessments. Canadian Journal of Fisheries and Aquatic Sciences, 56: 1078–1086.

Moore, J. E., Curtis, K. A., Lewison, R. L., Dillingham, P. W., Cope, J. M., Fordham, S. V., Heppell, S. S., *et al.* 2013. Evaluating sustainability of fisheries bycatch mortality for marine megafauna: a review of conservation reference points for data-limited populations. Environmental Conservation, 40: 329–344.

Musick, J. 1999. Criteria to define extinction risk in marine fishes: the American Fisheries Society initiative. Fisheries, 24: 6–14.

Myers, R. A., Baum, J. K., Shepherd, T. D., Powers, S. D., and Peterson, C. H. 2007. Cascading effects of the loss of apex predatory sharks from a coastal Ocean. Science, 315: 1846–1850.

Myers, R. A., Mertz, G., and Fowlow, P. S. 1997. Maximum population growth rates and recovery times for Atlantic cod, *Gadus morhua*. Fisheries Bulletin, 95: 762–772.

NEFSC. 2013. 55th Northeast Regional Stock Assessment Workshop (55th SAW) Assessment Report. US Department of Commerce, Northeast Fisheries Science Center Reference Document 13–11. 845 pp.

NEFSC. 1998. Report of the shark evaluation workshop. National Oceanographic and Atmospheric Administration. Panama City, Florida. 109 pp.

NMFS. 2010. Southeast Data, Assessment and Review (SEDAR) 21: Stock assessment report—Sandbar, dusky and Gulf of Mexico blacknose shark. NMFS, North Charleston, SC.

NMFS. 2012. Southeast Data, Assessment and Review (SEDAR) 29: stock assessment report—Gulf of Mexico blacktip shark. NMFS, North Charleston, SC.

NMFS. 2013a. Southeast Data, Assessment and Review (SEDAR) 34: stock assessment report—Atlantic sharpnose shark. NMFS, North Charleston, SC.

NMFS. 2013b. Amendment 5a to the 2006 Consolidated Atlantic Highly Migratory Species Fisheries Management Plan. Office of Sustainable Fisheries, Silver Spring, MD.

Patrick, W. S., Morrison, W., Nelson, M., and Gonzalez Marrero, R. L. 2013. Factors affecting management uncertainty in U.S. fisheries and methodological solutions. Ocean and Coastal Management, 71: 64–72.

Patrick, W. S., Spencer, P., Link, J., Ormseth, O., Cope, J., Field, J., Kobayashi, D., *et al.* 2009. Use of productivity and susceptibility indices to determine stock vulnerability, with example applications to six U.S. fisheries. NOAA Technical Memorandum NMFS-F/SPO 101. 90 pp.

Patrick, W. S., Spencer, P., Link, J., Ormseth, O., Cope, J., Field, J., Kobayashi, D., *et al.* 2010. Using productivity and susceptibility indices to assess the vulnerability of United States fish stocks to overfishing. Fishery Bulletin, 108: 305–322.

Porch, C. E. 2003a. A preliminary assessment of Atlantic white marlin (*Tetrapturus albidus*) using a state-space implementation of an age-structured production model. Collective Volume of Scientific Papers, International Commission for the Conservation of Atlantic Tunas, 55: 559–527.

Porch, C. E. 2003b. Pro-2Box v.2.01 User's Guide. 40 pp.

Porch, C. E., Ecklund, A. M., and Scott, G. P. 2006. A catch-free stock assessment model with application to goliath grouper (*Epinephelus itajara*) off southern Florida. Fishery Bulletin, 104: 89–101.

Pribac, F., Punt, A. E., Taylor, B. L., and Walker, T. I. 2005. Using length, age and tagging data in a stock assessment of a length selective fishery for gummy shark (*Mustelus antarcticus*). Journal of Northwest Atlantic Fishery Science, 35: 267–290.

Punt, A. E. 2000. Extinction of marine renewable resources: a demographic analysis. Population Ecology, 42: 19–27.

Punt, A. E., Butterworth, D. S., and Penney, A. J. 1995. Stock assessment and risk analysis for the South Atlantic population of albacore *Thunnus alalunga* using an age-structured production model. South African Journal of Marine Science, 16: 287–310.

Punt, A. E., and Walker, T. I. 1998. Stock assessment and risk analysis for the school shark *Galeorhinus galeus* (Linnaeus) off southern Australia. Marina and Freshwater Research, 49: 719–731.

Punt, A. E., Pribac, F., Taylor, B. L., and Walker, T. I. 2005. Harvest strategy evaluation for school and gummy shark. Journal of Northwest Atlantic Fishery Science, 35: 387–406.

Quinn, T. J., and Deriso, R. B. 1999. Quantitative Fish Dynamics. Oxford University Press, New York. 542 pp.

Restrepo, V. R., Thompson, G. G., Mace, P. M., Gabriel, W. L., Low, L. L., MacCall, A. D., Methot, R. D., *et al.* 1998. Technical guidance on the use of precautionary approaches to implementing National Standard 1 of the Magnuson–Stevens Fishery Conservation and Management Act. NOAA Technical Memorandum NMFS-F/SPO 31. 54 pp.

Rice, J., and Harley, S. J. 2012. Stock assessment of oceanic whitetip sharks in the western and central Pacific Ocean. Western and Central Pacific Fisheries Commission. WCPFC-SC8-2012/SA-WP-06. 53 pp.

Rosenberg, A. A., Acosta, A., Babcock, E., Harrington, J., Hobday, A., Mogensen, C. B., O'Boyle, R., *et al.* 2009. Use of Productivity-Susceptibility Analysis (PSA) in setting annual catch limits for US fisheries: a workshop report. MRAG Americas report to the Lenfest Ocean Program, Washington, DC. 18 pp.

Rosenberg, A. A., and Restrepo, V. R. 1994. Uncertainty and risk evaluation in stock assessment advice for U.S. marine fisheries. Canadian Journal of Fisheries and Aquatic Sciences, 51: 2715–2720.

Sant, G., Goodman, G., Crook, V., Lack, M., and Oldfield, T. E. E. 2012. Fish and Multilateral Environmental Agreements: developing a method to identify high risk commercially-exploited aquatic organisms in trade and an analysis of the potential application of MEAs. JNCC Report No 453.

Schaefer, M. B. 1954. Some aspects of the dynamics of populations important to the management of commercial marine fisheries. Inter American Tropical Tuna Commission Bulletin, 2: 247–285.

Schnute, J. T. 1991. The importance of noise in fish population models. Fisheries Research, 11: 197–223.

Shertzer, K. W., Prager, M. H., Vaughan, D. S., and Williams, E. H. 2008. Fishery Models. *In* Population Dynamics. Vol. 2 of Encyclopedia of Ecology, pp. 1582–1593. Ed. by S. E. Jorgensen, and B. D. Fath. Elsevier, Oxford.

Simpfendorfer, C. A. 1999. Demographic analysis of the dusky shark fishery in southwestern Australia. *In* Life in the Slow Lane: Ecology and Conservation of Long-Lived Marine Animals, pp. 149–160. Ed. by J. A. Musick. Symposium 23, American Fisheries Society, Bethesda, MD. 265 pp.

Simpfendorfer, C. A., Donohue, K., and Hall, N. G. 2000. Stock assessment and risk analysis for the whiskery shark (*Furgaleus macki* (Whitley)) in south-western Australia. Fisheries Research, 47: 1–17.

Smith, A. D. M. 1994. Management strategy evaluation – the light on the hill. *In* Population Dynamics for Fisheries Management, pp 249–253. Ed. by D. A. Hancock. Australian Society for Fish Biology Workshop Proceedings, Perth, 24–25 August, 1993.

Spiegelhalter, D., Thomas, A., and Best, N. 2000. WinBUGS User Manual Version 1.4. MRC Biostatistics Unit, Cambridge, UK.

Stevens, J. D., Bonfil, R., Dulvy, N. K., and Walker, P. A. 2000. The effects of fishing on sharks, rays, and chimaeras (chondrichthyans), and the implications for marine ecosystems. ICES Journal of Marine Science, 57: 476–494.

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

Stobutzki, I. C., Miller, M. J., Heales, D. S., and Brewer, D. T. 2002. Sustainability of elasmobranchs caught as bycatch in a tropical prawn (shrimp) trawl fishery. Fishery Bulletin, 100: 800–821.

Wayte, S. E., and Klaer, N. L. 2010. An effective harvest strategy using improved catch-curves. Fisheries Research, 106: 310–320.

Zhou, S., Milton, D. A., and Fry, G. C. 2012. Integrated risk analysis for rare marine species impacted by fishing: sustainability assessment and population trend modelling. ICES Journal of Marine Science, 69: 271–280.

Zhou, S., Smith, A. D. M., and Fuller, M. 2011. Quantitative ecological risk assessment for fishing effects on diverse non-target species in a multi-sector and multi-gear fishery. Fisheries Research, 112: 168–178.

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Contribution to the Themed Section: 'Risk Assessment'

Original Article

A hub and spoke network model to analyse the secondary dispersal of introduced marine species in Indonesia

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Indonesia is a biodiversity hotspot threatened with new introductions of marine species. As with many countries, Indonesia has a stratified shipping network of international ports linked to a large suite of domestic ports. We developed a hub and spoke network model to examine the risk associated with the secondary transfer of introduced marine species from the port hub of Tanjung Priok in Jakarta Bay to the 33 Indonesian provinces (including other ports in the Jakarta province). An 11-year shipping dataset was used (vessel next port of call records for maritime vessels that originated in Jakarta Bay and that remained in domestic waters) to derive a province ranking of vulnerability. Fifteen provinces represented almost 94% of the traffic frequency, with East Java and Jakarta provinces dominating. All urban provinces featured within the top seven highest frequency traffic provinces. Traffic patterns reflect an intra-coastal reliance on shipping, with traffic frequency decreasing with distance from Jakarta Bay. Provinces were regionalized into three categories (Lampung to East Java, Makassar Straits, and Malacca Straits) each with different vulnerabilities based on their values.

Keywords: dispersal, non-indigenous species, risk management, risk model, shipping, vector, vulnerability.

Introduction

Once an introduced species establishes within a port, management shifts its focus from prevention (e.g. Hewitt *et al.*, 2004a) to control, eradication, and management (e.g. Wotton and Hewitt, 2004; Campbell, 2008). Of significant concern is that once established, an introduced marine species can continue to disperse (dispersal via natural or secondary human-mediated transfer; Carlton and Hodder, 1995; Carlton, 2001; Floerl and Inglis, 2005; Wyatt *et al.*, 2005; Minchin *et al.*, 2006; Ruiz *et al.*, 2011) to new regions within domestic borders, leading to the need for management to expand its focal range. This spread effect can be a drain on resources as managers attempt to make practical decisions about potential loss of values caused by the spread of the introduced species (Johnson *et al.*, 2001; Ashton *et al.*, 2006).

To proactively manage the secondary dispersal of introduced species, biosecurity managers can implement marine vessel traffic analyses (pathway connection and vector strength) that examine vessel movements between ports within domestic borders. From these analyses, hub and spoke network models can be created to identify potential risky pathways that may require "future watch" activities (vigilant surveillance of a pathway for species introductions) or more engaged management. These models can be useful in regions where limited species data exist but where shipping patterns are well known. To test the utility of hub and spoke network models for introduced marine species, we examined Jakarta Bay (Port of Tanjung Priok), Indonesia.

The Port of Tanjung Priok is the largest port in Indonesia and has strong vector connections with the major port hub cities of Singapore and Tanjung Pelepas (Malaysia; Azmi, 2010). Tanjung Priok is within the Coral Triangle Initiative region (http://www.cti-secretariat.net/) that focuses on conserving biodiversity, developing food security, and establishing a sustainable future for the region. Domestic shipping from other Indonesian provinces constitutes the greatest amount of ship traffic into this port (62% of port calls; Azmi, 2010).

Shipping patterns in the Port of Tanjung Priok follow an intra-coastal transport model (Lee *et al.*, 2008); being connected to

major port hub cities and itself acting as a hub, gateway, or feeder port to connect smaller coastal ports and towns (spokes). These strong traffic connections suggest a high likelihood that introduced marine species that enter Tanjung Priok from Singapore (or elsewhere) can then be spread via secondary dispersal along the Indonesian archipelago. In this context, the pathway and vector strengths that operate on a domestic level become a high priority management target.

Secondary dispersal of species can occur by natural methods (e.g. rafting); however, our focus is on vector strengths associated with vessel movements because they are manageable in a biosecurity context. From a natural dispersal perspective, Indonesia is an Archipelago that consists of 17 000 islands that are connected by waterways, with a total sea area of 6279 million km^2 (http://www.bakosurtanal.go.id/bakosurtanal/peluncuran-kapal-survei-jenis-katamaran-km-tanjungperak-bakosurtanal-pantai-marina-ancol-25-maret-201, accessed 19 September 2012). Understanding the water movement patterns, corridors, and connections in the archipelago is complex and costly. Economic and social connections across the archipelago are maintained through a heavy reliance on ships for domestic and inter-islands transportation (Gurning, 2008). There are at least 1245 ports spread across 33 provinces, of which 645 ports are managed by the government and PELINDO (Indonesia Port Corporation; Ministry of Transportation, 2010). The remaining 500 ports are managed and owned by private companies (http://www.bps.go.id/aboutus.php?glos=1&ist=1&var=P&cari=&kl=4, accessed 19 September 2012).

Port locations range from highly urbanized areas to locations near sensitive, high-value areas, such as biodiversity hot spots and marine protected areas. Although the natural connectedness is complex, the management of shipping already occurs; thus focusing on ship-related transport of species will potentially reduce biosecurity management costs, while bringing some form of introduced marine species management into the region.

In Indonesia, the prevention of species introduction and prevention of pest transfer for plant, animals, fish, and other organism is mandated by *UNDANG-UNDANG REPUBLIK INDONESIA NOMOR 16, TAHUN 1992* (Republic of Indonesia Regulation No. 16, 1992) that mentions all organisms that might be pests, with its carriers (animals, plants, or other materials) that are subject to quarantine (http://www.bkipm.kkp.go.id/files/regulasi/Law%20of%20the%20Republic%20of%20Indonesia%20No%2016%20of%201992.pdf). For aquatic organisms, the above regulations are strengthened by *PERATURAN MENTERI KELAUTAN DAN PERIKANAN NOMOR: PER. 5/MEN/2005* (Ministry of Marine and Fishery Affairs Regulation No. 5, 2005; http://www.bkipm.kkp.go.id/files/regulasi/2.%20PERMEN%20KP%20NO.%2005%20TAHUN%202005.pdf), which recognizes all media as the carrier of diseases or pests for aquatic organisms and thus these are subject to quarantine. However, the implementation of this regulation is limited to the intentional transfer of aquatic organisms such as fish species and fish disease. The quarantine for aquatic organism is managed under the Ministry of Marine and Fishery Affairs (http://www.kkp.go.id/en/). Like many countries, this system reflects a proactive, outward focus on intentional introductions. Expanding to focus on unintentional introductions is the next step for Indonesia to protect its borders and post-border areas from the threats of unintentional non-indigenous species.

Hub and spoke network models are a proven method of analysing transport pathways/corridors and strengths. They were initially developed and applied to airline industries and have since been applied to shipping, logistic delivery, and other transportation activities (e.g. Aykin, 1995; Bendall and Stent, 2001; Bryan and O'Kelly, 2005; Hsu and Hsieh, 2007; Imai *et al.*, 2009), as well being applied to human health services to examine the epidemiology of disease spread and healthcare management (Richards *et al.*, 1997; Sibthorpe *et al.*, 2005). The epidemiological aspect of the models makes them ideal for risk assessment within a biosecurity context. The vectoring of introduced aquatic species has been conceptualized in a hub and spoke network context (e.g. Carlton, 1996; Johnson *et al.*, 2005), yet few have created or used hub and spoke network models to examine realized vector connectivity (except see Lavoie *et al.*, 1999; Muirhead and MacIsaac, 2004).

To examine this in a marine context, we developed a hub and spoke network model to assess the strength of transport pressure (frequency of maritime vessel transfers) from the Port of Tanjung Priok (the hub, gateway, or feeder port) to other domestic Indonesian ports (spokes or outports). The model provides an assessment of possible secondary dispersal of introduced marine species within the Indonesian region, recognizing that this is limited to the Port of Tanjung Priok as the single point of entry into the Indonesian domestic system.

Based on the model outcomes, vulnerable provinces or regions are identified. Although this paper is focused on Indonesia, the model can be applied to efficiently analyse the domestic transfer of introduced marine species in other countries, especially when species data may be limited, but shipping strength is known. For example, this model is currently being used to examine both international and domestic connections for ports in Australia and the Galapagos Islands (Campbell *et al.*, 2013).

Methods

We used an 11-year (1999–2009) shipping dataset purchased from the Lloyd's Maritime Intelligence Unit (Azmi, 2010). The records of next port of call (NPOC) for ships that departed the Port of Tanjung Priok provide the pathway connections and vector strength to determine the dispersal patterns. Due to the large number of Indonesian ports, the analysis was undertaken at the level of province (Figure 1), which is the most likely management level considering jurisdictional and political boundaries. Each NPOC was assigned membership to one of the 33 Indonesian provinces [excluding the province of Jakarta Special Capital Region (i.e. the hub); $n = 32$]. The location of each province and its distance from Jakarta Bay is shown in Figure 1, with Table 1 presenting the name of each province and its sequential number code.

The dispersal strength was assessed as the percentage of ships that departed the Port of Tanjung Priok and had a domestic NPOC. The result was totalled for each province, to derive the province ranking of vulnerability to secondary spread of introduced marine species from the Port of Tanjung Priok.

Results

Between 1999 and 2009, 74 635 vessel departures were recorded from the Port of Tanjung Priok, with 51 214 records (68.6%) indicating a domestic (Indonesian) NPOC. Almost all of the domestic records (99.7%, $n = 51\,085$) could be assigned to individual provinces in Indonesia, with the 129 (0.3%) remaining unassigned due to the use of Indonesia as an unspecified location (i.e. a failure of the ships record keeping to accurately record the NPOC). The unassigned NPOC records were excluded from further analyses.

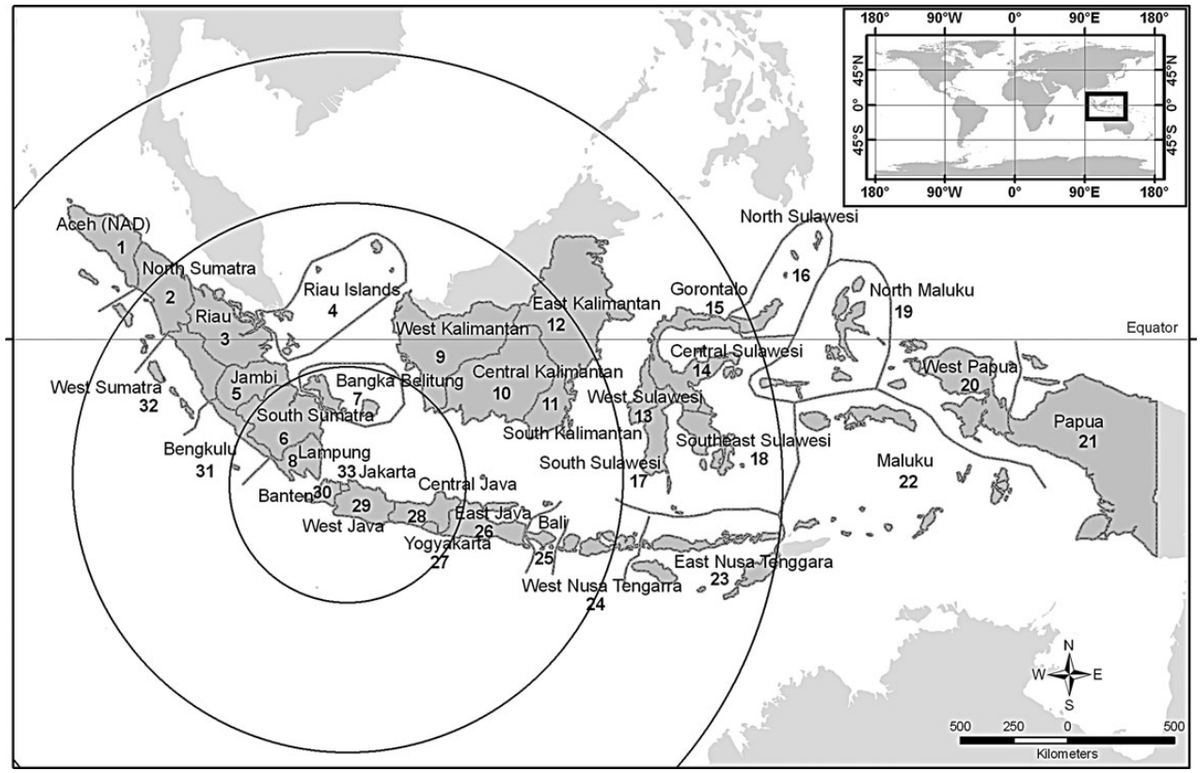

Figure 1. Provinces of the Republic of Indonesia (modified from http://www.enotes.com/topic/Provinces_of_Indonesia) illustrating the distance to each province. Provinces are numbered from 1 to 32. Radial distances are at intervals of 650 km from Jakarta.

Domestic trade from the Port of Tanjung Priok averaged 4655.8 ($\pm$1857.0) vessels per annum with fluctuation between 1962 vessels in 2002 and 8424 vessels in 2009 (Figure 2). The vessels were primarily commercial (23 003 carriers, 6302 tankers, 2059 RoRo vessels, and 6 others), non-trading (2078 barges, 1280 tugs, 138 dredges, 38 research vessels, and 50 others), and passenger vessels (Figure 3; Supplementary Table S1).

The strength of pressure that the Port of Tanjung Priok exerts to each Indonesian province is illustrated in Figure 4 (see also Supplementary Table S2). This information was then used to develop a hub and spoke network model (Figure 4). The model describes the relationship (based on dispersal strength) of the likely secondary dispersal of introduced marine species from the Port of Tanjung Priok to the remaining 32 Indonesian provinces.

East Java was the major province of maritime vessel trade (14.7%) from the Port of Tanjung Priok (Table 1; Figure 4). The second largest trading pattern (13.8% of trade) occurred within Jakarta itself (larger Port of Tanjung Priok trading with smaller ports in this province; Table 1; Figure 4). Together, these two provinces experienced the most significant amount of vessel transits from the Port of Tanjung Priok, with visit frequencies being almost double that of the next five provinces of Central Java (7.4%), North Sumatra (7.2%), East Kalimantan (6.6%), Lampung (6.5%), and West Kalimantan (5.6%; Table 1; Figure 4). Seven other provinces (Riau Islands, Riau, Banten, Bangka-Belitong, South Kalimantan, South Sumatra, and West Sumatra) accounted for more than 1000 ships visits during this period. The remaining provinces each had $<$1000 vessel visits during the study period (Table 1; Figure 4). These 15 provinces made up 93.6% of the domestic maritime vessel traffic that originated from the Port of Tanjung Priok.

Among the seven provinces with the greatest domestic trade (62% of trade) from the Port of Tanjung Priok, three provinces (Jakarta, Central Java, and Lampung) occur within a radius of 650 km, which is the closest radius to the Port of Tanjung Priok. East Java is the only major shipping frequency province that occurred at a radius of 1300 km, and the remaining three provinces (East Kalimantan, North Sumatera, and South Sulawesi) occur within a radius of 1950 km from the Port of Tanjung Priok. Thus, the pattern of vessel trade suggests a decreasing trade connection (and therefore dispersal strength) with distance from the hub. This decreasing connection may be reflected in decreasing port vulnerability.

Discussion

A hub and spoke network model was developed to analyse the secondary dispersal of introduced marine species from the Port of Tanjung Priok to other domestic ports in Indonesia that are located within the remaining 32 provinces. Of the seven provinces that constitute the majority 62% of trade connectivity, all (except East Kalimantan) are the main urban areas of Indonesia, with a combined population density of 108 902 586 (http://www.bps.go.id/tab_sub/view.php?tabel=1&daftar=1&id_subyek=12&notab=1, accessed 19 September 2012).

Table 1. The Indonesian provinces with their sequential number code that is used in Figure 1.

Sequential number code	Indonesian province	"Region" membership	Number of ports	Number of ships from Tanjung Priok	Per cent of total (%)
1	Aceh	–	8	79	0.15
2	North Sumatra	3	6	3 665	7.16
3	Riau	3	13	2 177	4.25
4	Riau Islands	3	12	2 396	4.68
5	Jambi	3	3	318	0.62
6	South Sumatra	3	5	1 682	3.28
7	Bangka Belitong	3	7	1 856	3.62
8	Lampung	1	5	3 331	6.50
9	West Kalimantan	3	5	2 883	5.63
10	Central Kalimantan	–	4	325	0.63
11	South Kalimantan	2	13	1 759	3.43
12	East Kalimantan	2	17	3 371	6.58
13	West Sulawesi	2	1	6	0.01
14	Central Sulawesi	–	5	114	0.22
15	Gorontolo	–	3	15	0.03
16	North Sulawesi	–	3	460	0.90
17	South Sulawesi	2	5	3 021	5.90
18	Southeast Sulawesi	–	4	110	0.21
19	North Maluku	–	5	39	0.08
20	West Papua	–	8	225	0.44
21	Papua	–	7	295	0.58
22	Maluku	–	8	102	0.20
23	East Nusa Tenggara	–	9	54	0.11
24	West Nusa Tenggara	–	4	55	0.11
25	Bali	–	4	117	0.23
26	East Java	1	10	7 527	14.70
27	Yogyakarta	–	0	0	0.00
28	Central Java	1	4	3 773	7.37
29	West Java	–	4	425	0.83
30	Banten	1	10	2 014	3.93
31	Bengkulu	–	3	411	0.80
32	West Sumatra	–	1	1 358	2.65
33	Jakarta	1	4	7 085	13.83

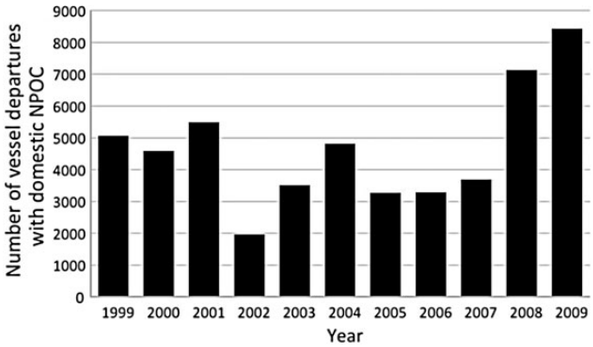

Figure 2. Annual domestic trade from the port of Tanjung Priok between 1999 and 2009 (Lloyds MIU dataset).

East Java, Lampung, and Central Java received the largest volume (number) of ship visits from the Port of Tanjung Priok and are in proximity to Jakarta Bay (Figure 4). Hence, these provinces are more likely to experience immediate transfer of introduced marine species from the hub if an introduction occurs. This is further emphasized as vectors such as ballast water show a correlation between transit length and viability of propagules, with shorter transit times (such as <15 days), resulting in healthier species with an increased probability of survival upon release into a new port area (Smith *et al.*, 1999; Barry *et al.*, 2008; Simkanin *et al.*, 2009).

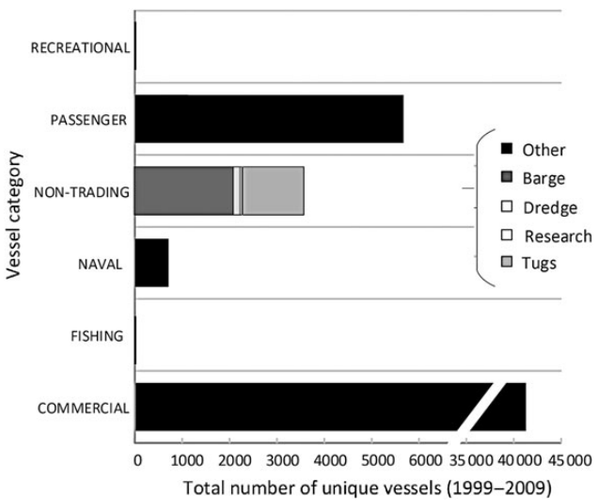

Figure 3. Number of unique vessels (by vessel category) departing the port of Tanjung Priok between 1999 and 2009 (Lloyds MIU dataset).

In contrast, the shipping-related trends for biofouling species are less clear. Typically, slower vessels have less exposure to shear forces and wave exposure, which reduces transit stress and may encourage the development of biofouling (Hewitt *et al.*, 2007; Coutts *et al.*,

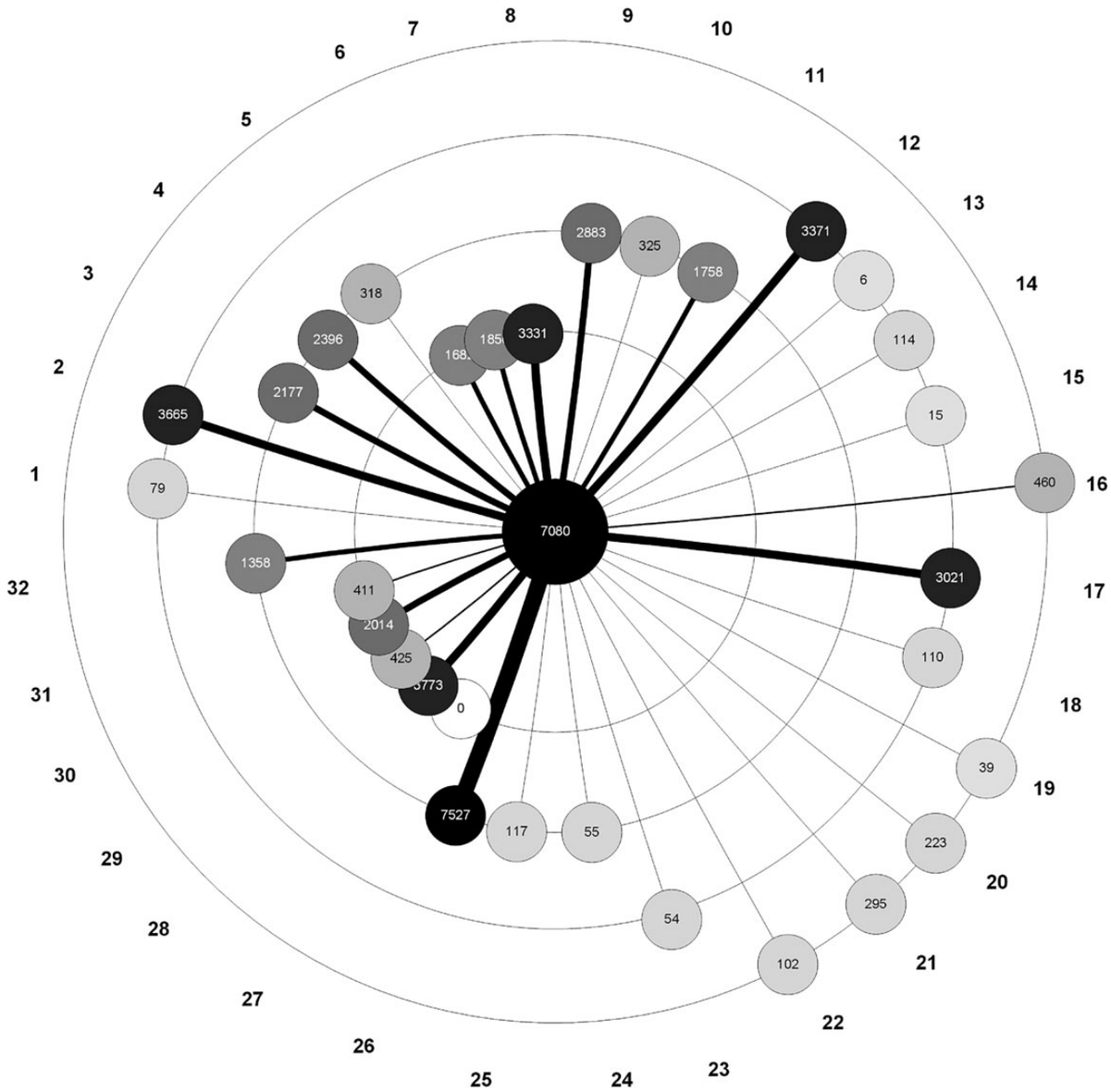

Figure 4. Hub and spoke network model for the Port of Tanjung Priok and the Indonesian provinces. Note: the numbers around the outside are the province codes from Table 1; the line thickness and colour of province markers (small circles) represent the strength of Jakarta Bay pressure and the number of vessels are represented within the province markers; and the distances between the "hub" (the Port of Tanjung Priok) and the "spokes" (the provinces) are at radial intervals of 650 km. Note that the province of Jakarta is represented as the hub (central circle).

2010a, b). Ships with maximum speeds of 18 knots or more decrease the risk of fouling community settlement, while ships with speeds below <10 knots are more likely to translocate larger numbers of fouling communities (Davidson *et al.*, 2008; Coutts *et al.*, 2010a; Hopkins and Forrest, 2010). In contrast, research by Davidson *et al.* (2009) on commercial hulls in California has suggested that vessels travelling at 15 knots (vs. 21–24 knots) are also capable of developing biofouling communities. Additionally, niche areas associated with decreased hydrodynamic exposure such as sea chests, thrusters, stern regions including rudder, propeller, and propeller shaft represent areas of high fouling (e.g. Coutts *et al.*, 2003; Coutts and Taylor, 2004; Coutts and Dodgshun, 2007; Davidson *et al.*, 2008; Frey *et al.*, 2014) that may obviate the influence of vessel speed on reducing biofouling risk (Hewitt *et al.*, 2011).

Regional shipping has shorter travel distances and hence shorter travel duration, which is also reported to increase levels of biofouling (Skerman, 1960; Coutts and Taylor, 2004; Davidson *et al.*, 2008; Coutts *et al.*, 2010a; Hopkins and Forrest, 2010). Yet, work by Lewis *et al.* (2004) also illustrates that viable fouling can survive prolonged voyages. The implications for Indonesia is that the short travel distances and potentially slower vessel speeds for vessels that are used within intra-coastal shipping in this region (personal observations) would suggest a higher likelihood of viable biofouling and ballast water communities arriving in spoke ports.

Management implications for Indonesia

Indonesia's geography [large Exclusive Economic Zone (seventh largest in the world) and spatial spread] and its large number of ports make biosecurity management difficult, especially the post-border containment and control of spread of introduced marine species. Furthermore, the provinces are spread across similar physical environments (tropical ecosystems). The connectedness between the Indonesian provinces is strong but follows a pattern of decreasing connectedness with distance from the hub port, which is a common intra-coastal shipping pattern (Azmi, 2010). Clearly, the large urban areas (East Java, Jakarta, Central Java, North Sumatra, South Sulawesi, and Lampung) have a higher connectivity with the Port of Tanjung Priok, and this connectivity needs to be managed to reduce the likelihood of introduced marine species transfers.

What is limited in the Indonesian context is knowledge about the introduced marine species present in the port. However, shipping data are readily available and hence a useful biosecurity information, in this context. There are several options for managing introduced marine species in a post-border situation such as the one described here. These options include:

- Attempting to prevent introduced marine species at a pre-border stage. As described in Azmi (2010) and Azmi *et al.* (in press), risk assessment tools such as bioregion pathway and species exposure risk models can be used to further understand the international shipping trends and subsequent risks to Indonesia. Biosecurity effort focused on high-risk pathways is often considered the most viable. Yet, it must be recognized that borders are leaky (e.g. Hewitt *et al.*, 2004a; Wotton and Hewitt, 2004), that people's attitudes and behaviours are linked to incursions (e.g. Bewsell *et al.*, 2012; Cliff and Campbell, 2012) and hence a post-border biosecurity strategy also needs to be in place.
- Establishing sustained surveillance systems to ensure early detection of introduced marine species (Hewitt *et al.*, 2004a), which will facilitate eradication attempts and trigger control measures (Simberloff, 2000; Inglis *et al.*, 2006). For example, the black-striped mussel was detected in Darwin Harbour (where it was not present 6 months earlier) and within 1 week, a successful eradication attempt had been started (Willan *et al.*, 2000). Coupled with early detection is the need to delimit the distributional extent and density of newly detected species and to determine that a species is new to an area (Willan *et al.*, 2000; Kean *et al.*, 2008). All of these factors require baseline information that is then updated and extended when required.
- Attempt eradication and/or control when an incursion occurs. These are often the first options of management choice when dealing with introduced species (Wittenberg and Cock, 2005); however, eradication or control can be difficult if capacity and/or resources are limited as they can be costly to implement, and might result in more problems (Wittenberg and Cock, 2005; Locke and Hanson, 2009). To consider this option, investment and capacity building needs to occur.
- Implementing trading restrictions that take into account the trade (bioregion pathway) and species exposure patterns. Under the WTO, trade restrictions need to be based on science (e.g. Campbell *et al.*, 2009) and need to be seen as being justified. Trade restrictions can be difficult when the social implications (social welfare; e.g. Cook and Fraser, 2008) of shipping (i.e. the provision of goods and services provided and associated with shipping) might outweigh the biological implications of an introduced marine species. This is particularly relevant in situations where food security is involved (Hewitt and Campbell, 2007; Cook and Fraser, 2008).

Managing vectors and pathways are common biosecurity measures to prevent species introduction (Hewitt *et al.*, 2004a, b; Wotton and Hewitt, 2004; Hewitt and Campbell, 2007; Minchin, 2007; Barry *et al.*, 2008; Campbell, 2008, 2009; Kean *et al.*, 2008; Forrest *et al.*, 2009; Campbell and Hewitt, 2011). In this research, the post-border mechanisms to manage introduced marine species are similar to the pre-border measures that can be undertaken: that is managing the ships that enter the ports. Floerl and Inglis (2005) have shown that using antifouling paints on boat hulls can be an effective tool to prevent secondary spread of introduced species caused by recreational vessels within Australia. Yet, this is costly and places the burden of management onto the ship/boat owners.

If antifouling methods were applied in Indonesia then the burden would be placed upon the ships transiting from the Port of Tanjung Priok to other provinces. This action would most likely result in higher prices for the goods and services being delivered, which consequently would result in the public bearing this cost. This in turn would potentially result in food-insecurity within the region as trade becomes more expensive.

Transportation development in Indonesia follows an Asian hub port city consolidation model (Lee *et al.*, 2008), with limited hinterland penetration, and a large number of islands that require connections. Any management restrictions need to be cognizant of the need for a continued maritime linkage via the current intra-coastal shipping trade. The Asian hub port city consolidation model may evolve through time, but due to Indonesia being an archipelago, it is unlikely that inland transportation routes will have much impact. Thus, it is unlikely that maritime shipping pressures will be reduced from what is currently experienced.

Similarly, managing all ships in all Indonesian ports would be an exhaustive and resource intensive effort, given many ports and their spatial distribution. Prioritization needs to occur. Typically, the ports that receive the highest number of shipping transits are a priority when managing introduced marine species (Forrest *et al.*, 2009). However, the priority should also address the values associated with each port and/or province that need to be protected. This ensures that environmental, social, human health, and economic values can be examined and compared to ensure that decisions are based on what is important for the people and the country of Indonesia.

The values among Indonesian provinces that experience more contact with Jakarta Bay are varied. The top 15 provinces (>1000 ship visits from the Port of Tanjung Priok during the study period) can be grouped into three "regions" based on their locations and the values that they have in common (Table 1). Region 1 is the coastal area from Lampung (south end Sumatra Island) to East Java, especially *Pantai Utara Jawa (Pantura)* or the North coast of Java. Five provinces are included within this area: Banten, Jakarta, Central Java, East Java, and Lampung. These provinces have extensive aquaculture and mariculture activities along these coastlines (personal observations) and hence tend to focus more on economic values.

Region 2 comprises the provinces located around the Makassar Straits and Bone Gulf. Three of the top 15 provinces fall within this region (East Kalimantan, South Kalimantan, and South

Sulawesi). These three provinces occur within proximity, with a series of protected areas such as Wakatobi, Taka Bonerate, and Derawan that are internationally recognized as important biodiversity hotspots (e.g. Mittermeier *et al.*, 1998; Hoeksema, 2000; Bellwood and Hughes, 2001; Roberts *et al.*, 2002; Koh and Wilcove, 2007) or biodiversity hotspots under the Coral Triangle Initiatives (http://www.cti-secretariat.net/news-a-updates/cti-updates/123-the-solomon-star, accessed 19 September 2012). Derawan in Berau district (East Kalimantan) has the second highest hard coral biodiversity in the world which is included in the Sulu-Sulawesi ecoregion (Ambarwulan, 2010), and thus, environmental values are important in this region.

Furthermore, the CTI is developing conservation programmes for conservation based Marine Protected Areas where ecotourism is one of the key elements (http://www.cti-secretariat.net/about-cti/about-cti, accessed 19 September 2012); and ecotourism relies on good environmental quality (Tisdell, 2005). Therefore, protection of biodiversity in these areas is paramount; however, we do note that mining (gas, oil, and coal) is one of the most important economic sources in this region.

The third region consists of the provinces around the Malacca Straits, which includes the provinces of North Sumatra, Riau, Riau Islands, Bangka-Belitong, and West Kalimantan. This is a value diverse region, with no specific values that dominate. Riau and Riau Islands are areas mined for oil, gas, and tin (http://www.indonesia.go.id/id/index.php?option=com_content&task=view&id=2957&Itemid=157, accessed 19 September 2012). The coastal area is also used for aquaculture and mariculture, while some islands within this region are designated as tourism sites (e.g. Batam Island, Bintan Island, Tanjung Pinang). For this region, the total annual visitor numbers ranges from 1.5 million to more than 2 million (http://www.indonesia.go.id/id/index.php?option=com_content&task=view&id=3018&Itemid=158, accessed 19 September 2012).

The trade characteristics of each region will influence vessel frequency and vessel type (see Supplementary Tables S1 and S2); with each vessel type believed to pose different risks based on ballast water capacity, operational speed, antifouling coatings and maintenance, total wetted area, and available niche spaces (e.g. Verling *et al.*, 2005; Otani, 2006; Davidson *et al.*, 2009; but also see Hewitt *et al.*, 2011). The implications are that regions that are densely populated, such as Region 1 (Lampung to East Java), might receive more cargo or general container vessel traffic to carry goods; while mining intensive provinces (such as those in Region 3) may have proportionally more tanker and bulk carrier traffic. Tanker and bulk carriers may carry more ballast water and therefore potentially supply more propagules (Otani, 2006; Simkanin *et al.*, 2009) to the spoke ports. Moreover, the export region (the area where mining occurs) experiences more threat than the import region (Otani, 2006) as mining often imports ballast water (vessels arrive empty) and export little ballast water (vessels leave fully laden).

Thus, although the mining provinces that occur along the east coast of Kalimantan in the Makassar Strait (Region 2) received fewer visits than the provinces in Java (Region1), the risk of receiving an introduced marine species via secondary dispersal from the Port of Tanjung Priok could be higher than those in Java or Sumatra. We did not undertake this analysis but suggest such an analysis is needed to further understand ballast water threats in this region.

Therefore, for Indonesia, it is important to recognize and conduct post-border risk assessments for each province that will assess the different provincial values. This information can then be combined with the hub and spoke network model to provide a more meaningful set of priorities to help manage introduced marine species within the Indonesian region.

Conclusions

The hub and spoke network model identified the top 15 Indonesian provinces that may be at high risk of secondary invasions of introduced marine species due to the level of shipping connectivity to the Port of Tanjung Priok in Jakarta Bay. These provinces can be further grouped into three regions based on the values that they represent, which can be used to tailor biosecurity management options. The hub and spoke network model illustrates the vastness of the Indonesian marine biosecurity issue. Indonesia is a large country, with many ports that heavily rely of intra-coastal shipping. Now, managing biosecurity across all provinces within Indonesia is challenging, given the vast extent of this country with its unique geography. Hence, priority should be given to protect the resources and values possessed locally.

Supplementary data

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

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References

Ashton, G., Boos, K., Shucksmith, R., and Cook, E. 2006. Risk assessment of hull fouling as a vector for marine non-natives in Scotland. Aquatic Invasions, 1: 214–218.

Ambarwulan, W. 2010. Remote Sensing of Tropical Coastal Waters: Study of the Berau Estuary, East Kalimantan, Indonesia. Unpublished PhD thesis. University of Twente. Enschende, The Netherlands, 153 pp.

Aykin, T. 1995. Networking policies for Hub-and-Spoke systems with application to the air transportation system. Transportation Science, 29: 201–221.

Azmi, F. 2010. Biosecurity Risk Assessment of Introduced Marine Species in the Port of Jakarta, Indonesia. Master of Applied Science (marine environment) thesis, University of Tasmania, Australia. 130 pp.

Azmi, F., Campbell, M. L., Hewitt, C. L., and Primo, C. in press. Managing introduced marine species in the Coral Triangle: two marine biosecurity risk assessment models for Jakarta Bay, Indonesia. ICES Journal of Marine Research.

Barry, S., Hayes, K. R., Hewitt, C. L., Behrens, H. L., Dragsund, E., and Bakke, S. M. 2008. Ballast water risk assessment: principles, processes, and methods. ICES Journal of Marine Science, 65: 121–131.

Bellwood, D. R., and Hughes, T. P. 2001. Regional-scale assembly rules and biodiversity of coral reefs. Science, 292: 1532–1534.

Bendall, H. B., and Stent, A. F. 2001. A scheduling model for a high speed containership service: A hub and spoke short-sea application. International Journal of Maritime Economics, 3: 262–277.

Bewsell, D., Bigsby, H., and Cullen, R. 2012. Using involvement to understand individual responses to an issue: the case of New Zealand biosecurity. New Zealand Journal of Agricultural Research, 55: 73–88.

Bryan, D. L., and O'Kelly, M. E. 2005. Hub-and-Spoke networks in air transportation: an analytical review. Journal of Regional Science, 39: 275–295.

Campbell, M. L. 2008. Organism impact assessment: risk analysis for post-incursion management. ICES Journal of Marine Science, 65: 795–804.

Campbell, M. L. 2009. An overview of risk assessment in a marine biosecurity context. *In* Biological Invasions in Marine Ecosystems, pp. 353–373. Ed. by G. Rilov, and J. A. Crooks. Springer-Verlag, Berlin, Heidenberg.

Campbell, M. L., Grage, A., Mabin, C., and Hewitt, C. L. 2009. Conflict between International Treaties: failing to mitigate the effects of introduced marine species. Dialogue, 28: 46–56.

Campbell, M. L., and Hewitt, C. L. 2011. Assessing the port to port risk of vessel movements vectoring non-indigenous marine species within and across domestic Australian borders. Biofouling, 27: 631–644.

Campbell, M., Hewitt, C., and Azmi, F. 2013. The utility of hub and spoke network models to analyse pre-border and post-border dispersal of introduced marine species. ICAIS Conference, http://www.icais.org/html/previous18.html (last accessed 2 August 2013).

Carlton, J. T. 1996. Biological invasions and cryptogenic species. Ecology, 77: 1653–1655.

Carlton, J. T. 2001. The scale and ecological consequences of biological invasions in the World's oceans. *In* Invasive Species and Biodiversity Management, pp. 195–212. Ed. by O. T. Sandlund, P. J. Schei, and A. Viken. Kluwer Academic Publishers, The Netherlands.

Carlton, J. T., and Hodder, J. 1995. Biogeography and dispersal of coastal marine organisms: experimental studies on a replica of a 16th century sailing vessel. Marine Biology, 212: 721–730.

Cliff, N., and Campbell, M. L. 2012. Perception as a tool to inform aquatic biosecurity risk assessments. Aquatic Invasions, 7: 387–404.

Cook, D. C., and Fraser, R. W. 2008. Trade and invasive species risk mitigation: reconciling WTO compliance with maximising the gains from trade. Food Policy, 33: 176–184.

Coutts, A. D. M., and Dodgshun, T. J. 2007. The nature and extent of organisms in vessel sea-chests: a protected mechanism for marine bioinvasions. Marine Pollution Bulletin, 54: 875–886.

Coutts, A. D. M., Moore, K. M., and Hewitt, C. L. 2003. Ships' sea-chests: an overlooked transfer mechanism for non-indigenous marine species? Marine Pollution Bulletin, 46: 1504–1515.

Coutts, A. D. M., Piola, R. F., Hewitt, C. L., Connell, S. D., and Gardner, J. P. A. 2010a. Effect of vessel voyage speed on survival of biofouling organisms: implications for translocation of non-indigenous marine species. Biofouling, 26: 1–13.

Coutts, A. D. M., Piola, R. F., Taylor, M. D., Hewitt, C. L., and Gardner, J. P. A. 2010b. The effect of vessel speed on the survivorship of biofouling organisms at different hull locations. Biofouling, 26: 539–553.

Coutts, A. D. M., and Taylor, M. D. 2004. A preliminary investigation of biosecurity risks associated with biofouling on merchant vessels in New Zealand. New Zealand Journal of Marine and Freshwater Research, 38: 215–229.

Davidson, I. C., Brown, C. W., Sytsma, M. D., and Ruiz, G. M. 2009. The role of containerships as transfer mechanisms of marine biofouling species. Biofouling, 25: 645–655.

Davidson, I. C., McCann, L. D., Fofonoff, P. W., Sytsma, M. D., and Ruiz, G. M. 2008. The potential for hull-mediated species transfers by obsolete ships on their final voyages. Diversity and Distributions, 14: 518–529.

Floerl, O., and Inglis, G. J. 2005. Starting the invasion pathway: the interaction between source populations and human transport vectors. Biological Invasions, 7: 589–606.

Forrest, B. M., Gardner, J. P. A., and Taylor, M. D. 2009. Internal borders for managing invasive marine species. Journal of Applied Ecology, 46: 46–54.

Frey, M. A., Simard, N., Robichaud, D. D., Martin, J. L., and Therriault, T. W. 2014. Fouling around: vessel sea-chests as a vector for the introduction and spread of aquatic invasive species. Management of Biological Invasions, 5: 21–30.

Gurning, S. 2008. Indonesian ports 2008, what remain to be tackled? *In* Indonesian Economic Almanac 2008, 4, pp. 105–115. Ed. by Y. Ardi, and R. Amri. Pustaka Indonesia, Jakarta.

Hewitt, C. L., and Campbell, M. L. 2007. Mechanisms for the prevention of marine bioinvasions for better security. Marine Pollution Bulletin, 55: 395–401.

Hewitt, C. L., Campbell, M. L., Coutts, A., Dahlstrom, A., Shields, S., and Valentine, J. 2011. Species Biofouling Risk Assessment. Report produced for the Australian Department of Agriculture, Fisheries and Forestry, Canberra. http://www.daff.gov.au/__data/assets/pdf_file/0008/2059154/species-biofouling-risk-assessment.pdf (last accessed 18 September 2012).

Hewitt, C. L., Campbell, M. L., and Schaffelke, B. 2007. Introductions of seaweeds: accidental transfer pathways and mechanisms. Botanica Marina, 50: 326–337.

Hewitt, C. L., Campbell, M. L., Thresher, R. E., Martin, R. B., Boyd, S., Cohen, B. F., Currie, D. R., *et al.* 2004b. Introduced cryptogenic species in Port Phillip Bay, Victoria, Australia. Marine Biology, 144: 182–202.

Hewitt, C. L., Willing, J., Bauckam, A., Cassidy, A. M., Cox, C. M. S., and Wotton, D. M. 2004a. New Zealand marine biosecurity: delivering outcomes in a fluid environment. New Zealand Journal of Marine and Freshwater Research, 38: 429–438.

Hoeksema, B. W. 2000. The East Indies triangle of marine biodiverisyt. *In* Ninth International Coral Reef Symposium. World Coral Reefs in the New Millennium: Bridging Research and Management for Sustainable Development. Ed. by D. Hopley, P. M. Hopley, J. Tamelander, and T. Done. State Ministry for the Environment; Indonesiain Institute of Sciences, the International Society for Reef Studies, Bali, Indonesia, 45 pp.

Hopkins, G. A., and Forrest, B. M. 2010. Challenges associated with pre-border management of biofouling on oil rigs. Marine Pollution Bulletin, 60: 1924–1929.

Hsu, C-I., and Hsieh, Y-P. 2007. Routing, ship size, and sailing frequency decision-making for a maritime hub-and-spoke container network. Mathematical and Computer Modelling, 45: 899–916.

Imai, A., Shintani, K., and Papadimitriou, S. 2009. Multi-port vs Hub-and-Spoke port calls by containerships. Transportation Research Part E, 45: 740–757.

Inglis, G. J., Hurren, H., Oldman, J., and Haskew, R. 2006. Using habitat suitability index and particles dispersion models for early detection of marine invaders. Ecological Applications, 16: 1377–1390.

Johnson, D. R., Perry, H. M., and Graham, W. M. 2005. Using nowcast model currents to explore transport of non-indigenous jellysfish into the Gulf of Mexico. Marine Ecology Progress Series, 305: 139–146.

Johnson, L. E., Ricciardi, A., and Carlton, J. T. 2001. Overland dispersal of aquatic invasive species: a risk assessment of transient recreational boating. Ecological Applications, 11: 1789–1799.

Kean, J. M., Phillips, C. B., and McNeill, M. R. 2008. Surveillance for early detection: lottery or investment? *In* Surveillance for Biosecurity: Pre-border to Pest Management, pp. 11–17. Ed. by K. J. Froud, A. I. Popay, and S. M. Zydenbos. New Zealand Plant Protection Society, Hastings.

Koh, L. P., and Wilcove, D. S. 2007. Cashing in palm oil for conservation. Nature, 448: 993–994.

Lavoie, D. M., Smith, L. D., and Ruiz, G. M. 1999. The potential for intracoastal transfer of non-indigenous species in the ballast water of ships. Estuarine, Coastal and Shelf Science, 48: 551–564.

Lee, S-W., Song, D-W., and Ducret, C. 2008. A tale of Asia's world ports: that spatial evolution in global hub port cities. Geoforum, 39: 372–385.

Lewis, P. N., Riddle, M. J., and Hewitt, C. L. 2004. Management of exogenous threats to Antarctica and the sub-Antarctic Islands: balancing risks from TBT and non-indigenous marine organisms. Marine Pollution Bulletin, 49: 999–1005.

Locke, A., and Hanson, J. M. 2009. Rapid response to non indigenous species. 1. Goals and history of rapid response in the marine environment. Aquatic Invasions, 4: 237–247.

Minchin, D. 2007. Aquaculture and transport in a changing environment: overlap and links in the spread of alien biota. Marine Pollution Bulletin, 55: 302–313.

Minchin, D., Floerl, O., Savini, D., and Occhipinti-Ambrogi, A. 2006. Small craft and the spread of exotic species. *In* The Ecology of Transportation: Managing Mobility for the Environment, pp. 99–118, Ed. by J. Davenport, and J. L. Davenport. Springer, The Netherland.

Ministry of Transport. 2010. Transportation Statistics. Ministry of Transport, Jakarta, Indonesia.

Mittermeier, R. A., Myers, N., Thomsen, J. B., da Fonseca, G. A. B., and Olivieri, S. 1998. Biodiversity hotspots and major tropical wilderness areas: approaches to setting conservation priorities. Conservation Biology, 12: 516–520.

Muirhead, J. R., and MacIsaac, H. J. 2004. Development of inland lakes as hubs in an invasions network. Journal of Applied Ecology, 42: 80–90.

Otani, M. 2006. Important vectors for marine organisms unintentionally introduced to Japanese waters. *In* Assessment and Control of Biological Invasion Risks. Ed. by F. Koike, M. N. Clout, M. Kawamichi, M. De Poorter, and K. Iwatsuki. Shoukadoh Book Sellers, Kyoto, Japan and IUCN, Gland, Switzerland.

Richards, M., Sainsbury, R., and Kerr, D. 1997. Inequalities in breast cancer care and outcome. British Journal of Cancer, 76: 634–638.

Roberts, C. M., McClean, C. J., Veron, J. E. N., Hawkins, J. P., Allen, G. R., McAllister, D. E., Mittermeier, C. G., *et al.* 2002. Marine biodiversity hotspots and conservation priorities of tropical reefs. Science, 295: 1280–1284.

Ruiz, G. M., Fofonoff, P. W., Steves, B., Foss, S. F., and Shiba, S. N. 2011. Marine invasion history and vector analysis of Califonria: a hotspot for western North America. Diversity and Distributions, 17: 362–373.

Sibthorpe, B. M., Glasgow, N. J., and Wells, R. W. 2005. Questioning the sustainability of primary health care innovation. The Medical Journal of Australia, 183: 52–53.

Simberloff, D. 2000. No reserve is an island: marine reserves and nonindigenous species. Bulletin of Marine Science, 66: 567–580.

Simkanin, C., Davidson, I., Falkner, M., Sytsma, M., and Ruiz, G. 2009. Intra-coastal ballast water flux and the potential for secondary spread of non-native species on the US West Coast. Marine Pollution Bulletin, 58: 366–374.

Skerman, T. 1960. Ship fouling in New Zealand waters: a survey of marine fouling organisms from vessels of the coast and overseas trade. New Zealand Journal of Science, 3: 620–648.

Smith, L. D., Wonham, M. J., McCann, L. D., Ruiz, G. M., Hines, A. H., and Carlton, J. T. 1999. Invasion pressure to a ballast-flooded estuary and an assessment of inoculant survival. Biological Invasions, 1: 67–87.

Tisdell, C. A. 2005. Economics of Environmental Conservation. Edward Elgar Publishing, MA. 288 pp.

Verling, E., Ruiz, G. M., Smith, L. D., Galil, B., Miler, A. W., and Murphy, K. R. 2005. Supply-side invasion ecology: characterizing propagule pressure in coastal ecosystems. Proceedings of the Royal Society B, 272: 1249–1257.

Willan, R. C., Russel, B. C., Murfet, N. B., Moore, K. L., McEnnulty, F. R., Horner, S. K., and Dally, G. M. 2000. Outbreak of Mytilopsis sallei (Recluz, 1849) (Bivalvia: Dressenidae) in Australia. Molluscan Research, 20: 25–30.

Wittenberg, R., and Cock, M. J. W. 2005. Best practices for the prevention and management of invasive alien species. *In* Invasive Alien Species, pp. 209–232. Ed. by H. A. Mooney, R. N. Mack, J. A. McNeely, L. E. Neville, P. J. Schei, and J. K. Waage. Island Press, Washington, DC.

Wotton, D. M., and Hewitt, C. L. 2004. Marine biosecurity post-border management: developing incursion response systems for New Zealand. New Zealand Journal of Marine and Freshwater Research, 38: 553–559.

Wyatt, A. S. J., Hewitt, C. L., Walker, D. I., and Ward, T. J. 2005. Marine introductions in the Shark Bay World Heritage Property, Western Australia: a preliminary assessment. Diversity and Distributions, 11: 33–44.

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Contribution to the Themed Section: 'Risk Assessment'

Original Article

Assessing marine biosecurity risks when data are limited: bioregion pathway and species-based exposure analyses

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We evaluated two risk models (bioregion pathway and species-based exposure), with the aim to determine an effective strategy to implement marine biosecurity risk management in regions/countries where biological data are limited. We used the Port of Tanjung Priok, Jakarta Bay, Indonesia, as a case study to test both models. The bioregion pathway model illustrates that Tanjung Priok is highly connected to the East Asian Sea (~91%), and the Northwest Pacific, Mediterranean, and Australia & New Zealand bioregions ("Very Low" risk), with other bioregions posing "Negligible" risk, highlighting the importance of understanding regional port linkages. The bioregion pathway model strength is grounded by using readily available shipping data; however, it does not classify species into threat categories but considers a larger number of species as an increasing threat. The species exposure model found that 51 species pose a theoretical risk (10 "Moderate", 20 "High", and 21 "Extreme" risks) to Tanjung Priok. These 51 species can be used as a "watch list" for this port. If biosecurity measures for this port were restricted to the outcomes of the bioregion pathway model only 4 of the 51 species highlighted by the species exposure model would have been captured. The species model was data intensive, requiring extensive species datasets and consequently may be unsuitable when data are limited.

Keywords: ballast water, biofouling, biological invasions, developing countries, Indonesia, risk management.

Introduction

Effective management of non-indigenous marine and estuarine species (hereafter NIMES) relies on data about place (pathway epidemiology on local, regional, and international scales), vector (what are the likely transfer mechanisms, exposure, and vector strength), and species (what species are already present in the waters of concern, what species are present along the vector pathways). These data often inform risk assessment processes to enhance capabilities to protect a country's external (Hayes and Sliwa, 2003; Hewitt *et al.*, 2004, 2009a, 2011; Floerl *et al.*, 2005; Campbell, 2011; Ruiz *et al.*, 2011) and internal borders (Wyatt *et al.*, 2005; Campbell, 2008; Herborg *et al.*, 2008; Therriault and Herborg, 2008; Hulme, 2009; Campbell and Hewitt, 2011). However, these types of data are often lacking in an aquatic ecosystem context, especially for developing economies and economies in transition (e.g. Raaymakers and Hilliard, 2002; Endresen *et al.*, 2004; International Maritime Organisation GloBallast Partnership, http://globallast.imo.org/index.asp?page=gef_interw_project.htm&menu=true). Consequently, strategies to implement biosecurity risk assessments that are robust when data are deficient are critically needed (e.g. Barry *et al.*, 2008; Dahlstrom *et al.*, 2011).

To evaluate the biosecurity implications of the absence of biological data, we focused on the Coral Triangle Initiative (CTI) region (encompassing the Philippines, parts of Malaysia, Indonesia, Timor Leste, parts of Papua New Guinea, and the Solomon Islands; http://www.coraltriangleinitiative.org/, accessed 16 December 2014), in particular the port of Tanjung Priok. Tanjung Priok is the largest and busiest port in Indonesia (Nur *et al.*, 2001), sitting with Jakarta Bay and bordered by the Thousand Islands archipelago and the coastal megacity of Jakarta, which suffers from a high level of pollution (Nicholls, 1995;

Nur *et al.*, 2001; Arifin, 2004; van der Meij *et al.*, 2009). There is limited publicly available species data in this region; however, good shipping records are available (from Lloyds Maritime Intelligence Unit). The CTI political agenda is to safeguard the region's marine and coastal (including estuarine) biological resources to enable sustainability. Simultaneously, there is an awareness of NIMES in the region and a willingness to address this issue (e.g. APEC, 2005).

Our approach was to evaluate two risk assessment procedures (a bioregion pathway analysis and a species-based analysis) that differed in primary knowledge requirements to inform coastal managers on the implications of data gaps in relation to biosecurity outcomes (specifically management directions to mitigate NIMES incursions). This could be used to create a viable biosecurity risk strategy that would meet country (Indonesia) and regional [Asia-Pacific Economic Cooperation (APEC), Association of Southeast Asian Nations (ASEAN), and South Pacific Regional Environment Program (SPREP)] needs. To be effective for quarantine purposes, the model would necessarily focus on international borders, with a second aligned model developed for internal borders (Azmi, 2010; Azmi et al., 2015).

The bioregion pathway analysis focused on identifying the exposure of the Port of Tanjung Priok to NIMES recognized in global marine bioregions. This was done by assessing the presence (concentration) of NIMES in each global bioregion then determining the strength of association (pathway strength) that existed between the bioregions and the port. The outcome derived from this analysis was a ranking of bioregions from highest to lowest likely potential source of NIMES to the Port of Tanjung Priok, with the level of threat considered to increase with the total number of NIMES present in the source region. The analysis had a quarantine endpoint to prevent all NIMES from breaching Indonesia's border, assuming a precautionary approach (UNEP, 1992) that considers all NIMES represent an equal threat.

The species-based exposure analysis assessed the distribution of individual NIMES in each of the global bioregions relative to the vessel traffic to Tanjung Priok. The outcomes derived from this analysis are a ranking of individual NIMES that are most likely to be introduced to Tanjung Priok and likely to cause harm. This analysis had an impact-driven endpoint, with each NIMES being assigned a particular level of impact severity based on literature analyses (see Hewitt *et al.*, 2009a, 2011, for additional information). Both risk models focus on vessels (ships) as the vector (transport) mechanism and do not differentiate between biofouling and ballast water.

In this paper, we examine the bioregion pathway analysis and the species-based exposure analysis to determine both risk models effectiveness to best inform biosecurity management practices. The models focus on the Port of Tanjung Priok in Jakarta Bay, Indonesia, as a case study and assess the frequency of contact between the NIMES in each bioregion with the Port of Tanjung Priok.

Methods

Model assumptions

A number of assumptions was made to manage the levels of uncertainty associated with data availability and to meet desired quarantine or impact outcomes.

(i) These risk assessments identify the vessel as the vector and do not differentiate between biofouling and ballast water. As a consequence, factors that potentially affect transfer survival (e.g. vessel speed, transit time, time in source port) are not considered here, as these tend to influence the numbers of individuals, but not the presence of NIMES associated with a vessel (Gollasch, 2002; Minchin and Gollasch, 2002; Hewitt *et al.*, 2009a, b, 2011).

(ii) All species are assumed to survive in the Port of Tanjung Priok. Environmental factors, such as temperature and salinity, are typically used to generate an "environmental matching" in risk assessments. These have been excluded in the bioregion pathway analysis and risk characterization process because they do not portray the likelihood of arrival, but influence establishment (see also discussion in Hewitt and Hayes, 2002; Leppäkoski and Gollasch, 2006; Barry *et al.*, 2008; Hewitt *et al.*, 2009a, 2011).

(iii) As previously stated, if there is a record of an NIMES occurrence in a location within a bioregion, then the species is assumed to occur throughout that bioregion (Hewitt *et al.*, 2009a, 2011).

(iv) Jakarta Bay occurs in the "East Asian Seas" bioregion (Bioregion 13; Figure 1), which is excluded from the bioregion pathway analysis based on assumption 3. Hence, any species in Bioregion 13 is considered to be present in Jakarta Bay. If published data were available at a finer resolution then this assumption could be modified to represent a finer resolution. We have not undertaken a sensitivity analysis to justify the resolution because the data availability is very patchy. Thus, the bioregion pathway analysis only assesses pathways from the other 17 bioregions.

(v) In the bioregion-based pathway analysis:

  (a) all NIMES were considered to pose the same level of threat; no distinction was made between NIMES because the aim is to evaluate which bioregions would be more likely to be donors of NIMES (regardless of potential impact) into the Port of Tanjung Priok;

  (b) all NIMES were considered to have the same likelihood of being transported by vessels and of surviving the journey between donor region and the Port of Tanjung Priok.

(vi) For the species-based exposure analysis:

  (a) no distinction was made between NIMES in terms of their likelihood of transportation;

  (b) the differences in impact and in global distribution between NIMES provide the opportunity to rank species that would be more likely to be transported, introduced, and pose a risk to the Port of Tanjung Priok.

Data

The global identification and distribution of NIMES was drawn from the Hewitt and Campbell (2010) database that lists 1807 marine (and estuarine) species that are known to be introduced outside of their native ranges. Hewitt and Campbell (2010) searched published literature, websites, and grey literature to identify species with records of demonstrable, or assumed invasion history, to create a master list of 1807 species. The global distribution of these NIMES were then determined by a targeted literature search, recording the presence/absence distribution score (i.e. [1] as the species present; and [0] as the species absent) for each IUCN bioregion (Kelleher *et al.*, 1995a,b,c,d; Figure 1) regardless of whether an NIMES was native, cryptogenic, or introduced to that bioregion.

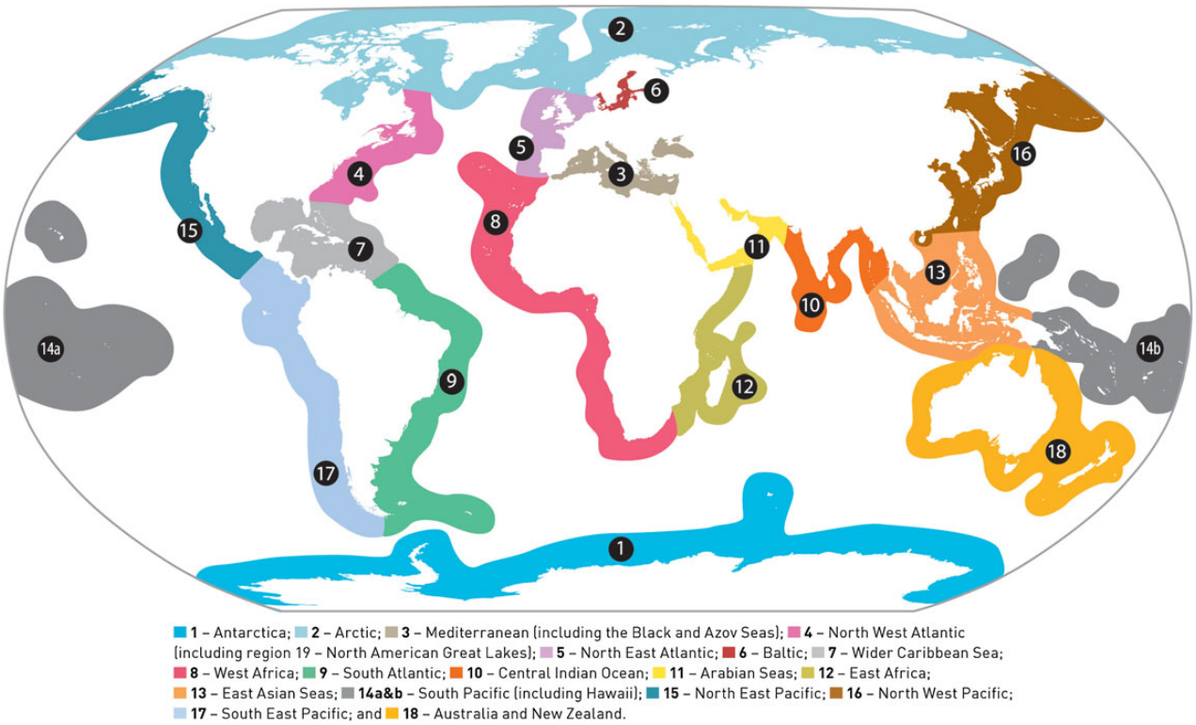

Figure 1. IUCN bioregionalization scheme used in this study (Kelleher et al., 1995a,b,c,d; figure from Hewitt *et al.*, 2011; diagram based on Robinson Projection).

This list was updated (Azmi, 2010) to include additional information on the global distribution of NIMES and recognized or inferred impacts. As with Hewitt *et al.* (2011), NIMES that have an occurrence record in a bioregion were assumed to be present at all locations within that bioregion, regardless of published evidence (see Hewitt *et al.*, 2011, for discussion). To assessing risk to the Port Tanjung Priok, only NIMES not currently known to be present in the East Asian Seas bioregion or NIMES for which impact information could be determined were assessed. This reduced the total number of assessed NIMES to 1074.

An 11-year (1999–2009) shipping dataset for the Port of Tanjung Priok was purchased from the Lloyd's Maritime Intelligence Unit (hereafter LMIU) and used in this analysis. The Last Port of Call (LPOC) data were used to determine the possible source of NIMES. While we recognize that LPOC is unlikely to represent the breadth of risk for vessel-related transport (see discussions in Barry *et al.*, 2008; Campbell, 2009; Hayes and Hewitt, 2001; Hewitt *et al.*, 2011), it provides the most readily available dataset to biosecurity managers and is used here for demonstration. Each last port of call was assigned to a bioregion, providing the bioregion exposure dataset. Records for vessels arriving in Jakarta Bay that could not be unequivocally assigned to a bioregion due to unclear information regarding the LPOC (e.g. "Unknown", "Pacific Ocean") were removed from the dataset.

Bioregion-based pathway analysis

Pathway strength connecting a trading bioregion with the receiving port was calculated as the proportion of the global set of NIMES with an invasion history that was available for transport from that bioregion. The pathway strength was then multiplied by the proportion of arriving vessels, whose last port of call was that bioregion, that enter the receiving port (Port of Tanjung Priok): expressed mathematically by Equation (1). The transport likelihood (pathway strength) is thus presented as a percentage. The quantitative outcomes of Equation (1) were translated into a categorical likelihood measure using the likelihood measures presented in Table 1.

Table 1. Likelihood table for marine biosecurity (from Hewitt *et al.*, 2011).

Descriptor	Description	Proportion of event occurring (%)
Negligible (N)	Event[a] is unlikely to occur	<1
Extremely low (EL)	Event will only occur in exceptional circumstances	1–10
Very low (VL)	Event could occur but not expected	11–25
Low (L)	Event could occur	26–50
Moderate (M)	Event will occur in many circumstances	51–75
High (H)	Event will occur in most circumstances	76–100

[a]Event refers to an incursion of an introduced marine species.

$$L(B_j) = \left(\frac{\sum_{i=1}^{1074} S_{ij}}{N}\right)\left(\frac{V_j}{\sum_{j=1}^{17} V_j}\right) \times 100\%,$$

where $L(B_j)$ is the likelihood that bioregion-j will be the source of introduction of n species from total N marine and estuarine species with an invasion history.

Table 2. Risk matrix (from Hewitt *et al.*, 2011).

	Consequence					
Likelihood	**Negligible**	**Very Low**	**Low**	**Moderate**	**High**	**Extreme**
Negligible	N	VL	VL	L	L	L
Extremely low	VL	L	L	L	M	M
Very Low	VL	L	L	M	M	M
Low	L	L	M	M	H	H
Moderate	L	M	M	H	H	E
High	L	M	M	H	E	E

Used to assess the risk of exposure that each bioregion poses to Tanjung Priok. N, Negligible; VL, Very Low; L, Low; M, Moderate; H, High; E, Extreme.

$$s_{ij} = \begin{cases} 1 & \text{if species} - i \text{ occur in bioregion} - j \\ 0 & \text{if species} - i \text{ does not occur in bioregion} - j \end{cases}$$

where V_j is the total number of vessels who last port of call occurred in bioregion $-j$, where $j = 1, 2, 3, \ldots, 17$ and N is the total number of known marine and estuarine species with an invasion history of 1074.

Given that the endpoint of this model is quarantine focused, any NIMES that breaches the border is considered to have an extreme consequence for all values, regardless of whether the realized consequences would be extreme. This endpoint assumption is common in many quarantine risk assessments used by governments because it purports to protect biodiversity and trade. This method allows the focus to remain on managing the vessel as a vector instead of the individual NIMES that were carried on-board. Risk is then derived using a matrix that assesses the likelihood and consequence measures to produce a relative measure of risk (Table 2). We determined the risk for the three core values that were assessed (environment, economics, and social) and total risk. Total risk is the highest level of risk from the three individual risk values. The highest level of risk was then used to represent total risk when illustrating the models, but a range could be used, or all three values risk rankings depending on the weighting given by the management agencies that use the models.

Species-based exposure analysis

In the species-based exposure analysis, the likelihood that a particular NIMES arrived in Jakarta Bay is the proportion of vessels arriving in the Port of Tanjung Priok in which the NIMES was present in the LPOC's bioregion. The likelihood for the species-based exposure analysis is derived from Equation (2) with the quantitative outcomes represented as a proportion and assessed against Table 1 to derive a categorical measure of likelihood.

$$L(S_i) = \left[\sum_{j=1}^{17} \left(S_{ij} \times \frac{V_j}{\sum_{j=1}^{17} V_j} \right) \right] \times 100\%,$$

where $L(S_i)$ is the Likelihood of species-i that distributed along bioregions $-j$, where $j = 1, 2, 3, \ldots, 17$ arrived in Jakarta Bay, V_j is the total last port of call occur in bioregion $-j$, where $j = 1, 2, 3, \ldots, 17$.

$$s_{ij} = \begin{cases} 1 & \text{if species} - i \text{ occur in bioregion} - j \\ 0 & \text{if species} - i \text{ does not occur in bioregion} - j \end{cases}.$$

Due to the large number of species in the analysis, we restricted the consequence analysis to examine only those NIMES that had a likelihood of arrival $\geq 50\%$ (likelihood categories of "Moderate", "High", and "Extreme") into the Port of Tanjung Priok. This decision is also based on marine biosecurity experience where little effort is expended on species that have a low probability of arrival (personal observations). To assess the consequences of each species arriving in the Port of Tanjung Priok, we used the consequence tables developed by Hewitt *et al.* (2011; Table 3) to evaluate the impact information derived from the literature and assign one of the six value categories: "Negligible", "Very Low", "Low", "Moderate", "High", and "Extreme". The criteria and the level of impacts were adjusted to the scale of the Greater Jakarta Bay Ecosystem (GJBE) (inclusive of Tanjung Priok, Jakarta Bay, and the Thousand Island Archipelago) region. As a result, species that might have "Moderate" impact at a national scale can potentially be assigned a "High" or "Extreme" consequence to the GJBE region because the impact could be severe if occurring locally.

The theoretical thresholds (% values in Table 3) within the consequence tables provide a benchmark of acceptable level of NIMES impact. The thresholds have been created based on data available in the literature and expert opinion (Campbell, 2008, 2009, 2011; Hewitt *et al.*, 2011). Following the Hewitt *et al.* (2011) methods, the consequence criteria (descriptor) used to determine the level of severity was based on the demonstrated, inferred, or unknown (including missing information) impacts for each species within its invaded ranges. NIMES with no available impact data were identified and categorized as "Negligible" in this study; however, we note that this is non-precautionary and further discuss this suite of NIMES in the conclusions. Inferred impacts were categorized as "High" or "Extreme" if they related to: (i) obligations applied nationally or internationally, such as endangered species or habitat; (ii) economic interests of the region; and (iii) human morbidity or mortality.

Risk was determined as for the bioregion-based pathway analysis, using the matrix that assesses the likelihood and consequence measures to produce a relative measure of risk (Table 2). Due to the large number of NIMES assessed, risk is presented for those NIMES that posed a "Moderate", "High", or "Extreme" risk only.

Results

Descriptive patterns

In all, 1074 marine and estuarine species with a known invasion history (those not present in the East Asian Seas bioregion and for which information was complete or accessible) were used in the analyses. During the period 1999–2009, there were 67 826 domestic and international ship calls that entered Jakarta Bay, with the vast majority (90.8%) of these vessels having an LPOC within the East Asian Seas bioregion. Hence, only 6240 ships were analysed for the bioregion pathway assessment.

The patterns of trade within the same East Asian Seas bioregion show that the highest frequency of visits to Tanjung Priok was from domestic (Indonesian) ships that made 42 208 (62%) port calls (Figure 2). Singapore (12 138 visits, ~18%) and then Malaysia (4348 visits, 6%) were the most frequent international port of call visits (Figure 2). The remaining bioregions contributed significantly less to shipping pressure: three bioregions contributed less than 8% [Northwest Pacific Ocean (4.7%), Australia and New Zealand (2.3%), and the Central Indian Ocean (0.8%)] of the overall ship

Table 3. Consequence table used for the species-based exposure analysis for all values [environment, economic, and social; modified from Hewitt *et al.* (2011)].

Descriptor	Impacts
Environmental values	
Negligible to Very Low	Very small environment (<10%) impact from introduced marine species, compared with total impact by other hazards
	Reduction in species richness and composition are not readily detectable (<10% variation)
	In the absence of further introduced marine species impact, recovery is expected within days; no change in species richness or composition
Low	Small environmental (10–20%) impact from introduced marine species, compared with total impact by other hazards
	Reduction in species richness and composition are not readily detectable (10–20% variation)
	If no further introduced marine species impact is experienced, recovery is expected within days to weeks; no loss of species population
Moderate	Medium environmental (20–30%) impact from introduced marine species, compared with total impact by other hazards
	Reductions in species richness and composition are moderate (20–30%)
	Impacts occur at a local scale
	If no further introduced marine species impact is experienced, recovery is expected within months to years; loss of at least one population
High	Limited information is available on the distribution of the environment relative to the introduced marine species distribution; limited information is available on the susceptibility to introduced marine species or the vulnerability of life history stages of these species
	High environmental (30–70%) impact from introduced marine species, compared with total impact by other hazards
	High reductions in species richness and composition (30–70%)
	Impacts occur at a regional scale
	If no further introduced marine species impact is experienced, recovery is expected within years to decades; one local extinction
Extreme	Impacts occurring at a regional scale
	Large environment (>70%) impact from the introduced marine species, compared with total impact by other hazards
	Large reductions in species richness and composition (>70%)
	Impacts occur at a regional scale
	Even if no further introduced marine species impact is experienced, loss of multiple species populations causing significant local extinctions; regional extinction of at least one species
Economic values	
Negligible to Very low	No discernible reduction in regional income (including access to national markets and/or trade) resulting from introduced marine species impacts
	No discernible reduction in local income resulting from introduced marine species impact
	No discernible change to the strength of economic activities
	No damage or deterioration of infrastructure used by a significant proportion of people (>80% of local population) over a local area
	If introduced marine species were removed, recovery is expected within days
Low	Reduction in regional income (including access to national markets and/or trade) resulting from introduced marine species impact <1%
	Reduction in local income resulting from introduced marine species impact is <30%
	Reduction on the strength of economic activities <1%
	10% damage or deterioration of infrastructure used by a significant proportion of people (>80% of local population) across a local area
	If introduced marine species were removed, recovery is expected within days to weeks with no loss of economic industry
Moderate	Reduction in regional income (including access to national markets and/or trade) resulting from introduced marine species impact 1–5%
	Reduction in local income resulting from introduced marine species impact is 30–50%
	Reduction on the strength of economic activities 1–5%
	Economic activity is reduced to less than 95% of its original area (GJBE size)
	10–30% damage or deterioration of infrastructure used by a significant proportion of people (>80% of local population) across a local area
	If introduced marine species were removed, recovery is expected within weeks to months with no loss of economic industry
High	Reduction in regional income (including access to national markets and/or trade) resulting from introduced marine species impact 5–10%
	Reduction in local income resulting from introduced marine species impact is 50–70%
	Reduction on the strength of economic activities 5–10%
	Economic activity is reduced to less than 90% of its original area (GJBE size)
	30–70% damage or deterioration of infrastructure used by a significant proportion of people (>80% of local population) across a local area
	If introduced marine species were removed, recovery is expected within months to years with the loss of at least one economic activity

Continued

Table 3. *Continued*

Descriptor	Impacts
Extreme	Reduction in regional income (including access to national markets and/or trade) resulting from introduced marine species impact >10%
	Reduction in local income from introduced marine species impact is >70%
	Reduction on the strength of economic activities >10%
	Economic activity is reduced to less than 90% of its original area (GJBE size)
	>70% damage or deterioration of infrastructure used by a significant proportion of people (>80% of local population) across a local area
	If introduced marine species were removed, recovery is not expected, with loss of multiple economic activities
Social values	
Negligible to Very low	Social activity reduction is minimal (<1%)
	Degradation of amenities used by 80% of people across a local scale is minimal (<1%)
	No significant changes to regionally important places
	No discernible change in the strength of social activities
	No discernible impact to human health
	If the introduced marine species was removed, recovery is expected within days
Low	Social activity reduction is minimal (1–10%)
	1–10% degradation of amenity used by 80% of people across a local scale
	Small changes (<10%) to regionally important places
	Reduction in the strength of social activities (<10%)
	No discernible impact to human health
	If the introduced marine species was removed, recovery is expected within weeks to months
Moderate	Social activity reduction is 10–30%
	10–30% degradation of amenity used by 80% of people across a local scale
	Medium changes (10–30%) to regionally important places
	Medium reduction in the strength of social activities (10–30%)
	Some impact to human health, medication or treatment required
	If the introduced marine species was removed, recovery is expected within months to years
High	Social activity reduction is 30–40%
	30–70% degradation of amenity used by 80% of people across a regional scale (two ecosystems)
	Moderate changes (30–70%) to regionally important places
	Moderate reduction in the strength of social activities (30–40%)
	Social activities reduced to less than 30–70% of the original area
	Medium impact to human health, hospitalization required
	If the introduced marine species was removed, recovery is expected within years to decades, with the loss of at least one social activity
Extreme	Social activity reduction is >40%
	>70% degradation of amenity used by 80% of the people across a regional scale (two ecosystems)
	Large changes (>70%) to regionally important places
	Large reduction in the strength of social activities (>40%)
	Social activities reduced to more than 70% of the original area
	Extreme impact to human health, including mortality
	Social activities reduced in the nearby region
	If the introduced marine species was removed, recovery is not expected and there would be loss of multiple social activities

traffic arriving in Jakarta Bay, with the remaining 1.4% of shipping pressure spread across 14 bioregions. Small proportions of records (~0.07%, 45 records) had unassigned LPOC and were subsequently excluded from further analyses.

Bioregion-based pathway analysis outcomes

The likelihood of NIMES arriving from each bioregion in the Port of Tanjung Priok varied, with the Northwest Pacific Ocean bioregion having a "Very Low" (17%) likelihood, followed by Australia and New Zealand and the Mediterranean with "Extremely Low" (9.31 and 1.73%, respectively) likelihoods (Table 2). Fifteen bioregions pose a hazard to the Port of Tanjung Priok, but the level of likelihoods that an individual bioregion will act as a pathway of introduction ranges from "Negligible" (<1%) to "Very Low" (10–25%) (Table 4). As previously stated, the consequence for the bioregion pathway analysis is set at extreme to reflect the quarantine endpoint. Thus, the risk derived using Table 3, resulted in 14 bioregions posing a "Low" risk and 3 bioregions (Northwest Pacific Ocean, Australian and New Zealand, and the Mediterranean) posing a "Moderate" risk of introducing marine and estuarine species with a known invasion history (Table 5).

Species-based exposure analysis outcomes

The likelihood of each of the 1074 species entering Tanjung Priok based on exposure was calculated using Equation (2). The likelihood of arrival for more than 450 of the NIMES entering Jakarta Bay was less than 10% ("Extremely Low" or "Negligible" likelihood). Three-hundred and fifty-seven species have a greater than 50% likelihood ("Moderate" and "High" likelihood) of being transported to the Port of Tanjung Priok (Figure 3) given the shipping patterns from the 17 bioregions. Of these, 178 species have a "High" likelihood of being introduced to Jakarta Bay (Figure 3). To demonstrate

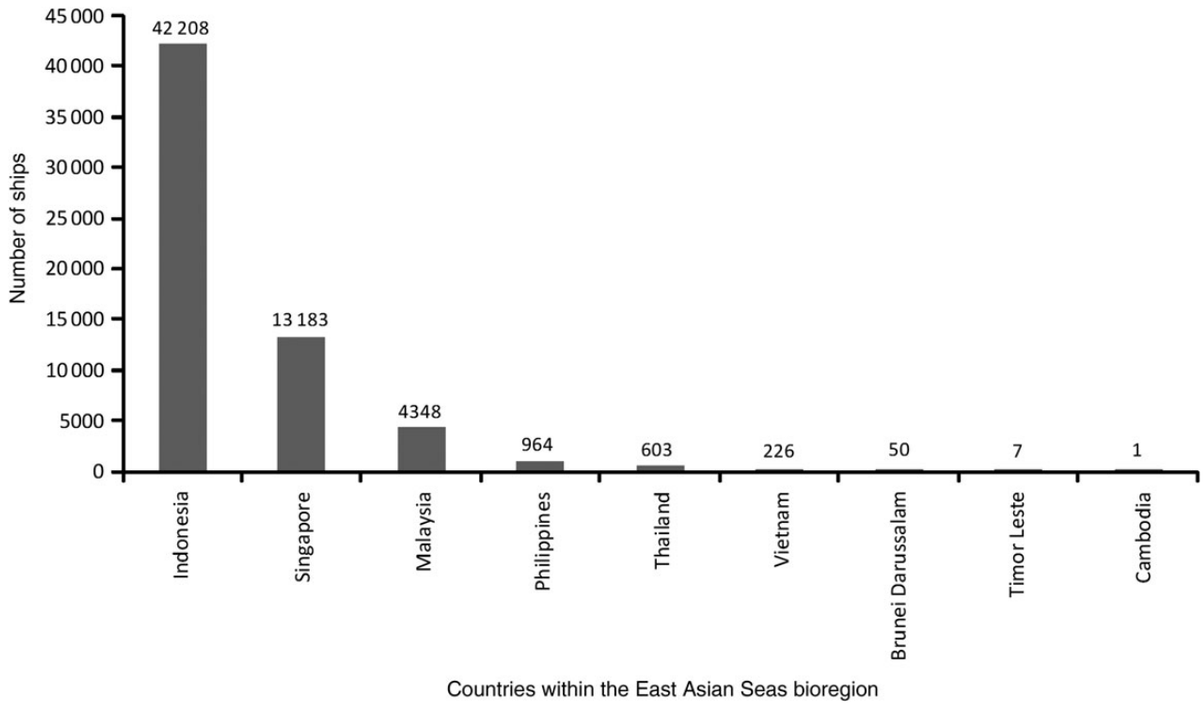

Figure 2. Shipping patterns within the East Asian Seas bioregion from 1999 to 2009 (based on Lloyd's Maritime Intelligence data).

Table 4. The likelihood outcomes for the bioregion pathway analysis of introduced marine species arriving in Tanjung Priok, Indonesia, via biofouling and ballast water.

Bioregion	Likelihood percentage	Likelihood descriptor
Antarctica	0.00	Negligible
Arctic	0.00	Negligible
Mediterranean	1.73	Very Low
Northwest Atlantic	0.01	Negligible
Northeast Atlantic	0.45	Negligible
Baltic Sea	0.01	Negligible
Wider Caribbean	0.13	Negligible
West Africa	0.45	Negligible
South Atlantic Ocean	0.24	Negligible
Central Indian Ocean	0.88	Negligible
Arabian Seas	0.44	Negligible
East Africa	0.10	Negligible
South Pacific Ocean	0.15	Negligible
Northeast Pacific Ocean	0.53	Negligible
Northwest Pacific Ocean	17.1	Very Low
Southeast Pacific Ocean	0.01	Negligible
Australia and New Zealand	9.31	Very Low

The percentage value is derived from Equation (1), with the likelihood descriptor derived from the likelihood table (Table 1).

Table 5. The risk outcomes for the bioregion pathway analysis of introduced marine species arriving in Tanjung Priok, Indonesia, via biofouling and ballast water.

Bioregion	Likelihood	Consequence	Risk
Antarctica	Negligible	Extreme	Low
Arctic	Negligible	Extreme	Low
Mediterranean	Very Low	Extreme	Moderate
Northwest Atlantic	Negligible	Extreme	Low
Northeast Atlantic	Negligible	Extreme	Low
Baltic Sea	Negligible	Extreme	Low
Wider Caribbean	Negligible	Extreme	Low
West Africa	Negligible	Extreme	Low
South Atlantic Ocean	Negligible	Extreme	Low
Central Indian Ocean	Negligible	Extreme	Low
Arabian Seas	Negligible	Extreme	Low
East Africa	Negligible	Extreme	Low
South Pacific Ocean	Negligible	Extreme	Low
Northeast Pacific Ocean	Negligible	Extreme	Low
Northwest Pacific Ocean	Very Low	Extreme	Moderate
Southeast Pacific Ocean	Negligible	Extreme	Low
Australia and New Zealand	Very Low	Extreme	Moderate

The risk descriptor is derived from using the risk matrix (Table 3).

the model, it is these 357 NIMES that were assessed for consequence across the core values of environment, economics, and social values.

Risk was then derived by the authors using the risk matrix (Table 2), with the outcomes for the NIMES that posed a "Moderate", "High", or "Extreme" risk summarized in Table 6. Of the total of 357 NIMES assessed, 51 pose an overall (or total) risk to the Port of Tanjung Priok: 21 NIMES pose an "Extreme" risk, a further 20 NIMES pose a "High" risk, and the final 10 pose a "Moderate" risk (Table 6, total risk). Focusing on individual values, 41% of the identified 51 NIMES have a theoretically "High" risk of impacting upon environmental values based on their previous track record (Figure 4). A further 27% of the NIMES have a "Moderate" theoretical risk of impacting upon environmental values. Whereas 45% of the identified NIMES have a theoretically "Low" risk of impacting economic values, with an additional 35% of NIMES having a "High" risk of impacting on

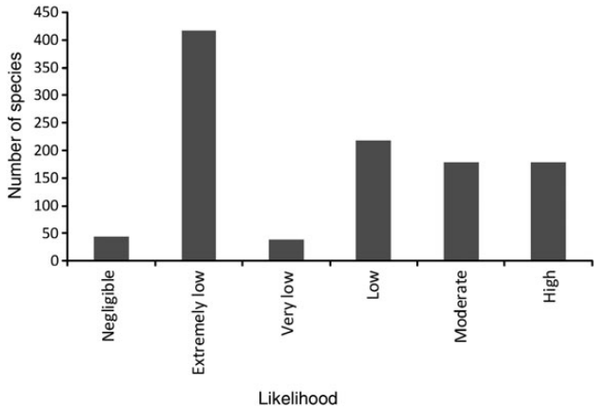

Figure 3. The likelihood of each individual species arriving in Jakarta Bay. Likelihood was derived using Equation (1).

these values (Figure 4). The majority (86%) of NIMES have a "Low" theoretical risk of impacting upon social values (Figure 4). A summary of the impact these 51 NIMES pose to the region, with associated probable vectors, is provided in Supplementary Table S1.

Discussion

Many countries have limited information about the marine and estuarine species within their jurisdictional boundaries. Yet, these countries are all potentially affected by the introduction of species that can impact upon their environment, economics, and social values. To address this, mechanisms need to be developed that can provide guidance to biosecurity management in the regions where there is limited biological information. Fortunately, these regions typically have access to good shipping data. Based on this, we have developed two different biosecurity risk models to assess the threat of marine (including estuarine) invasions. The models can be used to develop foundations for marine biosecurity risk assessment and management in countries with limited data, recognizing the trade-offs to implement precaution.

We used the Port of Tanjung Priok in Indonesia to illustrate these models, but the models can be readily applied to other regions. The models determine the likelihood of an unwanted marine or estuarine species, with a history of invasion, arriving in the Port of Tanjung Priok based on a number of conservative assumptions. The bioregion pathway analysis is typically applied in a pre-border context because it allows management to focus on particular bioregions that might pose an increased risk of species' introduction while acknowledging that the risk exposure of an individual port will vary based on the exposure to its trading partners (Costello *et al.*, 2007).

Outcomes of bioregion pathway analysis models may result in risk management actions such as restricting entry or banning a vessel that arrives from a high risk last port of call without undertaking specific management actions (e.g. undertaking ballast water exchange at sea, provide evidence of vessel cleaning or inspection; e.g. Hewitt *et al.*, 2004, 2009c, 2011; Wyatt *et al.*, 2005; Leppäkoski and Gollasch, 2006; Gollasch *et al.*, 2007), or requesting (either via voluntary or mandatory regulations; e.g. Savarese, 2005) that vessels coming from a bioregion of concern undergo a hull inspection and hull cleaning if biofouling is detected (e.g. Hewitt and Campbell, 2007; Roberts and Tsamenyi, 2008; Hewitt *et al.*, 2011).

The bioregion-based pathway model we present has a quarantine endpoint which assumed that all species posed the same level of extreme threat to the recipient area(s). However, this assumption can lead to incorrect conclusions because not all species will pose an equivalent threat to all core values (environment, economic, social, cultural, human health). In a global context, it is estimated that less than 10% of introduced species are invasive and result in a loss of values (Ricciardi and Rasmussen, 1998). Thus, a bioregion can be a high risk source of new species arrival, but the species that occur in that bioregion may not necessarily all be harmful. As such, the bioregion pathway analysis is simplistic and potentially will result in a greater number of type I errors ("false alarms") due to an over-expectation that all species will cause impact—here, all species are considered "guilty until proven innocent".

In a biodiversity conservation context, this worst case scenario may be appropriate but, as several authors (Possingham *et al.*, 2002; Davis *et al.*, 2011) point out, managing species that do not cause impact (or that we do not know cause an impact) potentially diverts funds that could be applied to species that are having an impact. The opposing position based on the precautionary approach is that a lack of biological data or "scientific truth" is not an excuse for inaction (e.g. Blaikie, 1995; Simberloff, 2003; Dahlstrom *et al.*, 2012). Precaution is most appropriate when the potential for long-term harm (and the inability to mitigate or reconcile the impact if not acted upon early) requires immediate action; practical decisions can often be made based on expert advice when data are deficient (e.g. Therriault and Herborg, 2008; Dahlstrom *et al.*, 2012). Decision-makers need to decide if, in the face of data deficiencies, we are better taking no action or applying a precautionary approach and taking action? Thus, there needs to be a biosecurity trade-off, which is often based on a country's underlying stance towards NIMES (Campbell *et al.*, 2009; Dahlstrom *et al.*, 2011).

The decision to manage NIMES risk based on bioregional risk levels has been applied in many contexts, such as the AQIS Decision Support System used in Australia (Hayes and Hewitt, 2001; Hewitt and Hayes, 2002) and the Risk Assessment Guidelines supporting the IMO Ballast Water Convention (Gollasch *et al.*, 2007; Barry *et al.*, 2008). Biosecurity systems often highlight regions of concern within non-stringent and stringent border controls that include medical (e.g. Dorolle, 1968; Horvath *et al.*, 2006), equipment (e.g. Sanson, 1994; Hewitt *et al.*, 2004; Cliff and Campbell, 2012), and livestock/plant controls (e.g. Williams and West, 2000; Campbell, 2011; Paskin, 2011). These policies are effective in these contexts because they target specific pests, pathogens, or equipment associated with a bioregion(s). Thus, species level information still feeds into the risk management strategy.

In contrast, the species-based exposure analysis for NIMES provides information on individual species that will aid in narrowing a management focus, coupled with bioregional data. For unintentional introductions with ships as vectors, a species exposure analysis does not help in preventing the introductions. Instead, the targeted species list provides a potential watching list of species that can be used as border measures or after an incursion occurs. We identified 51 introduced NIMES of "Moderate" to "High" risk potential to Tanjung Priok. These species have impacts across environmental, economic, and social values and as such should be considered within a biosecurity management strategy for the region.

For the species-based exposure analysis, assigning species with a known invasion history but no apparent investigation of impact in the published or grey literature to "Negligible" consequence has

Table 6. Risk associated with 51 species that are most likely ("Moderate" and "High" likelihood of introduction) to be introduced to Tanjung Priok, Indonesia.

		Consequence			Risk			
Scientific name	**Likelihood**	**Environment**	**Economics**	**Social**	**Environment**	**Economics**	**Social**	**Total risk**
Proteobacterium								
Photobacterium damsela Love, Teebken-Fisher, Hose, Farmer III, Hickman and Fanning, 1981	Moderate	High	High	Negligible	High	High	Low	Extreme
Protozoa								
Orchitophyra stellarum Cepede 1907	Moderate	Moderate	Negligible	Negligible	High	Low	Low	High
Dinophyta								
Alexandrium catenella (Whedon and Kofoid) E. Balech, 1985	High	High	High	Extreme	Extreme	Extreme	Extreme	Extreme
Dinophysis acuminata Claparède and Lachmann, 1859	High	Negligible	Moderate	Moderate	Low	High	High	Extreme
Dinophysis norvegica Claparède and Lachmann, 1859	Moderate	Negligible	High	Moderate	Low	High	High	Extreme
Dinophysis rotundata Claparède and Lachmann, 1859	High	Negligible	Moderate	Moderate	Low	High	High	Extreme
Dinophysis tripos Gourret, 1883	High	Negligible	Moderate	Moderate	Low	High	High	Extreme
Pfiesteria piscicida Steidinger and Burkholder, 1996	Moderate	High	High	Extreme	High	High	Extreme	Extreme
Chlorophyta								
Codium fragile fragile (Suringar) Hariot, 1889	High	Moderate	Moderate	Negligible	High	High	Low	Extreme
Heterokontophyta								
Ectocarpus fasciculatus Harvey, 1841	High	Moderate	Negligible	Negligible	High	Low	Low	High
Rhodophyta								
Corallina officinalis Linnaeus, 1758	High	Low	Negligible	Negligible	Moderate	Low	Low	Moderate
Grateloupia turuturu Yamada, 1941	High	Low	Negligible	Negligible	Moderate	Low	Low	Moderate
Magnolyophyta								
Zostera (Zosterella) japonica Ascherson and Graebner, 1907	Moderate	High	Negligible	Negligible	High	Low	Low	High
Cnidaria								
Tubastraea coccinea Lesson, 1829	Moderate	Very low	Negligible	Negligible	Moderate	Low	Low	Moderate
Annelida								
Alitta succinea (Leuckart, 1847)	High	Moderate	Negligible	Negligible	High	Low	Low	High
Boccardia proboscidea Hartman, 1940	High	Very low	Moderate	Negligible	Moderate	High	Low	High
Dipolydora socialis (Schmarda, 1861)	High	Moderate	Moderate	Negligible	High	High	Low	Extreme
Ficopomatus enigmaticus (Fauvel, 1923)	High	Low	Moderate	Negligible	Moderate	High	Low	High
Hydroides dianthus (Verrill, 1873)	High	Moderate	High	Negligible	High	Extreme	Low	Extreme
Hydroides diramphus Mörch, 1863	High	High	Negligible	Negligible	Extreme	Low	Low	Extreme
Polydora ciliata (Johnston, 1838)	High	Moderate	Moderate	Negligible	High	High	Low	Extreme
Pseudopolydora paucibranchiata (Okuda, 1937)	High	Low	Low	Negligible	Moderate	Moderate	Low	Moderate
Mollusca								
Laguncula pulchella Benson, 1842	Moderate	Extreme	Extreme	Negligible	Extreme	Extreme	Low	Extreme
Limnoperna securis (Lamarck, 1819)	High	High	Negligible	Negligible	Extreme	Low	Low	Extreme
Meretrix petechialis (Lamarck, 1818)	Moderate	Moderate	Negligible	Negligible	High	Low	Low	High
Mytilus edulis Linnaeus, 1758	High	Very low	Negligible	Negligible	Moderate	Low	Low	Moderate
Ocenebra inornata (Recluz, 1851)	Moderate	Moderate	Moderate	Negligible	High	High	Low	Extreme
Patinopecten yessoensis Jay, 1857	Moderate	Negligible	Low	Negligible	Low	Moderate	Low	Moderate
Potamopyrgus antipodarum (J. E. Gray, 1843)	High	Moderate	Low	Negligible	High	Moderate	Low	High
Rhinoclavis (Proclava) kochi (Philippi, 1848)	High	Low	Negligible	Negligible	Moderate	Low	Low	Moderate
Urosalpinx cinerea (Say, 1822)	Moderate	Low	Moderate	Negligible	Moderate	High	Low	High
Arthropoda								
Asellus hilgendorfii Birstein, 1947	Moderate	Low	Low	Negligible	Moderate	Moderate	Low	Moderate
Astacus leptodactylus (Eschscholtz, 1823)	Moderate	Moderate	Low	Negligible	High	Moderate	Low	High
Bythotrephes longimanus Leydig, 1860	Moderate	Moderate	Negligible	Negligible	High	Low	Low	High
Caprella mutica Schurin, 1935	High	Moderate	Negligible	Negligible	High	Low	Low	High
Carcinus maenas (Linnaeus, 1758)	High	Extreme	High	Negligible	Extreme	Extreme	Low	Extreme
Hemigrapsus takanoi Asakura and Watanabe, 2005	Moderate	Moderate	Negligible	Negligible	High	Low	Low	High
Pacifastacus leniusculus (Dana, 1852)	Moderate	High	High	Negligible	High	High	Low	Extreme

Continued

Table 6. *Continued*

Scientific name	Likelihood	Consequence			Risk			Total risk
		Environment	Economics	Social	Environment	Economics	Social	
Palaemon macrodactylus Rathbun, 1902	High	High	Moderate	Negligible	Extreme	High	Low	Extreme
Pseudocalanus newmani Frost, 1989	Moderate	Moderate	Negligible	Negligible	High	Low	Low	High
Ectoprocta								
Tricellaria inopinata d'Hondt and Occhipinti Ambrogi, 1985	High	High	Negligible	Negligible	Extreme	Low	Low	Extreme
Entoprocta								
Barentsia benedeni (Foettinger, 1886)	High	Low	Negligible	Negligible	Moderate	Low	Low	Moderate
Echinodermata								
Asterias amurensis Lutken, 1871	High	High	Moderate	Negligible	Extreme	High	Low	High
Chordata								
Acanthogobius flavimanus (Temminck and Schlegel, 1845)	High	High	Low	Negligible	Extreme	Moderate	Low	Extreme
Gambusia holbrooki Girard, 1859	High	High	Negligible	Negligible	Extreme	Low	Low	Extreme
Mugil soiuy Basilewsky, 1855	Moderate	Moderate	Negligible	Negligible	High	Low	Low	High
Oncorhynchus kisutch (Walbaum, 1792)	Moderate	Moderate	Negligible	Negligible	High	Low	Low	High
Tridentiger bifasciatus Steindachner, 1881	Moderate	Low	Negligible	Negligible	Moderate	Low	Low	Moderate
Platyhelminthes								
Diplogonoporus grandis (Blanchard, 1894)	Moderate	Negligible	Negligible	High	Low	Low	High	High
Heteraxine heterocerca (Goto, 1894) Yamaguti, 1938	Moderate	Low	High	Negligible	Moderate	High	Low	High
Koinostylochus ostreophagus (Hyman, 1955)	Moderate	Low	High	Negligible	Moderate	High	Low	High

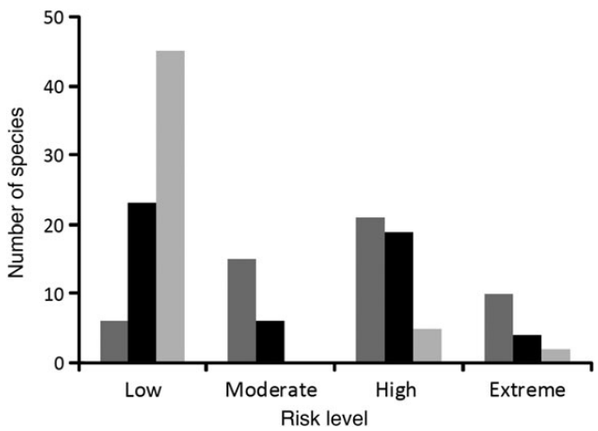

Figure 4. The theoretical impact to values (environmental, dark grey; economic, black; social, light grey) based on the risk outcomes of the species exposure analysis.

significant implications and requires further consideration. This approach is often justified as required treatment under the World Trade Organisation (WTO) Sanitary and Phyto-Sanitary standards (Campbell *et al.*, 2009; Hewitt *et al.*, 2011). However, this "innocent until proven guilty" approach is both non-precautionary and does not provide incentives to develop further understanding. If we were to balance the conflicts between WTO and precaution as described within the Convention on Biological Diversity (CBD; Campbell *et al.*, 2009), then these species should be identified as having intermediate (or neutral) impact equivalent to "Moderate", with a requirement to demonstrate impact (WTO) or demonstrate no impact (CBD). We do not recommend using the terminology "unknown" or "indeterminate" on these occasions, because this has, in the past, been used as a mechanism to reinforce the fallacy of the false negative and state that there is no evidence of impact and therefore no impact (and therefore risk) exists (*sensu* the use of precaution within WTO cases; Campbell *et al.*, 2009; Davidson and Hewitt, 2014).

Strengths and weaknesses of the models

The bioregion pathway and the species-based exposure analyses have different outcomes because they address different issues. The bioregion pathway analysis suggests a lower likelihood of introductions (ranging from "Negligible" to "Very Low"). Indeed, the bioregion pathway model identified three bioregions (NW Pacific, Australia & New Zealand, and Mediterranean) as representing a higher risk than others. The latter analysis suggests that potentially 357 of 1074 (~33%) of the known global NIMES are likely to be introduced to Tanjung Priok (>50%, or a "Moderate" to "High" likelihood). This means that in many or most circumstances, 357 species already introduced elsewhere are likely to be introduced to this region (assuming they have not already been introduced and remain undetected). Fifty-one of these NIMES pose a theoretical risk to Tanjung Priok of sufficient magnitude to raise concern (10 "Moderate", 20 "High", and 21 "Extreme" risks; Table 6). Although all 51 species identified in the species exposure model were represented in the three identified bioregions (NW Pacific, Australia & New Zealand, and Mediterranean), only 4 were restricted solely to those three regions. Thus, if a biosecurity response focused wholly on the three bioregions was in place, the risks would have been inadequately managed. Although the outcomes differ, within a management context both models may prove useful because of the data demands of the models (Table 7).

Bioregion pathway models are useful tools in circumstances where little species data are available or known and where the ability to manage shipping (via regional alliances and collaborations) can be utilized for biosecurity outcomes. This is a very common circumstance and enables proactive decisions to occur based on management of a vector. For example, Australia proactively targets

Table 7. Strengths and weaknesses of the bioregion pathway and species-based exposure risk models.

Model	Strength	Weakness
Bioregion pathway	Works with limited species data	Fails to consider species and therefore is less robust (data poor)
	Focus on the tangible management action of managing vessels (vectors) via port to port connectedness	Lack of differential species impact may result in poor allocation of management resources
	Creates priority actions for managing the driver of the problem (shipping)	Uses a quarantine endpoint—all species that breach the border lead to extreme consequences, which may overestimate risk
	Relatively fast analysis to undertake	
	Relatively low costs	
Species-based exposure	Provides a watch list of species that can be used for surveillance programmes and "black lists" (import prohibition)	Relies on knowledge of species in the port, which can quickly become out-dated
		Data intensive and hence resource intensive
	Rigour—analyses species connections not just vector connections between ports	Requires accurate port surveys and active surveillance to detect watch list species (resource intensive)
	Impact endpoint that enables a magnitude of consequences to be assessed	

vectors to manage both ballast water and biofouling on vessels and equipment (http://www.daff.gov.au/mp, accessed 31 July September 2013). Within Indonesia, species data were difficult to obtain and insufficient for a species level approach. Yet, biosecurity remains an issue that needs to be managed and hence the bioregion pathway model still provides a mechanism to manage biosecurity vector issues. In our study, the bioregion pathway model outcome suggests that management should focus on the East Asian Seas, followed by the Northwest Pacific bioregion. A number of major Southeast Asian port hub cities (e.g. Singapore, Hong Kong, and Tanjung Pelepas) occurs in these two bioregions and our analysis shows that Tanjung Priok has extreme exposure to these ports.

The shipping into Tanjung Priok resembles the intracoastal transport model (Lee *et al.*, 2008), where there is little hinterland movement of goods (i.e. reduced inland penetration via road and rail linkages because of the high concentration of coastal cities, ports, and markets), with a correspondingly high coastal interconnectedness. As such, a hub and spoke network model may be imperative to the management of biosecurity risk in this region (Azmi, 2010; Azmi *et al.*, 2015). To strengthen the bioregion pathway model, regulations to better manage ballast water and biofouling should also be enacted simultaneously with the model.

As with any model, there are weaknesses (Table 7). Specifically, species data are not used and hence the analysis is potentially less robust. The model also uses a quarantine endpoint (which is common in biosecurity management contexts); therefore, all NIMES are treated equally as risk species once they enter the country. This type of endpoint results in potentially overestimation of risk.

In contrast, the species exposure analysis provides a watch list of species that biosecurity management can use to establish NIMES surveillance systems and black lists (import prohibition; Table 7). The foundation of a species exposure model is baseline species data within ports that is updated regularly. This type of data is costly and difficult to obtain (the data are not always in the published domain), with few countries having relevant datasets. To undertake the species analysis we used extensive datasets that were compiled by two of the authors (Hewitt and Campbell, 2010). Typically, this level of data is not available and hence is a major weakness of the species exposure model.

Having species watch lists requires accurate port marine species information (often gathered through surveys; Campbell *et al.*, 2007) and active surveillance to occur both spatially and temporally (Hewitt *et al.*, 2004; Dodgshun *et al.*, 2007). This is rarely evident in NIMES management programs due to resource limitations and because government/political environmental actions tend to be based on reactive (immediate needs focused), not proactive policy-making (e.g. http://www.environment.gov.au/epbc/publications/epbc-reform-better-for-the-environment.html, accessed 31 July September 2013; see also Papadakis and Grant, 2003; Wallington and Lawrence, 2008). Within a marine biosecurity context, national port baseline surveys for NIMES have been instigated in many regions (Campbell *et al.*, 2007); however, baseline evaluations have largely been replaced with surveillance activities for a targeted suite of species. Thus, a species exposure model is likely to be more robust than the bioregion pathway model but the data and resource intensiveness of this model makes it less practical to managers in regions where NIMES data are limited, such as Indonesia.

A further limitation of the species exposure model is that it relies on the knowledge of NIMES in other ports. This information rapidly becomes out-dated as species are moved from region to region and invasions go unnoticed or unmanaged. We reiterate, that port surveys and surveillance activities can be a costly investment (Campbell *et al.*, 2007) that is out of reach of many global ports (hence the establishment of the GloBallast Partnerships Program; http://globallast.imo.org/index.asp?page=gef_interw_project.htm&menu=true); however, the benefits to creating an informed biosecurity framework can rapidly outweigh the costs (Hewitt *et al.*, 2004; Hewitt and Campbell, 2007). Thus, the watch list of species needs to be reviewed and updated regularly, as more information becomes available.

If a conservative approach is to be applied (i.e. managing pathways that posed more risk), and if funds restrict the implementation of multiple management tasks, then the species exposure is a priority management tool because of its species focus. Given the amount of species that are likely to enter the Port of Tanjung Priok, the biosecurity management must proceed to assess the potential species impact to enable management to prioritize mitigation, control, or eradication measures. One of the first steps in this process is to determine what actual species are present within the Port of Tanjung Priok and the surrounding area (Jakarta Bay, including the Thousand Islands) through baseline surveys or intentional search efforts specifically tailored for introduced species (Campbell *et al.*, 2007). This information is useful in that it may reduce the number of watch list species if one (or more) of the 51 species identified here is found to already be present within the area.

Apparent trends and management implications for Indonesia

From the results, the likelihood of NIMES arrival in the Port of Tanjung Priok from a specific bioregion other than the East Asian Seas appears to be "Low" (only 9% of the bioregional exposure is from outside of the East Asian Seas). The fact that the majority (90.8%) of shipping originates from the East Asian Seas and that the major global port hub of Singapore is within the same bioregion suggests that risk management should switch its focus from the ships that come from other bioregions to ships from within the same bioregion, specifically targeting Singapore and other major hub ports in the region.

A further breakdown of the shipping within the East Asian Seas shows that more than half of the total ships that arrived in Tanjung Priok are domestic ships or inter-island shipping within Indonesia, followed by Singapore (19%) and Malaysia (6%) (Figure 2). Port developments in Malaysia are increasing (Slack and Wang, 2002) and hence this area may have a greater influence on shipping patterns in the future. Proximity to the port hub city of Singapore allows exporters from the origin bioregion to send consistent goods in large quantities to Singapore, where the goods are then picked up by feeder ships to Jakarta and other coastal towns, port cities, maritime cities, outports, and urban ports (e.g. Lee *et al.*, 2008). This pattern follows a hub and spoke model of transportation (Lee *et al.*, 2008; Azmi, 2010) and by itself reduces the pressure of other bioregions to Jakarta Bay, specifically the Port of Tanjung Priok.

Singapore is the dominant shipping hub in Southeast Asia (Slack and Wang, 2002; Lee *et al.*, 2008) and was a destination port where many ships were "parked" during the global economic crisis (Floerl and Coutts, 2009). Hence, ships that come from Singapore are likely to present more biosecurity threat to Tanjung Priok via the secondary transfers of NIMES (Azmi, 2010). Due to the proximity of Singapore and Tanjung Priok, a ship transiting between these ports might not have a long enough time for on-board ballast water treatment to be effective or for ballast water exchange to occur. This is further complicated by implications of the phasing out organotin-based antifouling (Champ, 2000, 2003; Hewitt *et al.*, 2009b; Sonak *et al.*, 2009). Thus, the biosecurity risk mitigation measures of ballast water exchange may not be effective coincident with the increasing threat of biofouling. Based on a bioregion pathway model, we would recommend that the port authority in Tanjung Priok prioritize efforts towards vessels that come from the East Asian Seas bioregion as a first step towards an effective biosecurity management strategy.

For biosecurity management to be efficient, knowledge of trading patterns and species within different ports (donor and recipient) is imperative. For Tanjung Priok, the main bioregions (based on the bioregion pathway analysis) of concern are the East Asian Seas, the Northwest Pacific ("Moderate" risk), Australia and New Zealand ("Moderate" risk), and the Mediterranean ("Moderate" risk), with the majority of exposure coming from the East Asian Seas. It is imperative for biosecurity management to know what species occur in the ports within the East Asian Sea bioregion and species status (are the species native or introduced). This will then aid the Jakarta Bay Port of Tanjung Priok Authority to better direct its biosecurity management measures appropriately.

For example, a "new" species arrival in port may not represent a new incursion as the species may already exist in the port. But if this existing information is not known, then resources may be wasted on control or mitigation efforts. Similarly, if species information from trading ports is unknown, that port presents an unmanageable risk as the underlying data used here (known NIMES distributions and vessel movements) are compromised. As the literature attests, a species' track record (i.e. is it introduced elsewhere) contributes to the biosecurity pre-border management of species (e.g. Willan *et al.*, 2000; Hayes and Sliwa, 2003; Hewitt *et al.*, 2004; Hewitt and Campbell, 2007; Whittington and Chong, 2007; Campbell, 2009). Despite this, a species' track record may not be an accurate management tool (e.g. Ricciardi and Cohen, 2007; Molnar *et al.*, 2008; however see Nyberg and Wallentinus, 2005), but at the moment it remains entrenched within a pre-border management paradigm.

The ability to manage the biosecurity risk within the East Asian Seas bioregion will rely on regional/cross border cooperation, which will need to build upon similar perceptions, interests, and management expectations. The biosecurity risks that are current for this region may increase in the future, once the free trade agreement among the ASEAN member countries and Australia and New Zealand begins. In general, conservation and protection of the natural resources is mandated by the Agreement on the Conservation of Nature and Natural Resources 1985 for ASEAN member countries (Shine *et al.*, 2005). This regulation also requires the assessment and measurement of any activities that might affect the environment (http://www.aseanbiodiversity.org/, last accessed 31 July September 2013), which can be the foundation that the ASEAN countries need to proceed to manage the region.

Conclusions

In conclusion, both risk models offer biosecurity directions that are useful but also have weaknesses as their focus is limited or constrained by data quality and availability. Within a marine biosecurity context, knowledge is often limited and the use of best-available information is the paradigm. As such, while acknowledging both model constraints, the models still provide valuable information that can be used by biosecurity managers to start the process of managing this issue. The bioregion pathway model provides insight into the importance of regional port linkages and the need to manage these linkages for better biosecurity outcomes, whereas the species exposure model provides a watch list of NIMES that theoretically pose a risk to Tanjung Priok. Both risk models provide different outcomes yet, instead of being used in isolation, we suggest that the models be used in conjunction to build a biosecurity strategy for Indonesia that combines a species and a bioregion approach.

Supplementary data

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

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References

APEC. 2005. Plan of Action: towards healthy oceans and coasts for the sustainable growth and prosperity of the Asia-Pacific Community.

Joint Ministerial Statement. The 2nd APEC Ocean-Related Ministerial Meeting (AOMM2), Bali, 16–17 September 2005. http://www.apec.org/Meeting-Papers/Ministerial-Statements/Ocean-related/2005_ocean.aspx (last accessed 31 July 2013).

Arifin, Z. 2004. Local Millennium Ecosystem Assessment: Condition and Trend of Great Jakarta Bay Ecosystem. The Ministry of Environment, Jakarta, Republic of Indonesia.

Azmi, F. 2010. Biosecurity risk assessment of introduced marine species in the Port of Jakarta, Indonesia. Unpublished Master of Applied Science (marine environment) Thesis, University of Tasmania. 130 pp.

Azmi, F., Hewitt, C. L., Primo, C., and Campbell, M. L. 2015. A hub and spoke network model to analyse the secondary dispersal of introduced marine species in Indonesia. ICES Journal of Marine Science, 72: 1069–1077.

Barry, S., Hayes, K. R., Hewitt, C. L., Behrens, H. L., Dragsund, E., and Bakke, S. M. 2008. Ballast water risk assessment: principles, processes and methods. ICES Journal of Marine Science, 65: 121–131.

Blaikie, P. 1995. Changing environments or changing views? Geography, 80: 203–214.

Campbell, M. L. 2008. Organism impact assessment: risk analysis for post-incursion management. ICES Journal of Marine Science, 65:795–804.

Campbell, M. L. 2009. An overview of risk assessment in a marine biosecurity context. *In* Biological Invasions in Marine Ecosystems, pp. 353–373. Ed. by G. Rilov, and J. A. Crooks. Springer-Verlag, Berlin, Heidenberg.

Campbell, M. L. 2011. Assessing biosecurity risk associated with the importation of microalgae. Environmental Research, 111: 989–998.

Campbell, M. L., Gould, B., and Hewitt, C. L. 2007. Survey evaluations to assess marine bioinvasions. Marine Pollution Bulletin, 55: 360–378.

Campbell, M. L., Grage, A., Mabin, C., and Hewitt, C. L. 2009. Conflict between International Treaties: failing to mitigate the effects of introduced marine species. Dialogue, 28: 46–56.

Campbell, M. L., and Hewitt, C. L. 2011. Assessing the risk of slow moving barges introducing marine species. Biofouling, 27: 631–644.

Champ, M. A. 2000. A review of organotin regulatory strategies: pending actions, related costs and benefits. Science of the Total Environment, 258: 21–71.

Champ, M. A. 2003. Economic and environmental impacts on ports and harbors from the convention to ban harmful marine anti-fouling systems. Marine Pollution Bulletin, 46: 935–940.

Cliff, N., and Campbell, M. L. 2012. Perception as a tool to inform aquatic biosecurity risk assessments. Aquatic Invasions, 7: 387–404.

Costello, C., Springborn, M., McAusland, C., and Solow, A. 2007. Unintended biological invasions: does risk vary by trading partner? Journal of Environmental Economics and Management, 54: 262–276.

Dahlstrom, A., Campbell, M. L., and Hewitt, C. L. 2012. Mitigating uncertainty using alternative information sources and expert judgement in aquatic non-indigenous species assessment. Aquatic Invasions, 7: 567–575.

Dahlstrom, A., Hewitt, C. L., and Campbell, M. L. 2011. A review of international, regional and national biosecurity risk assessment frameworks. Marine Policy, 35: 208–217.

Davidson, A. D., and Hewitt, C. L. 2014. How often are invasion-induced ecological impacts missed? Biological Invasions, 16: 1165–1173.

Davis, M., Chew, M. K., Hobbs, R. J., Lugo, A. E., Ewel, J. J., Vermeij, G. J., Brown, J. H., *et al.* 2011. Don't judge species on their origins. Nature, 474: 153–154.

Dodgshun, T. J., Taylor, M. D., and Forrest, B. M. 2007. Human mediated pathways of spread for non-indigenous marine species in New Zealand. DOC Research and Development Series 266. Department of Conservation, Wellington, New Zealand. 44 pp.

Dorolle, P. 1968. Old plagues in the jet age. International aspects of present and future control of communicable disease. British Medical Journal, 4: 789–792.

Endresen, O., Behrens, H. L., Brynestad, S., Andersen, A. B., and Skjong, R. 2004. Challenges in global ballast water management. Marine Pollution Bulletin, 48: 615–623.

Floerl, O., and Coutts, A. 2009. Potential ramifications of the global economic crisis on human-mediated dispersal of marine non-indigenous species. Marine Pollution Bulletin, 58: 1595–1598.

Floerl, O., Inglis, G. J., and Hayden, B. J. 2005. A risk-based predictive tool to prevent accidental introductions of nonindigenous marine species. Environmental Management, 35: 765–778.

Gollasch, S. 2002. The importance of ship hull fouling as a vector of species introductions into the North Sea. Biofouling, 18: 105–212.

Gollasch, S., David, M., Dragsund, E., Hewitt, C. L., and Fukuyo, Y. 2007. Critical review of the, I. M.O International Convention on the Management of Ships' Ballast Water and Sediments. Harmful Algae, 6: 585–600.

Hayes, K. R., and Hewitt, C. L. 2001. Ballast-Water Risk Assessment—Volume, I. CRIMP Technical Report 21, CSIRO Division of Marine Research, Hobart.

Hayes, K. R., and Sliwa, C. 2003. Identifying potential marine pests—a deductive approach to applied to Australia. Marine Pollution Bulletin, 46: 91–98.

Herborg, L-M., O'Hara, P., and Therriault, T. W. 2008. Forecasting the potential distribution of the invasive tunicate *Didemnum vexillum*. Journal of Applied Ecology, 46: 64–72.

Hewitt, C. L., and Campbell, M. L. 2007. Mechanisms for the prevention of marine bioinvasions for better biosecurity. Marine Pollution Bulletin, 55: 395–401.

Hewitt, C. L., and Campbell, M. L. 2010. The relative contribution of vectors to the introduction and translocation of invasive marine species. Commissioned by the Department of Agriculture, Fisheries and Forestry (DAFF), Canberra. 56 pp. http://www.marinepests.gov.au/marine_pests/publications (last accessed 7 July 2014).

Hewitt, C. L., Campbell, M. L., Coutts, A., Dahlstrom, A., Shields, S., and Valentine, J. 2009a. Assessment of marine pest risks associated with biofouling. RFQ22/2008. National Centre for Marine and Coastal Conservation Technical Report. Report produced for the Australian Department of Agriculture, Fisheries and Forestry, Canberra.

Hewitt, C. L., Campbell, M. L., Coutts, A., Dahlstrom, A., Shields, S., and Valentine, J. 2011. Species biofouling risk assessment. Report produced for the Australian Department of Agriculture, Fisheries and Forestry, Canberra. http://www.daff.gov.au/__data/assets/pdf_file/0008/2059154/species-biofouling-risk-assessment.pdf (last accessed 18 September 2012).

Hewitt, C. L., Everett, R. A., and Parker, N. 2009c. Examples of current international, regional and national regulatory frameworks for preventing and managing marine bioinvasions. *In* Biological Invasions in Marine Ecosystems, pp. 335–352. Ed. by G. Rilov, and J. A. Crooks. Springer-Verlag, Berlin, Heidenberg.

Hewitt, C. L., Gollasch, S., and Minchin, D. 2009b. The vessels as a vector—biofouling, ballast water and sediments. *In* Biological Invasions in Marine Ecosystems, pp. 117–131. Ed. by G. Rilov, and J. A. Crooks. Springer-Verlag, Berlin, Heidenberg.

Hewitt, C. L., and Hayes, K. 2002. Risk assessment of marine biological invasions. *In* Invasive Aquatic Species of Europe Distribution, Impact and Management, pp. 456–466. Ed. by E. Leppäkoski, S. Gollasch, and S. Olenin. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Hewitt, C. L., Willing, J., Bauckham, A., Cassidy, A. M., Cox, C. M. S., and Wotton, D. M. 2004. New Zealand marine biosecurity: delivering outcomes in a fluid environment. New Zealand Journal of Marine and Freshwater Research, 38: 429–438.

Horvath, J. S., McKinnon, M., and Roberts, L. 2006. The Australian response: pandemic influenza preparedness. Medical Journal of Australia, 185: S35–S38.

Hulme, P. E. 2009. Trade, transport and trouble: managing invasive species pathways in an era of globalization. Journal of Applied Ecology, 46: 10–18.

International Maritime Organisation. GloBallast Partnerships. http://globallast.imo.org/ (last accessed 16 December 2014).

Kelleher, G., Bleakley, C., and Wells, S. 1995a. A Global Representative System of Marine Protected Areas: Antarctic, Arctic, Mediterranean, Northwest Atlantic, Northeast Atlantic and Baltic. World Bank, Washington, DC.

Kelleher, G., Bleakley, C., and Wells, S. 1995b. A Global Representative System of Marine Protected Areas: Wider Caribbean, West Africa and South Atlantic. World Bank, Washington, DC.

Kelleher, G., Bleakley, C., and Wells, S. 1995c. A Global Representative System of Marine Protected Areas: Central Indian Ocean, Arabian Seas, East Africa, and East Asian Seas. World Bank, Washington, DC.

Kelleher, G., Bleakley, C., and Wells, S. 1995d. A Global Representative System of Marine Protected Areas: South Pacific, Northwest Pacific, Southeast Pacific and Australia and New Zealand. World Bank, Washington, DC.

Lee, S-W., Song, D-W., and Ducret, C. 2008. A tale of Asia's world ports: that spatial evolution in global hub port cities. Geoforum, 39: 372–385.

Leppäkoski, E., and Gollasch, S. 2006. Risk Assessment of Ballast Water Mediated Species Introductions—a Baltic Sea Approach. http://www.helcom.fi/stc/files/shipping/Ballast%20Water%20Mediated%20Species%20Introductions.pdf (last accessed 31 July 2013).

Minchin, D., and Gollasch, S. 2002. Vectors—how exotics get around. *In* Invasive Aquatic Species of Europe Distribution, Impact and Management, pp. 183–192. Ed. by E. Leppäkoski, S. Gollasch, and S. Olenin. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Molnar, J. L., Gamboa, R. L., Revenga, C., and Spalding, M. D. 2008. Assessing the global threat if invasive species to marine biodiversity. Frontiers in Ecology and the Environment, 6: 485–492.

Nicholls, R. J. 1995. Coastal megacities and climate change. GeoJournal, 37: 369–379.

Nur, Y., Fazi, S., Wirjoatmodjo, N., and Han, Q. 2001. Towards wise coastal management practice in tropical megacity-Jakarta. Ocean and Coastal Management, 44: 335–353.

Nyberg, C. D., and Wallentinus, I. 2005. Can species traits be used to predict marine macroalgal introductions? Biological Invasions, 7: 265–279.

Papadakis, E., and Grant, R. 2003. The politics of "Light Handed Regulation": new environmental policy instruments in Australia. Environmental Politics, 12: 25–50.

Paskin, R. 2011. Surveillance and prevention in a non-affected state: Victoria. Australian Veterinary Journal, 89: 89–91.

Possingham, H. P., Andelman, S. J., Burgman, M. A., Medellin, R. A., Master, L. L., and Keith, D. A. 2002. Limits to the use of threatened species lists. Trends in Ecology and Evolution, 17: 503–505.

Raaymakers, S., and Hilliard, R. 2002. Harmful aquatic organisms in ships' water—Ballast water risk assessment. CIESM Workshop Monographs, 2: 103–110.

Ricciardi, A., and Cohen, J. 2007. The invasiveness of an introduced species does not predict its impact. Biological Invasions, 9: 309–315.

Ricciardi, A., and Rasmussen, J. B. 1998. Predicting the identity and impact of future biological invaders: a priority for aquatic resource management. Canadian Journal of Fisheries and Aquatic Sciences, 55: 1759–1765.

Roberts, J., and Tsamenyi, M. 2008. International legal options for the control of biofouling on international vessels. Marine Policy, 32: 559–569.

Ruiz, G. M., Fofonoff, P. W., Steves, B., Foss, S. F., and Shiba, S. N. 2011. Marine invasions history and vector analysis of California: a hotspot for western North America. Diversity and Distributions, 17: 362–373.

Sanson, R. L. 1994. The epidemiology of foot-and-mouth disease: implications for New Zealand. New Zealand Veterinary Journal, 42: 41–53.

Savarese, J. 2005. Preventing and managing hull fouling: international, federal and state laws and policies. Proceedings of the 14th Biennial Coastal Zone Conference, New Orleans, Louisiana, 17–21 July 2005. http://ftp.csc.noaa.gov/cz/CZ05_Proceedings/pdf%20files/Savarese.pdf (last accessed 31 July 2013).

Shine, C., Williams, N., and Burhenne-Guilmin, F. 2005. Legal and institutional frameworks for invasive alien species. *In* Invasive Alien Species, pp. 233–284. Ed. by A. Mooney, R. N. Mack, J. A. McNeely, L. E. Neville, P. J. Schei, and J. K. Waage. Island Press, Washington, DC.

Simberloff, D. 2003. How much information on population biology is needed to manage introduced species? Conservation Biology, 17: 83–92.

Slack, B., and Wang, J. J. 2002. The challenge of peripheral ports: an Asian perspective. GeoJournal, 56: 159–166.

Sonak, S., Pangam, P., Giriyan, A., and Hawaldar, K. 2009. Implications of the ban on organotins for protection of global coastal and marine ecology. Journal of Environmental Management, 90: S96–S108.

Therriault, T. W., and Herborg, L-M. 2008. A qualitative biological risk assessment for vase tunicate *Ciona intestinalis* in Canadian waters: using expert knowledge. ICES Journal of Marine Science, 65: 781–787.

UNEP. 1992. Rio Declaration on Environment and Development. http://www.unep.org/Documents.Multilingual/Default.asp?documentid=78&articleid=1163 (last accessed 31 July 2013).

van der Meij, S. E. T., Moolenbeek, R. G., and Hoeksema, B. W. 2009. Decline of the Jakarta Bay molluscan fauna linked to human impact. Marine Pollution Bulletin, 59: 101–107.

Wallington, T. J., and Lawrence, G. 2008. Making democracy matter: responsibility and effective environmental governance in regional Australia. Journal of Rural Studies, 24: 277–290.

Whittington, R. J., and Chong, R. 2007. Global trade in ornamental fish from an Australian perspective: the case for revised import risk analysis and management strategies. Preventive Veterinary Medicine, 81: 92–116.

Willan, R. C., Russell, B. C., Murfet, N. B., Moore, K. L., McEnnulty, F. R., Horner, S. K., Hewitt, C. L., *et al.* 2000. Outbreak of *Mytilopsis sallei* (Recluz, 1849) (Bivalvia: Dreissenidae) in Australia. Molluscan Research, 20: 25–30.

Williams, J. A., and West, C. J. 2000. Environmental weeds in Australia and New Zealand: issues and approaches to management. Austral Ecology, 25: 425–444.

Wyatt, A. S. J., Hewitt, C. L., Walker, D. I., and Ward, T. J. 2005. Marine introductions in the Shark Bay World Heritage Property, Western Australia: a preliminary assessment. Diversity and Distributions, 11: 33–44.

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Contribution to the Themed Section: 'Risk Assessment'

Original Article

A development of ecological risk screening with an application to fisheries off SW England

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A development of the ecological risk screening (ERS) technique, scale intensity and consequence analysis (SICA), is described and application to the varied fisheries and ecosystem off the southwest of England on behalf of an industry steering group (SG) is summarized. The purpose was to prioritize ecological risks systematically and consistently in relation to policy goals agreed by the SG. Scientists listed and advised on ecosystem components, their units (individual species, stocks, habitats, or communities) and attributes, as well as agents of change in the SW, their activities, and generalized effects relevant to the policy goals. A working group (WG) of fishers, fishery observers, technical advisors, and marine scientists paired each unit with the activity thought most likely to impact the most sensitive policy goal, then scored risk according to defined rules spatially, temporally, and as intensity and duration of effects. The geometric mean of the four scores, slightly adjusted for unscored factors if necessary, was the relative impact score (RIS). With this standardized method, the main aspects of risk were considered separately and independently, thereby assisting objective prioritization. Nineteen unit–activity pairs were listed as priority risks (RIS >3) in the SW region during a 2-d meeting that fully exploited the wide range of information and experience available at the WG. Socio-economics was not considered. The ERS for the SW was designed to be compatible with other similar ERSs that might be carried out for neighbouring marine regions. ERS can minimize extra monitoring needed for ecosystem management and, in principle, collaborating non-fishery agents of change could be included. By engaging all stakeholders in the setting of initial priorities for action and by assembling all available sources of information, ERS offers a useful starting point for holistic ecosystem management.

Keywords: Celtic Sea, ecosystem approach to fisheries, ecosystem based fisheries management, ecological risk assessment, ecological risk screening, England (SW), English channel, SICA.

Introduction

Attempts to manage large aquatic systems can quickly become swamped by data describing the states of fisheries and other agents, the many species, physical habitats and communities present, and the ecological processes binding them all together. Although various multivariate methods are available to deal retrospectively with large numbers of indicators (see Table 3 in Cotter *et al.*, 2009), a more purposeful and efficient strategy is to (i) decide policy goals for the aquatic system, (ii) use a comprehensive screening process to weed out the controllable activities of man posing least risk to the achievement of those goals, and then (iii) to monitor only those indicators needed to inform about the state of the system in relation to the remaining, principal risks. In this way, monitoring can be more economical, interpretation of indicators is more direct, and the list of managerial action points can be shorter and more pertinent. Fletcher *et al.* (2005) describe a similar approach.

Methods for screening large numbers of possible ecological risks posed by fisheries have been developed in Australia (Astles, 2008; Scandol *et al.*, 2009). We refer to them collectively as ecological risk screening (ERS) methods within the wider field of ecological

risk assessment (ERA; Burgman, 2005). They include (i) the national ecologically sustainable development method (Fletcher, 2005); (ii) scale intensity and consequence analysis (SICA) which is level 1 of the hierarchical, ERA of the effects of fishing (ERAEF; Hobday *et al.*, 2007); and (iii) qualitative ERA (QERA; Astles *et al.*, 2006). All these methods involve subjective but systematic discussions of lists of potential ecological issues with respect to agreed policy goals at a working group (WG) of interested and informed people. The methods can be ecologically comprehensive, make use of all available sources of information—including publications, theses, and advice from specialists—and can directly engage stakeholders thereby boosting their acceptance of the findings (Fletcher, 2005). The policy goals might originate from government, international conventions, or from a politically relevant local group.

Despite their merits, three concerns with ERS methods may be impeding wider adoption. One is how to choose between the three competing methods that use different concepts of risk and other terms (Astles, 2008; Scandol *et al.*, 2009). Another is that ERS depends too much on the subjective decisions of the people involved. A third is that risk-scoring methods are not yet standardized and may be too imprecise. They include a five-compartment risk matrix (Astles *et al.*, 2006), the product of ranked consequence × ranked likelihood (Fletcher, 2005), and separate spatial and temporal scoring of the worst case for each component that feeds flexibly into an intensity score "judged based on the scale of the activity, its nature, and extent" (Hobday *et al.*, 2007, p. 61).

Our interest in ERS was motivated by fishers and processors based in the SW of England who had been asked to respond to questions from fish retailers about possible overfishing and ecological damage associated with the different fisheries operating from ports in Cornwall, Devon, and Somerset (Figure 1). Details of the fisheries are given elsewhere (Cotter *et al.*, 2006; Walmsley and Pawson, 2007). Five teleost species found in the SW (cod, plaice, Dover sole, whiting, and haddock) received full, annual analytical assessments for management under the European Common Fisheries Policy (CFP) but the results were too focused to answer the general ecological questions being asked. Fishery certification schemes, for example by the Marine Stewardship Council, might have provided fuller answers, but fishers were concerned about the delays and costs of certification. ERS was proposed as a more immediate and cost-effective solution.

This paper presents a development of ERS derived from SICA and implemented on behalf of a steering group (SG) of fishers

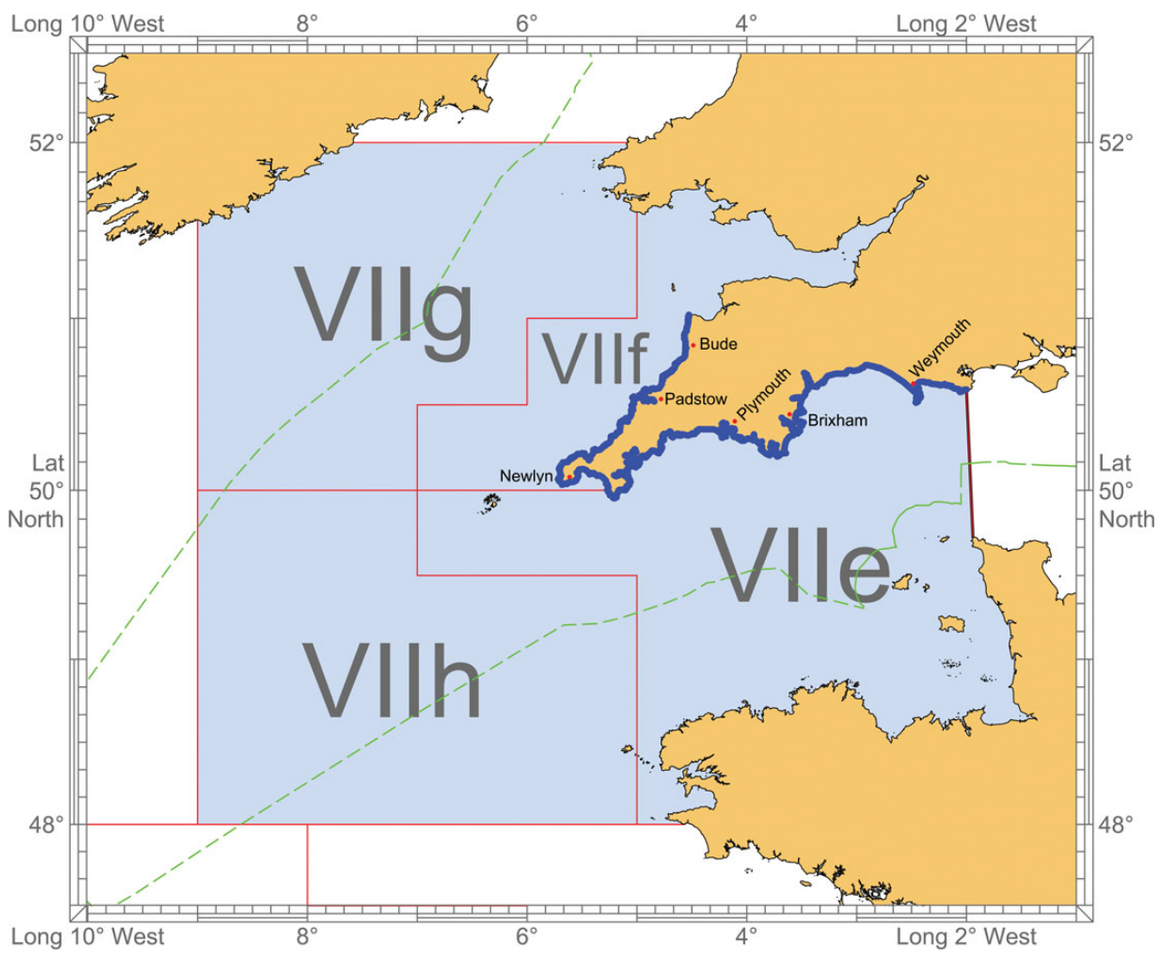

Figure 1. The SW marine ecosystem (ICES VIIe – h) defined for ERS conducted in 2013. The darkened coastline indicates the moorings of included fisheries.

and fish processors operating in the SW. The primary aim was to prioritize systematically and consistently the main ecological risks posed by fishing in the SW and, if possible, by other agents operating there, taking into account any adjusting factors such as existing management measures. The SG and other stakeholders would then be better informed to discuss with fish retailers which risks needed action and which were relatively unimportant. A secondary aim was that risk-scoring should link compatibly across neighbouring marine regions, thus leaving the way open to apply ERS elsewhere around Britain. The ERS scores were not intended to be linked with specific prompts for management actions as has been described in other applications (Fletcher, 2005).

Our ERS WG met for 2 d, in October 2013. Relative risks were decided for many ecological components with the new ERS method though not for all of them because of the limited time and, sometimes, lack of information. The available results, reported fully elsewhere (Seafish, 2014a), are briefly summarized to indicate the scope and output of discussions. The opportunity to extend our work was not available, so this study cannot discuss indicators or monitoring in depth. Our use of ERA terms, emphasized in *italics* at the first occurrence below and summarized in Table 1, mostly follows Hobday *et al.* (2007).

Methods

Initially, the industry SG was invited to discuss and agree (i) the boundaries of the SW marine ecosystem, (ii) the fisheries to be included in the risk assessment, and (iii) the top-level principle and policy *goals* for management of the region. We explained that their choices would govern the whole ERS process by allowing scientists to decide which *effects* of fishing might be contrary to their chosen policies and, later, if and when opportunities permitted, to set detailed *operational objectives* (OOs) and *indicators* for monitoring progress of the ecosystem towards the desired states (Fletcher *et al.*, 2005; Hobday *et al.*, 2007, 2011).

Scientific specialists prepared short background reports on each of the main ecological *components* of the SW system describing (i) its ecology and distribution, (ii) the current states of individual stocks or other subgroupings of populations in relation to recognized reference points or conservation objectives, (iii) known effects of SW fisheries on the component, (iv) measures known to mitigate the effects of fishing, and (v) any other *agent of change* (or just "*agent*") or conservation issues relevant to the component. The reports were circulated to members of the ERS WG.

The Seafish team (WL, AC, JC and M Pawson) prepared other essential documents in advance of the WG. We listed components and *units of analysis* ("*units*"), but differed from current Australian practice (see http://www.afma.gov.au/managing-our-fisheries/environment-and-sustainability/ecological-risk-management/) in not using separate components for target, discarded, and by-product species, or for protected, endangered, and threatened species. In this way, our lists were independent of varying fishery practices and conservation priorities. For species distributed as separate, recognized stocks one of which was local to the SW region, the stock, not the species, was equated with the unit affected by SW fisheries. The effect of this decision was to increase spatial scores, see below. Generalized *attributes* of units, e.g. abundance, were also listed. Background information describing the fisheries selected by the SG was taken from regional reports (Cotter *et al.*, 2006; Walmsley and Pawson, 2007), from ICES fish-stock WG reports, from a European database on fishing effort (Scientific, Technical, and Economic Committee on Fisheries of the European Commission), and from knowledgeable individuals taking part in the ERS WG. Maps of the spatial distribution of fishing grounds around the SW of England based on vessel monitoring data from those fishing vessels >15 m in length were also available (Jennings and Lee, 2012). Agents and their *activities* were listed based on knowledge of the fisheries and

Table 1. ERA terms and abbreviations as used in this paper. Mostly after Hobday *et al.* (2007).

Term	Meaning	Examples
Activity	Something an agent of change does	Fishing, steaming, nutrient input, dredging, making noise underwater
Agent of change or "agent"	Something that can affect an ecosystem	A fishery, agriculture, waste disposal, construction works, climate change
Attribute	A feature of a unit of analysis relevant to its survival and role in the ecosystem	Abundance, length composition (for species), area (for habitat), large fish (for a fish community)
Component	Colloquial grouping of related parts of an ecosystem	Teleosts, elasmobranchs, seabed habitats, ecological communities
Effect or hazard	Change to an attribute of a unit of analysis caused by an activity of an agent of change	Mortality, altered growth, physical disruption, loss of large species
Goal	Top-level policy objective for an ecosystem derived from law, international conventions, or a local political group	"To protect essential ecological processes"
Indicator	A measurable feature of an ecosystem showing its state relative to an operational objective (OO)	Catch per unit effort (cpue) of mature individuals of a species
Member of a unit	One individual of a unit	One organism, one colony, one separate instance of a habitat or community type
Operational objective	State of an indicator that is consistent with a goal	"Cpue of mature individuals is >X kg h^{-1}" consistent with "To maintain reproduction"
Relative impact score (RIS)	Geometric mean of spatial (S), temporal (T), intensity (I), and duration (D) scores	$= \sqrt[4]{S.T.I.D}$
Risk	Probability of a hazardous activity preventing achievement of a policy goal	As indexed relatively by RISs
Unit of analysis or "unit"	One unit of a component	A stock, a species, a habitat type, a community type

other activities occurring in the SW. Effects of activities were classified and named with the aim of creating mutually exclusive categories that were generally applicable, not just to fishing. The relevance of each effect was confirmed by linking it to the policy goals set out by the SG. A spreadsheet, with one sheet per component, was prepared for providing summary information to the ERS WG (Table 2, Prior information).

The ERS WG met at Cefas, Lowestoft from 16/17 October 2013. Members included active fishers, advisors to the fishing industry, specialists on fishery bycatch and fishing gear, fishery scientists, and marine ecologists. A flow diagram of the ERS method used is shown in Figure 2. The most sensitive attribute of each unit was paired with the activity of the agent thought most likely to prevent achievement of the policy goal most likely to be impacted. This is referred to as a unit–activity pair. Other, lesser impacts were ignored, though one unit was sometimes paired with more than one activity to help decide which posed most risk to policy. Cumulative impacts from multiple activities or agents were likewise ignored; this was because of the potential complexities of dealing with them within a simple risk-scoring framework. The WG worked down the prepared lists of units with the help of the background reports, scoring all unit–activity pairs by consensus according to the uniform rules described below. This procedure, though time-consuming, was intended to diminish the influences of preconceived or stereotyped ideas about individual risks, as well as to draw out any special knowledge of WG members.

Our scoring approach differed from that recommended for SICA (Hobday *et al.*, 2007). First, we scored all pairings, not just the "worst case" for each component since the worst cases would have been difficult to agree for the SW without previously applying the systematic scoring system to all cases. Second, we did not always assign a high score when information was lacking, as recommended for SICA for precautionary reasons (Hobday *et al.*, 2007). This would have led to a distracting profusion of high scores. Instead, we identified situations where more information seemed necessary, assigning a low score if that was our best understanding of the situation or, alternatively, postponing scoring of that unit–activity pair indefinitely to leave more time in the meeting to discuss the better-known risks. Third, we used differently defined risk-scoring systems.

Each unit–activity pair was assigned a *relative impact score* (RIS), a new term proposed to emphasize the relative nature of scores more explicitly than variably defined terms with broad usage such as "consequence" and "risk". The RIS was calculated as the geometric mean (fourth root of the product) of scores for spatial scale, temporal scale, intensity-of-effect, and duration-of-effect, each ranging from 0 to 5 and intended to contribute independent, non-overlapping information to the RIS. If any of the four scores was zero, the RIS, being a geometric mean, was also zero. For spatial, temporal, and intensity scores, the guidance given to the WG was 0 = negligible, 1 = <10%, 2 = 10–20%, 3 = 20–50%, 4 = 50–90%, and 5 = 90–100%, where percentage (or corresponding fractional value) refers to the total area, total time, or maximum intensity of an effect, respectively. For duration scores, time frames typically relevant to management were used, see below. Non-integer scores were permitted to resolve disagreements. Spreadsheet columns used to store the four scores, RISs, and other choices made during the WG are presented in Table 2, Findings of the ERS WG.

The spatial score was defined as *the overlap between (or, mathematically, the intersection of) the area of activity, the area occupied by the unit of analysis while the activity is occurring, and the SW region, expressed as a fraction of the total area occupied by the unit*. In Figure 3a, this is usually the grey area divided by the area outlined with dots and dashes though it may sometimes be relevant to notice that, if the unit is migratory, the "total area occupied" may

Table 2. ERS for fisheries off SW England: spreadsheet design used by the ERS WG.

Grouping of columns	Column # and heading	Purpose
Prior information		
Identification and distribution	1. Common name and stock	Identification of unit including the name of SW stock if defined
	2. Scientific name	Technical name of the species, or of the habitat/community
	3. Global distribution	Places occupied by the unit, including outside the SW region
	4. Ecology	Notes reminding of main ecological aspects
Status in SW region in 2013	5. SW stock as the percentage of stated stock	Lo/Mid/Hi estimates of proportion of unit (col. 1) within the SW region
	6. Selected indicators	Selection from available indicators of the status of the unit
	7. Time-trend	Indicators (column 6) for the unit (column 1) trending up/down/level?
	8. Information quality	Good/mid/poor to indicate the reliability of available indicators
	9. Issues	Notes on ecological, data-reporting, regulatory, rarity, or other issues
	10. Information sources	To record consultants' names, references, websites etc.
Findings of the ERS WG		
Selections by the WG	11. Agent of change	The one of most concern from Table 6
	12. Activity	The most risky from Table 6. If undecided, extra rows are used
	13. Attribute	The attribute from Table 8 of the unit most at risk from the activity
	14. Effect	The most damaging effect on the unit from Table 7
	15. OO	OO from Table 8 and indicator level to achieve goals for the unit
	16. Already achieved?	Whether or not the OO was already achieved, if known
Scores	17. Spatial scale	Score 0–5
	18. Temporal scale	Score 0–5
	19. Intensity	Score 0–5
	20. Duration-of-effect	Score 0–5
	21. RIS	$\sqrt[4]{\text{scores } 17 \times 18 \times 19 \times 20}$
Adjustments to score	22. Adjusting factors	Text field listing factors that might alter RIS ± 0.5
	23. Adjusted RIS	Column 21 $\pm$ adjustment from column 22

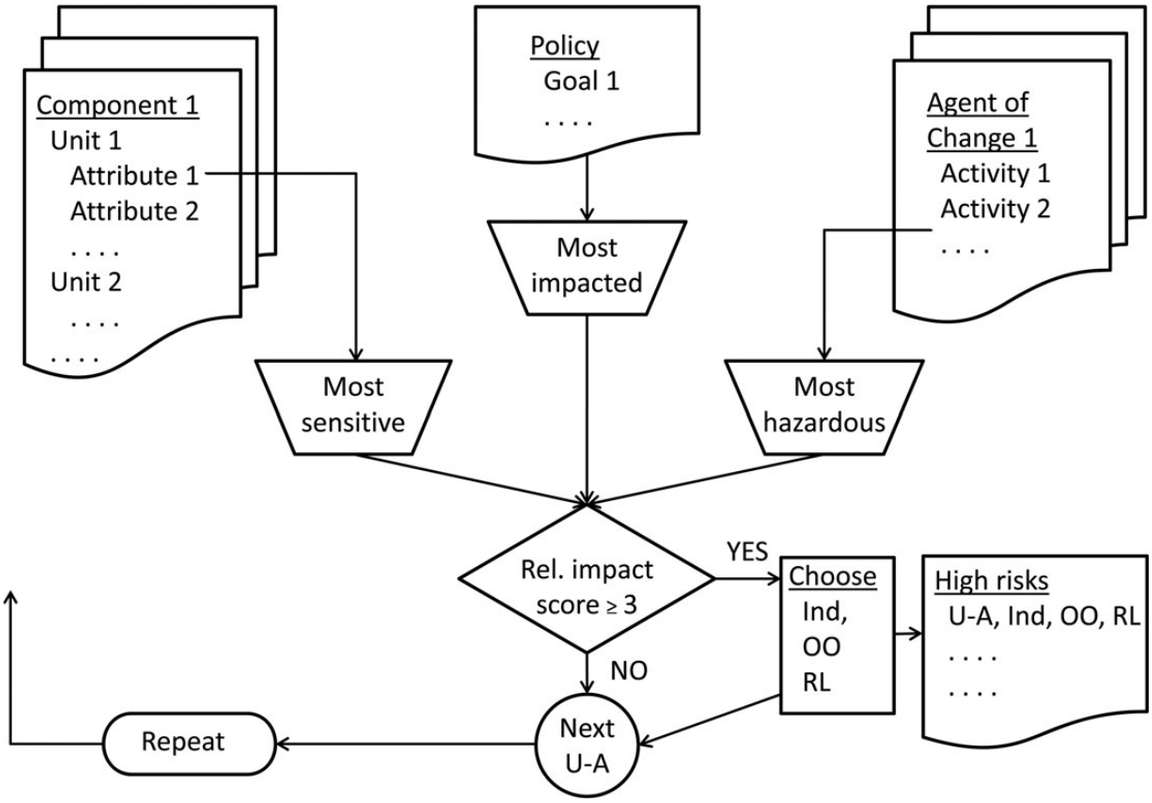

Figure 2. ERS: flow diagram for choosing unit – activity (U-A) pairings with highest RISs. Abbreviations: Ind, indicator; OO, operational objective; RL, reference level; . . ., continuation of list.

be larger than the "area occupied while the activity is occurring". Using the total area occupied as the denominator meant that, if the unit occurred in nearby regions also subjected to ERS, the sum total of spatial scores across all the regions occupied by the unit should never exceed the maximum, 5, and spatial scores were then assigned proportionately among the regions. A "high-mid-low" categorization in spreadsheet column 5, "SW stock as % of stated stock", see Table 2 (prior information), was important for deciding spatial scores. In practice, most spatial scores could only be estimated crudely, partly because fished areas tend to be patchy and depend heavily on variable frequencies of fishing in outlying grounds (Jennings and Lee, 2012), and partly because areas occupied by a unit may also be patchy, poorly known, or depend on population size.

The temporal score was defined for any single year as *the overlap (or intersection) between the period when the unit of analysis occurs in the SW region and the period when the activity occurs there, expressed as a fraction of 1 year (or of the lifespan of the impacted life stage of the unit if < 1 year)*. In Figure 3b, this is the length of the grey arrow as a fraction of the year (or of the vulnerable lifespan if less). The motivation for this definition was that the maximum temporal exposure of a member of a unit to an activity is continuously over its total lifespan though, by subdividing the time risk into years, the lifespan need not be known. Units whose impacted life stages live < 1 year are exceptions in the definition. In contrast to the spatial score, the temporal score could range independently from 0 to 5 in different ERS regions occupied by a unit. This was intended to match the possibility for independent controls on activities in the different regions at any time of year.

By these definitions, our spatial and temporal scores were scaled in relation to the geographic domains and lifespans of the units. The two scores were thus based on measures with biological relevance not possessed by the absolute units (nautical miles, days) employed by SICA (Hobday *et al.*, 2007); the intention was to improve the comparability of scores across different units. Both types of score contributed quantitatively to the calculated RIS, whereas, in SICA, they merely provide background scores from which an intensity score (and thus the final "consequence" score) is derived subjectively. Our view was that this subjective stage was unnecessary. A benefit of our method was that migrations could be allowed for simply: a unit migrating through the SW region annually received a spatial score dependent on the total area occupied by the unit but received a temporal score dependent on the proportion of the year spent in the SW.

The intensity score was defined as *the proportion of the members of the unit of analysis affected by an activity where and when it occurs.* For example, if 25% of a fish species encountering a trawl are caught because the selectivity is 0.25, the assigned intensity score is 3 (between 20 and 50%, see above). The same score would result if 25% of the members of a species present are killed by a spill of a toxicant, or 25% of a habitat is smothered by a single dump of dredge spoil. The words "where and when it occurs" were intended to make intensity scores independent of spatial and temporal scores: they could be high even though the activity rarely occurred in

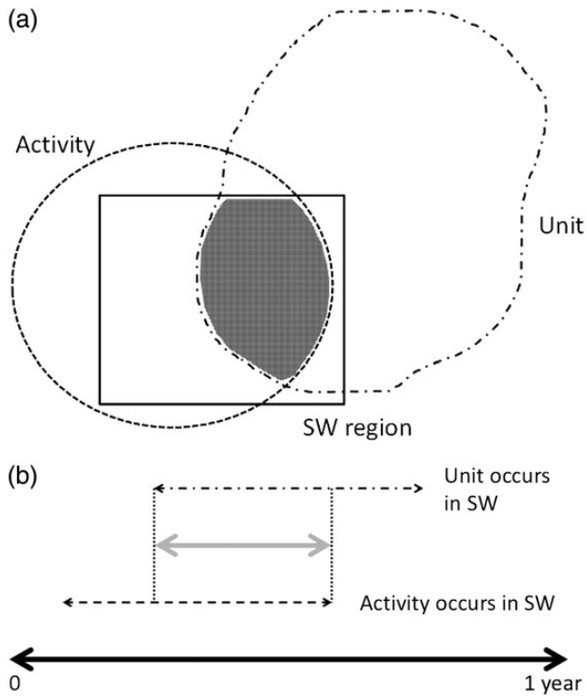

Figure 3. ERS for fisheries off the SW of England; two scoring systems used. (a) Spatial score is the intersection (grey) of the area of activity (dashes), the area occupied by the unit (dot-dashes) while the activity is occurring, and the SW region (rectangle), expressed as a fraction of the total area occupied by the unit (which, if the unit is migratory, may be larger than the dot-dashed region). (b) Temporal score is the length of the grey arrow as a proportion of a year for perennial species. The lifespan of vulnerable stages is used instead of 1 year for annual species.

space or time, and *vice versa*. Our intensity score thus measured a third, independent aspect of impact and was preferred to the subjective intensity score of SICA.

A fourth aspect of ecological impact is the duration of an effect, of obvious relevance for questions of sustainability. We defined a duration score as *the duration of impact on the unit of analysis given that it has been affected and supposing that the activity has stopped*. So, for example, although the effect of mortality is permanent for affected members of a unit, the unit itself may recover. For a species, community, or habitat with epifaunal structure, recovery would be by reproduction and growth of survivors. This idea is similar to "productivity" in productivity–susceptibility analysis (PSA; Stobutzki *et al.*, 2001; Hobday *et al.*, 2011) and "resilience" in QERA (Astles *et al.*, 2006). We preferred the term "duration of impact (or effect)" because it covers non-living cases, for example when the physical structure of a habitat is at risk. The duration score is 0 if immediate recovery of the unit is expected and 5 if the effect is, for practical purposes, permanent. Intermediate duration scorings adopted by the WG were 1 = several months, 2 = approximately 1 year, 3 = 1–3 years, and 4 = 3–10 years. A duration score was not used in the SICA by Hobday *et al.* (2007).

Having calculated a preliminary RIS for a unit–activity pair, the ERS WG considered unscored factors that might reasonably adjust it, for example existing regulations, voluntary practices by fishers, and extreme rarity throughout the range of a species. The RIS was then reduced or increased by up to 0.5 units within the 0–5 scoring scale. Larger adjustments were not permitted, so that the systematic scoring process would not be over-weighted by the subjective adjustment. Unless specified, "RIS" refers to the final outcome of both scoring and adjustment. We followed the arbitrary suggestion of Hobday *et al.* (2007) that consequence scores—in our case, RISs—of 3 or above indicated risks worth investigating further for confirmation and, possibly, consideration by management.

Having found unit–activity pairs with high RISs, the WG briefly considered appropriate OOs, indicators, and reference levels for them within the constraints of existing monitoring programmes, which included market sampling of landings, observer surveys of catches on fishing vessels, and research vessel (RV) surveys. Precise specifications were deferred given that no new monitoring opportunities were foreseen at the time, and that many of the candidate indicators then available from fishery monitoring programmes would serve poorly for ecological monitoring.

Results

The industry SG defined the marine ecosystem (Figure 1) and fisheries to be considered (Table 3), and specified the top-level principle and policy goals to govern the ERS (Table 4). The scale and geographic distribution of the fisheries in Table 3 may have been affected by double counting, particularly of smaller vessels, because of movements between ports and changes of gear seasonally.

The ecological components and units chosen before the ERS WG are listed in Table 5, together with the scientific reviews (Seafish 2014b) and other sources of information used. Proposals, accepted by the WG, for the agents and activities of most relevance, for possible effects categorized in relation to components and goals, and for standardized attributes and OOs are given in Tables 6–8, respectively.

Units with RISs ≥ 3 are listed in Table 9 along with the numbers of unit–activity pairs that were scored for each component, the policy goals (Table 4) thought to be most at risk, other relevant issues, the best currently available indicators and OOs, and the adjusting factors considered. Unless stated, the RISs only relate to fishing activities; risks from non-fishing activities were mostly judged to be lower. Table 9 serves as the list of priority issues with respect to the policy goals in Table 4. For a full presentation of the many detailed regional aspects considered, see Seafish (2014a), and for the completed scoring spreadsheet, see Seafish (2014c). The following notes supplementing Table 9 point out issues thought most important by the WG, together with comments on possible indicators.

Marketable crustaceans

Long-term viability of crustacean fisheries was at risk (goal 1) because of poor knowledge of the biology and ecology of the local stocks, all of which were heavily fished by netters, potters, and trawlers. Total landings and spawners per recruit—as a proxy for maximum sustainable yield (MSY)—were chosen as indicators, given that no more reliable measures of stock security were available from existing monitoring.

Marketable molluscs

Long-term viability of three molluscan fisheries was at risk (goal 1) because of low fecundities and high vulnerability of eggs to bottom trawlers. Heavy catches of scallops, *Pecten maximus*, by dredgers may have impaired their beneficial role in reducing phytoplankton populations and improving water clarity (goal 2) as has been observed for molluscan filter-feeders elsewhere (Newell and Ott,

Table 3. ERS for fisheries off SW England: fisheries selected for inclusion; descriptive data are approximate.

Selected fisheries	Typical vessel lengths (m)	Main target spp. and fishing grounds[a]	Number of ports used[b]	Number of active vessels 2003[c]	2004[c]	2005–2006[b]	Notes
Beam trawlers	25–30	Sole, plaice, megrim, monk; Channel and SW approaches	6	78	70	100	Two2 beams ≤12 m, 80–120 mm mesh, chainmat or open
Otter trawlers	<10–25	Roundfish all around SW peninsula	12	97	102	130	–
Scallopers	<10–30	Scallops, various grounds in Channel	9	40	48	55	Newhaven dredges, sprung teeth
Potters	Many <10	Lobsters, crabs, inshore	48	65	68	350	Also for whelks[a]
Fixed nets	Inshore: <10; offshore 15–25	Various fish, inshore, and SW approaches	46	62	46	370	Gill and tanglenets, various mesh sizes
Lines, angling	Many <10	Conger, ling, mackerel, sea bass	25	15	24	270	–
Ring netters	NA	Pilchard, S coast	NA	NA	NA	NA	Numbers small but unavailable
Pelagic trawlers	NA	Pilchard, scad, sea bass	5	10	11	50	–

Many vessels visited >1 port and fished >1 gear type; many vessels were part time.
[a]Species are: sole, *Solea solea*; plaice, *Pleuronectes platessa*; megrim, *Lepidorhombus whiffiagonis*; monk, mainly *Lophius piscatorius*; roundfish, mainly Gadidae; scallops, mainly *Pecten maximus*; lobster, *Homarus gammarus*; crabs, mainly *Cancer pagurus*; conger, *Conger conger*; mackerel, *Scomber scombrus*; sea bass, *Dicentrarchus labrax*; pilchard, *S. pilchardus*; scad, *Trachurus trachurus*; whelk, *Buccinum undatum*.
[b]All vessel sizes; data from Walmsley and Pawson (2007).
[c]Vessels ≥10 m length overall only; data from Cotter *et al.* (2006).

Table 4. ERS for fisheries off SW England: principle and policy goals agreed by representatives of fishing and processing industries.

Principle	To leave for future generations, the same or better opportunities to benefit from the marine environment around the southwest peninsula as the present generation has enjoyed.
Policy goals	1. To maintain an economically viable and regionally diverse fishing industry in southwest England.
	2. To maintain and protect essential ecological processes and foodwebs.
	3. To avoid taking more fish from a stock than can naturally be replenished.
	4. To protect biodiversity including vulnerable marine species and special types of habitat not specifically covered by legislation.
	5. To minimize pollution as a consequence of fishing so far as practical and economical.
	6. To comply with all legislation applicable to SW fisheries and fish products.

1999). Total landings and, for scallops, catches per unit of effort (cpues) from observer surveys were selected as the best currently available ecological indicators.

Elasmobranchs

Conservation concerns (Ellis *et al.*, 2005; Dulvy and Forrest, 2010) were raised for 14 species of elasmobranch found in the SW region and fished by trawls, nets, and lines (goals 1–4). Several spatial scores were high because of the importance of local stocks. Fisher sightings or observer cpues were thought to be the best indicators available from current monitoring; a few species could be monitored by RV surveys in the SW.

Teleosts

Heavy fishing pressures, lack of scientific knowledge, and discarding put 14 species of teleost at risk (goals 1–4). Spatial scores reflected the importance of local stocks. Some of these had benefitted from management under the CFP but one, the pilchard, *Sardina pilchardus*, was thought to be adversely affected by the low level of management practised in the SW. Fishing mortality (F) and spawning-stock biomass (SSB), along with their reference points recommended by the International Council for the Exploration of the Sea (ICES), were accepted as indicators and OOs for those teleost species that received stock assessments. RV cpues were accepted for several others. Total landings were the only indicator available for four unassessed species not regularly caught by trawl surveys. Several non-commercial species were not considered because of lack of time.

Sea turtles

All five species of sea turtle occurring within the SW region were listed by the International Union for the Conservation of Nature (IUCN), but spatial scores were low because of the smallness of the SW region relative to their global distributions. Intensity scores were low for fishing, because many interactions were thought to occur without a turtle being caught. Duration scores were high because of the low fecundity of sea turtles but only the leatherback, *Dermochelys coriacea*, received an RIS >3 (goal 4) because of its vulnerability to floating polythene litter. The agreed OO was "to avoid increasing the risk to global populations".

Marine mammals

Two cetaceans, *Tursiops truncatus* and *Phocoena phocoena*, received RISs >3 (goal 4) because they were the only known residents in the SW among several species of marine mammal sighted there, and

Table 5. ERS for fisheries off SW England: ecological components and their units of analysis screened by the WG, plus information sources.

Component	Unit of analysis	Number of units	Notes	Websites and references consulted
Commercial crustaceans	Species or local stocks	6	–	[Bell]
Commercial molluscs	Species or local stocks	6	"Squid" (=2 spp.)	marlin.ac.uk; wikipedia.org; iucnredlist.org; [Palmer and Roel]
Elasmobranchs and Lampreys	Species or local stocks	35	Included coastal, migratory, and deep-sea spp.	iucnredlist.org; fishbase.org; wikipedia.org; ices.dk; iccat.int [Ellis *et al.*], [Pawson]
Teleosts	Species or local stocks	37	"Shadd" (=2 spp.), "sea-horses" (=2 spp.), "gobies" (=2+ spp.), "monkfish" (=2 spp.)	iucnredlist.org; fishbase.org; wikipedia.org; ices.dk; Lythgoe and Lythgoe (1991); [Pawson]; [Catchpole]
Turtles	Species or Atlantic subpopulations	5	All spp. are migratory vagrants in SW waters	iucnredlist.org; [Penrose]
Marine mammals	Species or local groupings	18	Several spp. are highly migratory and sporadic in SW waters	Shirihai and Jarrett (2006); [Kingston, Smout, Northridge]; [Treganza]
Seabirds	Species or local breeding groups	24	Many spp. are present only seasonally in SW waters	Peterson *et al.* (1983); Onley and Scofield (2007); [Mander, Thomson, Cutts]
Habitats	Types of habitat	9 Benthic 1 Pelagic	Broad classifications of benthic habitats in SW used	Jennings and Lee (2012); [Bolam]; [Koch and Pacitto]
Communities	Types of community	1 Fish 3 Planktonic 9 Benthic	–	[Bolam]; [Koch and Pacitto]; [Le Quesne]

Author names in square brackets identify unpublished commissioned reviews (Seafish, 2014b).
spp., species.

Table 6. ERS for fisheries off SW England: agents of change and summarized activities, shown ✓.

	Activities							
Agents of change	Steaming	Towing gear on bottom	Other fishing activity	Discarding dead	Littering, pollution, gear loss	Subsea noise, sonar	Other activities	Notes
Beam trawlers	✓	✓	–	✓	✓	✓	–	–
Otter trawlers	✓	✓	–	✓	✓	✓	–	Noise from sounders
Scallopers	✓	✓	–	✓	✓	✓	–	–
Potters	✓	–	✓	–	✓	–	Bait collection	–
Fixed nets	✓	–	✓	✓	✓	–	Ghost fishing	Litter from lost gear
Lines, angling	✓	–	✓	✓	✓	–	Bait collection	Litter from lost lines
Ring netters, seines	✓	–	✓	✓	–	–	–	–
Pelagic trawlers	✓	–	✓	✓	–	✓	–	Noise from sounders
Shipping	✓	–	–	–	✓	✓	Import of invasive species	Noise from engines etc.
Waste discharges	–	–	–	–	✓	–	Pollution	Litter from land
Dredge spoil dumping	✓	–	–	–	✓	✓	Dumping of spoil, rock	Litter from ports
Mineral extraction	✓	✓	–	–	✓	✓	Dredging, drilling	Noisy dredges, drills
Construction works	–	–	–	–	✓	✓	Obstructions	Pile drivers etc.

they were repeatedly exposed to fixed nets and other fishing hazards. Goal 2 may also have been impacted if these species have a significant top-down regulatory effect on their local prey. The most practicable indicator was "Sightings in the SW" using bycatch or other ongoing monitoring programmes.

Seabirds

None of the 24 seabird-activity pairs received RISs >2.6 because of their wide distributions outside the SW and the rareness of significant mortalities of seabirds observed during fishing operations in the region. Some species may have been at risk from a possible

Table 7. ERS for fisheries off SW England: generalized possible effects on different ecosystem components of activities of agents of change, and the policy goals for SW fisheries (numbers in brackets, see Table 4) that might be at risk, shown ✗.

		Policy goals at risk					
Component	**Effect**	**Maintain economic, diverse fisheries (1)**	**Protect ecological processes and foodwebs (2)**	**Avoid overfishing (3)**	**Protect biodiversity (4)**	**Minimize pollution (5)**	**Comply with legislation (6)**
Species or stocks	Direct mortality or injury	✗	✗	✗	✗	–	–
	Indirect mortality or impairment	✗	✗	–	✗	–	–
Habitats	Loss of physical structure or niches	✗	✗	–	✗	–	–
	Increased mobilization of sediments	✗	✗	–	✗	–	–
	Accumulation of dead organic matter	–	✗	–	✗	✗	–
	Reduced clarity of water	–	✗	–	✗	–	–
	Obstruction of living space or migratory routes	✗	✗	–	–	–	–
	Littering with injurious materials	–	–	–	✗	✗	✗
	Contamination by toxic substances	✗	✗	–	✗	✗	✗
	Contamination by underwater noise	✗	✗	–	✗	✗	–
	Contamination of air	–	–	–	–	✗	✗
Communities	Loss of an important ecological function	✗	✗	–	✗	–	–
	Loss of an ecosystem service	–	✗	–	–	–	–
	Increased frequency of blooms or plagues	–	✗	–	✗	✗	–
	Simplification of ecological structure	–	✗	–	✗	✗	–
	Loss of a key supportive species	✗	✗	–	✗	–	–
	Loss of a rare species	–	–	–	✗	–	✗

reduction of small, surface-living fish within foraging range of nesting sites but others, such as gulls and gannets, were known to benefit from discarding. Breeding colonies of seabirds were regularly surveyed in the United Kingdom. The survey database might allow indicators and OOs to be set for monitoring the status of seabirds in the SW region.

Habitats

Although advisory papers (Table 5) were received concerning habitats, the WG decided that there was insufficient time in the meeting to deal with them effectively. Special habitats were being considered by the UK's Marine Management Organization, for example Maerl beds and Ross worm reefs. A general problem was that the extent and distributions of several types of habitat were not well known (Rice *et al.*, 2012, Section 3.1.2).

Communities

Demersal fish communities monitored with RV surveys using length-based indicators were given high spatial, temporal, and intensity scores, because they were treated as restricted to the SW region where fishing takes place throughout the year (goals 1–4). Duration-of-effect was also scored highly since fish communities are slow to respond to reduced fishing (Shephard *et al.*, 2011). Non-disruption of the foodweb was suggested as the reference level for an OO for these indicators. Ichthyoplankton communities received high RISs because of reduced spawning by fished adults but this was merely a secondary aspect of the risks to adult fish communities. Three epibenthic communities were thought to have been affected by trawling and dredging (goals 2 and 4) but four infaunal communities received lower RISs, because these activities, though widespread, exerted a low intensity of effect on buried fauna. Other special and fragile benthic communities found in deeper waters of the SW region, for example pink seafan colonies, were not scored by the WG because of lack of time and information. An OO suggested for such communities was that the key species are successful according to an area- or density-related criterion. Zooplankton communities were considered vulnerable to indiscriminate predation by invasive species such as ctenophores and other "jelly plankton" (Lynam *et al.*, 2006; Bastian *et al.*, 2011), but a high RIS was not thought justified given the open aspect of SW waters to the Atlantic. [See also a later paper on cnidarian jellyfish in the SW (Pikesley *et al.*, 2014).] Phytoplankton communities can be vulnerable to coastal nutrient enrichment, possibly leading to increased frequencies of blooms but they were considered rare in the SW region because of the open, oceanic aspect, so RISs were low.

Table 8. ERS for fisheries off SW England: attributes of units of analysis that may be vulnerable to activities of agents of change, and suggested OOs and applicabilities in brackets.

Unit of analysis	Attributes	Operational objectives (and applicabilities)
A species	Abundance (including reproduction)	$F < F_{msy}$; $SSB > B_{msy}$ (modelled species)
		Survey cpue $> k$ (surveyed species)
		Discarded proportion by number $< k$ (discarded species)
		Landings or other basic data as a proxy for $B_{msy} > k$ (poorly monitored species)
		Secure presence in SW (rare, resident species)
		Sightings in SW $> k$ (rare, migratory species)
		No increase in risk to global population (rare, highly migratory species)
	Growth	Adult cpue $> k$ (measured and surveyed species)
		Proportion of large individuals $> k$ (measured species)
		Average condition factor $> k$ (weighed and measured species)
	Habitat requirements	No further loss of essential habitat (for benthic, demersal spp.)
		Sediment quality parameter $<> k$ (for benthos)
		Water quality parameter $<> k$ (for sensitive species)
A type of habitat	Physical structure	No further alteration of physical structure/topography (seabeds)
		No further obstruction of living spaces (seabeds)
	Water quality	Water quality parameter $<> k$ (habitat subject to pollution)
	Sediment quality	Sediment quality parameter $<> k$ (habitat subject to pollution)
A type of community	Upper size quantile of any species	Proportion of large individuals per species $> k$ (fished communities)
	Proportion of large species	Proportion of potentially large species $> k$ (fished communities)
	Key species	Key species live securely (any community)
	Diversity of species	Species richness $> k$ (seabed communities)
		Species secure in SW or sighted as expected (rare or key species)
	Foodweb structure	Top predators secure, or their cpue $> k$ (fished communities)
		All trophic levels functioning (depleted communities)
		Diverse trophic functional groups (simplified communities)
	Total biomass	Biomass $> k$ (depleted communities)
		Biomass $< k$ (communities susceptible to blooms or plagues)
	Ecosystem service	Service effective, e.g. water clarity $> k$ (filter feeding communities)

F, fishing mortality; B, biomass; msy, maximum sustainable yield; cpue, catch per unit effort (by number or biomass, to be specified); k, a reference value; "surveyed" means subject to quantitative monitoring at sea; $<>$ means $<$ or $>$ as appropriate.

Discussion

The ERS reported here enabled a committee of people with a mix of skills and interests to review the many possible effects of fishing on the ecology of the SW region with "a disciplined and consistent approach" (Fletcher, 2005). Substantial detail was available from members of the WG on many species and their interactions with fisheries, on fishery regulations and bylaws, and on fishing tactics, gears, and markets. Similar benefits of ERA were reported by Fletcher (2005) for Western Australian fisheries. As a general conclusion, the ERS usefully supplemented scientific advice provided by ICES for commercial species managed individually under the European CFP. Since ERS finds priorities from among the, possibly, hundreds of concerns that might be raised about an aquatic ecosystem and, since it can productively involve stakeholders and tap all available sources of information, some form of ERS is likely to be a useful starting point for an ecosystem approach to management. Monitoring, research and, perhaps, short-term management actions then have an initial justification even if, later, calls are made to justify or adjust the priorities by more objective methods.

Our ERS method was intended to be objective and repeatable should a similar ERS ever be undertaken by a different WG, either to review our findings or as part of a repeating cycle to maintain and improve ecological awareness. Precisely defining the scoring methods set "rules for the game" and is recommended because all unit–activity pairs can then be treated uniformly, scoring disagreements can sometimes be resolved by reference back to the definitions, and any political influences at the WG can be held in check. Independence of the four scores we used prompted the WG to deal with the main aspects of ecological risk (Marasco *et al.*, 2007; Rice *et al.*, 2012) separately and without counting any of them more than once, thereby further helping to improve objectiveness. Spatial and temporal measurement scales were standardized in an ecological sense by measuring them in relation to total geographic distributions and lifespans, respectively, rather than in terms of absolute units that may have different relevance for different units of analysis, possibly leading to incorrectly ordered spatial and temporal scores. Spatial scoring scaled risks in relation to area so as to assign conservation responsibilities fairly among different fishery regions. This is important in the United Kingdom where spatial management zones tend to be small relative to the distributions of many marine species. The option to arbitrarily adjust RISs by ± 0.5 satisfied the WG's wishes to alter slightly some RISs thought inappropriate because of unscored factors but, for the sake of objectivity, did not allow the main systematic scoring procedure to be rendered redundant.

Based on the adjusted RISs and the arbitrary cut-off of 3, a prioritized list of sustainability and conservation issues was prepared (Table 9). The effects of varying the cut-off on the issues brought forward could be explored, if required, by referring back to the WG spreadsheet. However, the cut-off should not be set too low if the RISs tend to be clustered at lower values, because their ordering is then not dependable. The subjective basis of ERS, however rigorously it is carried out, implies that mandatory linkages between RISs and managerial actions should be avoided.

Table 9. ERS for fisheries off SW England: summarized list of principal risks of commercial fishing to policy goals (see Table 4) as decided by the WG.

Ecological component (U–A pairs scored)	Units with RISs ≥3	Main policy goals at risk (Table 4)	Relevant issues	Best available indicators and OOs	Adjusting factors	
					Existing management	Other factors
Marketable crustaceans (14)	*Palinurus elephas* *Homarus gammarus* *Cancer pagurus* *Maja brachydactyla*	1 (fisheries)	Poorly known growth, mortality rates and ecology	Landings > *k* Spawners per recruit > *k*	✓Closed seasons; MLSs; licensing; soft and berried not landed	✓High discard survival from pots; no market for small individuals; ✗damage in nets
Marketable molluscs (11)	*Buccinum undatum* *Sepia officinalis* *P. maximus*	1 (fisheries) 2 (processes)	Low fecundity, vulnerable eggs Scallops: water clarification	Landings > *k* Observer cpue > *k*	✓Discards reduced Scallops: rotation of fishing beds, effort controls	–
Elasmobranchs (33)	Twenty species of ray, dogfish, and shark	1 (fisheries) 2 (processes) 3 (stocks) 4 (diversity)	IUCN listings, predators, low fecundity	Observer cpue > *k* Sightings in SW Rays, dogfish: RV survey cpue > *k*	✓Protection of some spp.; fishery regulations on MLSs, landings	✓May survive discarding; ✗ high vulnerabilities to fishing
Teleosts (24)	Fourteen commercial species	1 (fisheries) 2 (processes) 3 (stocks) 4 (diversity)	Discarding, poorly known biology, vulnerable nursery areas and aggregations, foodweb roles	$F < F_{msy}$, $SSB > B_{msy}$ Landings > *k* 4 spp. only: RV survey cpue > *k*	✓CFP TACs, effort controls and technical measures giving some improvements	✓Spatial separation of nursery and fishing grounds or by age and sex; ✗non-UK catches, Pilchard: no management
Sea turtles (5)	*D. coriacea*	4 (diversity)	IUCN listings; low fecundities, highly migratory; floating litter	Sightings in SW reduce risks of discarding and littering	✗Conservation listings lack legal backing	✓Oceanic ranges imply a little impact of fisheries in SW England
Marine mammals (10)	*T. truncatus* *P. phocoena*	2 (processes) 4 (diversity)	Local and migratory spp.; entanglement in fixed nets; top predator roles	Sightings in SW; 1.7% annual removal rate (ASCOBANS)	✓ASCOBANS; EU reg. 812/2004 on pingers to reduce bycatches	✗Pinger trials inconclusive for *Tursiops*
Habitats (0)	No habitats were considered	2 (processes) 4 (diversity)	Distributions poorly known; Priority listings of special habitats in SW region	—	✓UK Biodiversity Action Plan	✗Impacted by aggregate extraction, dredge spoil dumping; construction works
Communities (16)	Demersal fish; ichthyoplankton Three epibenthic communities	1 (fisheries) 2 (ecology) 3 (stocks) 4 (diversity)	Already highly modified by fisheries; ichthyoplankton and fish linkages	Size-based indicators → trophic functioning; CPR-based indicators	✓CFP controls benefiting some fish spp.	✗Impacted by non-UK fishers and non-fishing activities

U–A, unit–activity; RIS, relative impact score; MLS, minimum landing size; RV, research vessel; CFP, Common Fisheries Policy of the European Union; ASCOBANS, Agreement for conservation of small cetaceans in the Baltic and North Seas; CPR, continuous plankton recorder operated by the Sir Alister Hardy Foundation for Ocean Science. ✓, reducing risk; ✗, increasing risk.

Future actions on priority issues identified by ERS were not discussed at the WG, but might involve higher level assessments such as PSA and special modelling to confirm the risks found (Hobday *et al.*, 2007, 2011). A danger, though, with this hierarchical approach is that the different levels utilize many of the same data and information and therefore are not independent (Hobday *et al.*, 2011, p. 380), implying that poorly determined RISs could be erroneously confirmed automatically by the more specialized studies. A better strategy is to seek new sources of information for new studies to confirm or explore high risks. A model-based approach to regional ERA at a higher level than ERS is presented by Fock (2011).

When ERS is accepted to have been well informed and implemented, corrective actions might be agreeable for priority issues without further investigations. They might include voluntary changes or financial incentives to improve fishing practices, publicity to increase awareness of important problems, new local regulations or bylaws, organization of fishers and observers to identify correctly and report sightings of rare species, as well as adjusted or specially designed monitoring if suitable indicators and OOs are available for units at risk. The ERS WG recognized that "SMART" (Specific, Measurable, Attainable, Relevant, and Time-bound) OOs are essential to effective monitoring of the status of units of analysis deemed to be at high risk (Fletcher *et al.*, 2005). However, difficulties were experienced in identifying promising candidate indicators from the monitoring programmes then existing in the SW, mainly to control landings of commercial species under the CFP. In this respect, the ERS helpfully provided a short list of units requiring indicators and monitoring if and when a more ecosystem-orientated approach is adopted for the SW.

Drawing up clearly stated policy goals (Table 4) before the ERS WG allowed it to decide almost immediately whether or not the effect of an activity was acceptable with respect to that policy. This feature, taken from the Australian ERS methods, almost certainly helped the WG to avoid sterile political arguments about conservation-*vs.*-commerce when discussing species or habitats of conservation importance. The policy goals for the SW had no legal status but, as they represented the views of the fishing industry, carried considerable political weight, particularly as they looked well beyond immediate commercial considerations and covered many peoples' aspirations for the future of the SW marine region. In contrast, a significant criticism of fisheries law under the European CFP was that policy was too imprecise for the effective guidance of management (EC, 2009).

Given additional funding for appropriate specialists, agents other than fisheries could be included compatibly in an ERS, for example gravel miners, offshore energy producers, and waste dischargers. This might enhance overall ecosystem management, though the activities at sea of many non-fishing agents are already regulated under United Kingdom and international legislation (Rees *et al.*, 2006). ERS takes no account of the socio-economic aspects of exploiting aquatic systems, a basic feature of the ecosystem approach to fisheries management (FAO, 2003, 2005). Since an ERS WG already has a long agenda, socio-economic aspects would probably need a separate WG, allowing different professional advisors to be present. The two sets of advice could then be weighed against each other and translated into actions using a political or a reporting process. An example of the latter is described by Fletcher *et al.* (2005).

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References

Astles, K. L. 2008. A systematic approach to estimating ecological risks in marine fisheries. CABI Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources 3. 16 pp.

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.

Bastian, T., Haberlin, D., Purcell, J. E., Hays, G. C., Davenport, J., McAllen, R., and Doyle, T. K. 2011. Large-scale sampling reveals the spatio-temporal distributions of the jellyfish *Aurelia aurita* and *Cyanea capillata* in the Irish Sea. Marine Biology, 158: 2639–2652.

Burgman, M. 2005. Risks and Decisions for Conservation and Environmental Management (Ecology, Biodiversity and Conservation). Cambridge University Press, Cambridge, UK. 488 pp.

Cotter, A. J. R., Course, G. P., Ashworth, J., Forster, R., Enever, R., Goad, D., Bush, R., *et al.* 2006. Summary of commercial fishing by English and Welsh vessels > 10 metres LOA for 2004. Scientific Series Technical Report No 134, Cefas, Lowestoft.

Cotter, J., Petitgas, P., Abella, A., Apostolaki, P., Mesnil, B., Politou, C. -Y., Rivoirard, J., *et al.* 2009. Towards an Ecosystem Approach to Fishery Management (EAFM) when trawl surveys provide the main source of information. Aquatic Living Resources, 22: 243–253.

Dulvy, N. K., and Forrest, R. E. 2010. Life histories, population dynamics and extinction risks in chondrichthyans. *In* Biology of Sharks and Their Relatives, pp. 635–675. Ed. by J. C. Carrier, J. A. Musick, and M. R. Heithaus. CRC Press, Washington, DC.

EC. 2009. Green Paper: Reform of the Common Fisheries Policy. COM(2009)163. Commission of the European Communities, Brussels.

Ellis, J. R., Dulvy, N. K., Jennings, S., Parker-Humphreys, M., and Rogers, S. I. 2005. Assessing the status of demersal elasmobranchs in UK waters: a review. Journal of the Marine Biological Association of the United Kingdom, 85: 1025–1047.

FAO. 2003. Fisheries management: 2. The ecosystem approach to fisheries. FAO Technical Guidelines for Responsible Fisheries. 4. 112 pp.

FAO. 2005. Putting into practice: the ecosystem approach to fisheries. ISBN 9789251053966. Food and Agriculture Organization, Rome. 82 pp.

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., Chesson, J., Sainsbury, K., Hundloe, T. J., and Fisher, M. 2005. A flexible and practical framework for reporting on ecologically sustainable development for wild capture fisheries. Fisheries Research, 71: 175–183.

Fock, H. 2011. Integrating multiple pressures at different spatial and temporal scales: a concept for relative ecological risk assessment in the European marine environment. Human and Ecological Risk Assessment, 17: 187–211.

Hobday, A. J., Smith, A., Webb, H., Daley, R., Wayte, S., Bulman, C., Dowdney, J., *et al.* 2007. Ecological risk assessment for the effects of fishing: methodology. Report R04/1072. Australian Fisheries Management Authority, Canberra.

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.

Jennings, S., and Lee, J. 2012. Defining fishing grounds with vessel monitoring system data. ICES Journal of Marine Science, 69: 51–63.

Lynam, C. P., Gibbons, M. J., Bjorn, E. A., Sparks, C. A. J., Coetzee, J., Heywood, B. G., and Brierley, A. S. 2006. Jellyfish overtake fish in a heavily fished ecosystem. Current Biology, 16: R492–R493.

Lythgoe, J., and Lythgoe, G. 1991. Fishes of the sea: the North Atlantic and Mediterranean. MIT Press, Cambridge, MA. 256 pp.

Marasco, R. J., Goodman, D., Grimes, C. B., Lawson, P. W., Punt, A. E., and Quinn, T. J., II. 2007. Ecosystem-based fisheries management: some practical suggestions. Canadian Journal of Fisheries and Aquatic Sciences, 64: 928–939.

Newell, R. I. E., and Ott, J. A. 1999. Macrobenthic communities and eutrophication. *In* Ecosystems at the Land-Sea Margin: Drainage Basin to Coastal Sea, pp. 265–293. Ed. by T. C. Malone, A. Malej, L. W. Harding, N. Smodlaka, and R. E. Turner. American Geophysical Union, Washington, DC.

Onley, D., and Scofield, P. 2007. Albatrosses, Petrels and Shearwaters of the World. Helm Field Guides, London. 240 pp.

Peterson, R., Mountfort, G., and Hollom, P. A. D. 1983. A Field Guide to the Birds of Britain and Europe. Collins, London. 239 pp. + maps.

Pikesley, S. K., Godley, B. J., Ranger, S., Richardson, P. B., and Witt, M. J. 2014. Cnidaria in UK coastal waters: description of spatio-temporal patterns and inter-annual variability. Journal of the Marine Biological Association of the United Kingdom. doi:10.1017/S0025315414000137.

Rees, H. L., Boyd, S. E., Schratzberger, M., and Murray, L. A. 2006. Role of benthic indicators in regulating human activities at sea. Environmental Science and Policy, 9: 496–508.

Rice, J., Arvanitidis, C., Borja, A., Frid, C., Hiddink, J. G., Krause, J., Lorance, P., *et al.* 2012. Indicators for sea-floor integrity under the European Marine Strategy Framework Directive. Ecological Indicators, 12: 174–184.

Scandol, J., Ives, M., and Lockett, M. 2009. Development of national guidelines to improve the application of risk-based methods in the scope, implementation and interpretation of stock assessments for data-poor species. Australian Government FRDC Project No. 2007/016.

Seafish. 2014a. Ecological risk assessment of the effects of fishing for South West fisheries; ICES Divisions VII e, f, g & h. Seafish Report SR 670. http://www.seafish.org/media/Publications/SR670ERAEFFinal_report1.pdf.

Seafish. 2014b. Ecological risk assessment of the effects of fishing for South West fisheries; ICES Divisions VII e, f, g & h. Supporting information. www.seafish.org/media/publications/SR672ERAEFSupporting_information.pdf.

Seafish. 2014c. Ecological risk assessment of the effects of fishing for South West fisheries; ICES Divisions VII e, f, g & h. Assessment spreadsheet. www.seafish.org/media/publications/SR671ERAEF_Assessment_spreadsheet.xlsx.

Shephard, S., Reid, D. G., and Greenstreet, S. P. R. 2011. Interpreting the large fish indicator for the Celtic Sea. ICES Journal of Marine Science, 68: 1963–1972.

Shirihai, H., and Jarrett, B. 2006. Whales, dolphins and seals. A field guide to the marine mammals of the world. A & C Black, London. 384 pp.

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.

Walmsley, S. A., and Pawson, M. G. 2007. The coastal fisheries of England and Wales. Part V: a review of their status 2005–6. Science Series Technical Report No. 140. Centre for Environment, Fisheries and Aquaculture Science, Lowestoft, UK.

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Contribution to the Themed Section: 'Risk Assessment'

Original Article

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

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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.

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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

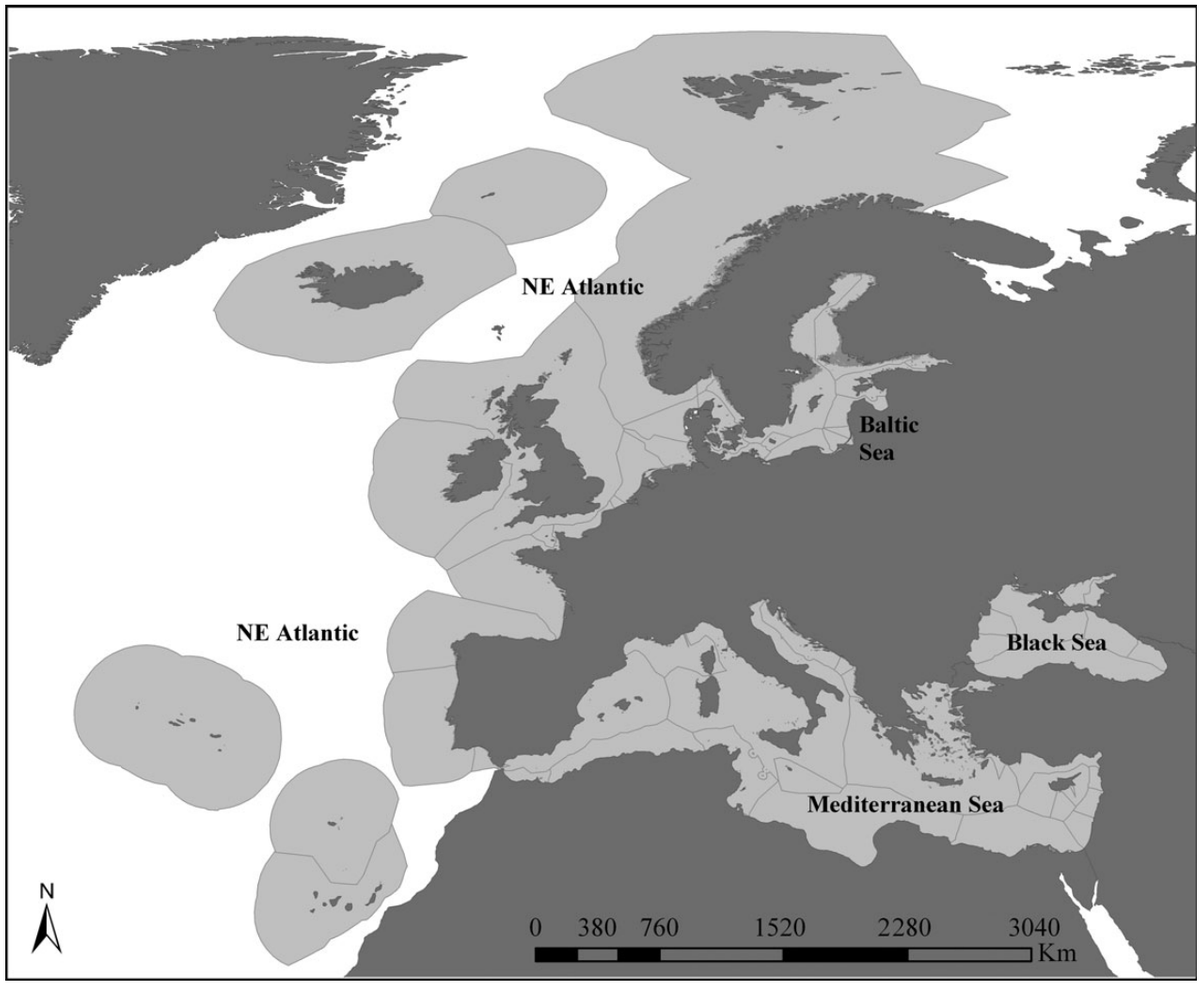

Figure 1. Regional Sea areas of Europe as defined by the MSFD (light grey areas indicate the spatial coverage of the directive). Impact chains were assessed at the scale of the region for the NE Atlantic, Baltic Sea, Black Sea, and Mediterranean Sea. Exclusive economic zone (EEZ) borders are shown.

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).

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

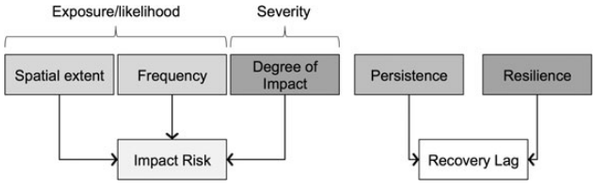

Figure 2. Exposure-effect assessment criteria used in the calculation of risk and RL. Criteria definitions are given in Robinson *et al.* (2013). Definitions: IR is a measure of the likelihood of an adverse ecological impact occurring following a sector–pressure introduction. The greater the IR, the 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).

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).

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

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

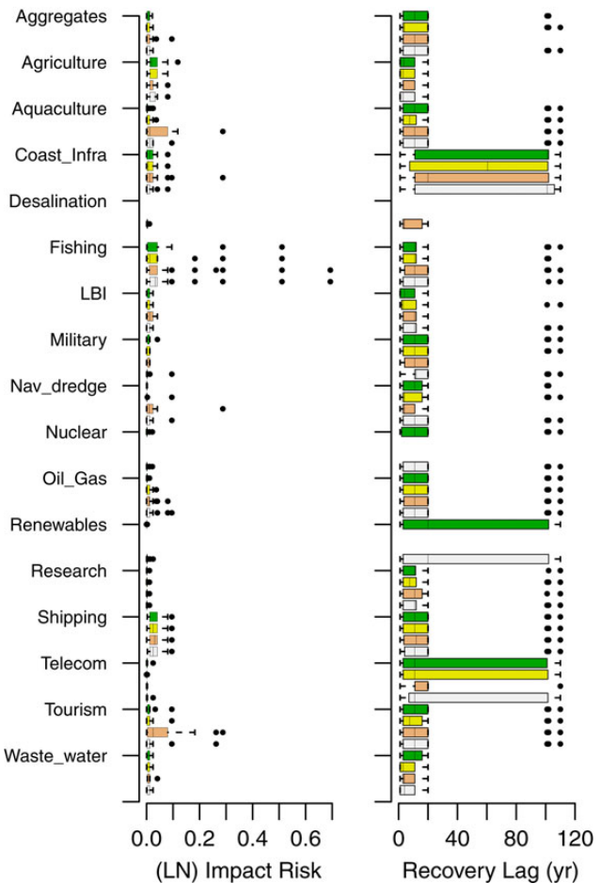

Figure 3. Distribution of IR and RL scores grouped by sector in each of four European regional seas (ordered as Baltic Sea, Black Sea, Mediterranean Sea and NE Atlantic). The maximum IR and RL score for any chain is 0.7 and 1.0, respectively. No bar indicates the absence of the sector in this region.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

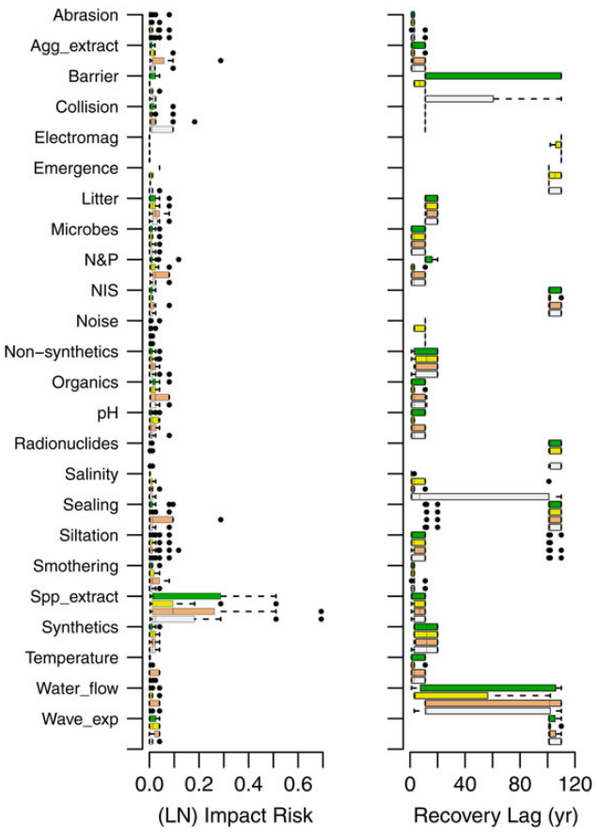

Figure 4. Distribution of IR and RL scores grouped by pressure type in each of four European regional seas (ordered as Baltic Sea, Black Sea, Mediterranean Sea and NE Atlantic). The maximum IR and RL score for any chain is 0.7 and 1.0, respectively. No bar indicates the absence of the pressure in the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

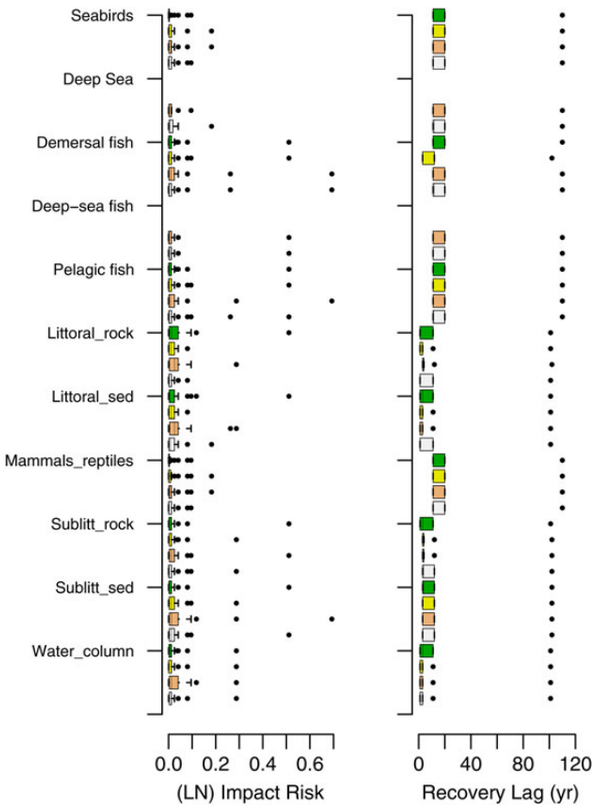

Figure 5. Distribution of IR and RL scores grouped by ecological component in each of four European regional seas (ordered as Baltic Sea, Black Sea, Mediterranean Sea and NE Atlantic). The maximum IR and RL score for any chain is 0.7 and 1.0, respectively. No bar indicates that the ecological component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

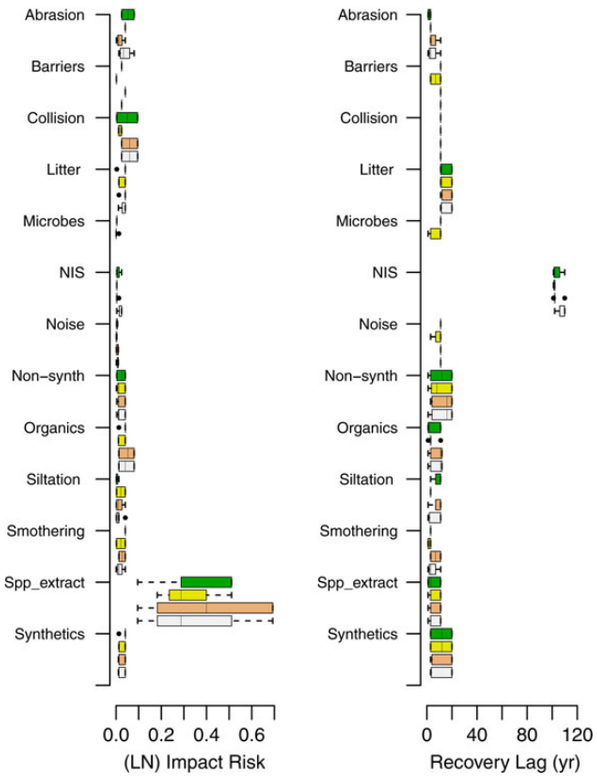

Figure 6. Distribution of IR and RL scores to all ecological components from fishing grouped by pressure in each of four European regional seas (ordered as Baltic Sea, Black Sea, Mediterranean Sea and NE Atlantic). The maximum IR and RL score for any chain is 0.7 and 1.0, respectively. Boxplot information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

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 (~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).

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

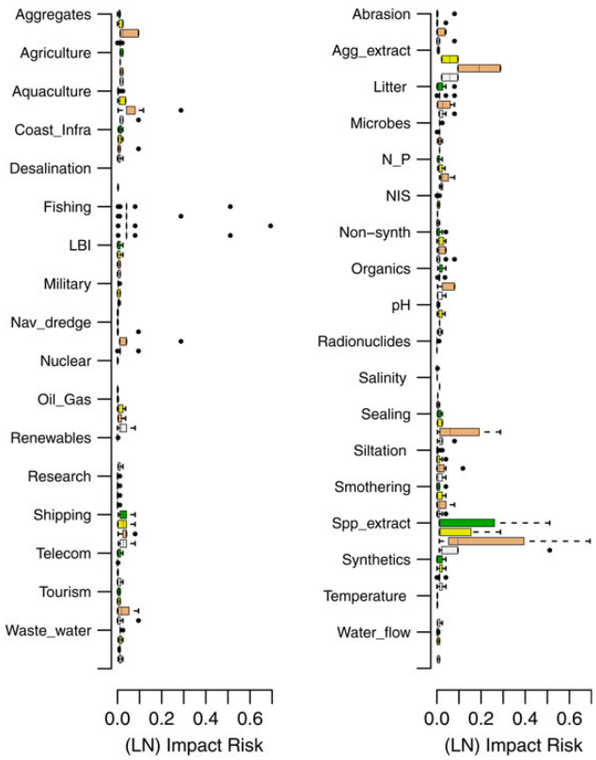

Figure 7. Distribution of IR and RL scores to sublittoral sediments grouped by sector and pressure in each of four European regional seas (ordered as Baltic Sea, Black Sea, Mediterranean Sea and NE Atlantic). Sectors/pressures posing no risk are excluded from the plot. The maximum IR score for any chain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

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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.

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Contribution to the Themed Section: 'Risk Assessment'

Food for Thought

Linking risk factors to risk treatment in ecological risk assessment of marine biodiversity

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Implementing marine ecosystem-based management at regional and small spatial scales is challenging due to the complexity of ecosystems, human activities, their interactions and multilayered governance. Ecological risk assessments (ERAs) of marine biodiversity are often used to prioirtise issues but only give broad guidance of how issues might be addressed in the form of strategies. However, at small and regional spatial scales marine natural resource managers have to make decisions within these strategies about how to manage specific interactions between human uses and ecological components. By using the transition between risk characterization and risk treatment in ERA for marine biodiversity tractable ways through the complexity can be found. This paper will argue that specific management and research actions relevant to smaller spatial scales can be developed by using the linkage between risk factors and risk treatment in ERA. Many risk factors require risk treatments that extend beyond the boundary of local agencies or sector responsibilities. The risk factor-treatment platform provides a practical way that these boundaries can be opened up by providing a scientifically based and transparent process to engage all actors who need to be involved in addressing the issues raised by an ERA. First, the principles of the mechanism will be described. Second, how the mechanism is constructed will be introduced using examples from an urban estuary. Application of the mechanism reveals three different types of risk factors (stressor, ecological, and knowledge gap) that can be used to develop specific management and research actions to treat risks. The systematic approach enables the dual complexities of marine ecosystems and multiple human pressures to be unravelled to identify and target issues effectively. The risk factor treatment linkage provides a platform to negotiate and develop effective management and research actions across jurisdictional, disciplinary, community and stakeholder boundaries.

Keywords: ecological risk assessment, marine ecosystem-based management, risk factor, risk treatment, small scale.

Introduction

One of the great challenges in implementing marine ecosystem-based management (MEBM) is determining what management and research actions will be effective in addressing specific issues at regional and small spatial scales (Cook *et al.*, 2013). Ecological risk assessments (ERAs) for marine biodiversity are often used to prioritize issues but only give broad guidance of how issues might be addressed in the form of strategies (e.g. Hobday *et al.*, 2011; Williams *et al.*, 2011; Samhorni and Levin, 2012). However, at small and regional spatial scales marine natural resource managers still have to make decisions within these strategies about how to manage specific interactions between human uses and ecological components, such as whether to allow foreshore constructions (e.g. marinas) that can potentially have direct and indirect effects on the sustainability of marine biodiversity (Clynick, 2008; Di Franco *et al.*, 2011). In essence, they need to know what to manage, why and how to manage it (Wilson *et al.*, 2007; Astles, 2008; Game *et al.*, 2013). Similarly, scientists need to decide which research questions are the most important to answer to provide specific support to marine natural resource managers to develop effective management actions (McNie, 2007).

Two other factors add to the difficulty of marine natural resource management (MNRM) at small and regional spatial scales. First, there are multiple human uses interacting within the same space and time. Each use has multiple stressors that potentially interact,

directly and indirectly, with multiple ecological components (e.g. Vinebrooke *et al.*, 2004). Therefore, identifying and prioritizing effective management actions are significantly more complex in these contexts, in contrast to single sector marine management (e.g. trawl fisheries). Second, there are often multiple and interacting layers of governance at small and regional spatial scales combined with diverse community and stakeholder groups (e.g. Lazarow *et al.*, 2006; Voyer *et al.*, 2012). This situation occurs most often in highly urbanized estuaries and coastal areas where human uses are intensified. For example, in the Hawkesbury estuary on the east coast of Australia, there are several complex and interacting layers of governance that have jurisdiction over the estuary. This includes three local governments, four state government agencies, and a recently established overarching state marine management authority. All these levels of government are responsible for implementing state and federal legislation and policies that impact the management of marine ecosystems and biodiversity (Clarke *et al.*, 2013), in addition to addressing local and regional issues. Interacting with these different layers of governance is multiple industry stakeholder, indigenous, and local community groups (Haines *et al.*, 2008). Therefore, there is no single agency or community group with sole responsibility for the natural resource management of the estuary, which is typical of urban estuaries within Australia (Lazarow *et al.*, 2006; Stocker *et al.*, 2012). These complex governance environments make implementing management and research actions to sustain marine biodiversity consistently across a whole estuary or coastal area extremely challenging (Stocker *et al.*, 2012).

Tractable ways through this complexity can be found using the transition between risk characterization and risk treatment in ERA for marine biodiversity. This paper will argue that specific management and research actions relevant to smaller spatial scales can be developed using the linkage between risk factors and risk treatment in ERA for marine biodiversity. First, the principles of the mechanism will be described. Second, how the mechanism is constructed will be introduced using examples from an urban estuary. Finally, the paper will discuss how the mechanism can be applied to assist meeting the complex challenges of MEBM for marine biodiversity at smaller spatial scales, its advantages, challenges, and areas of future development.

Risk factors and risk treatment

The World Health Organisation defines a risk factor as any attribute, characteristic, or exposure of an individual that increases the likelihood of developing a disease or injury (www.who.int/topics/risk_factors) and are differentiated into types based on their strength of correlation to an outcome and their response to manipulation (e.g. Kraemer *et al*, 1997, 2001). These factors are used to develop treatments for the management of diseases and injuries (e.g. Kazdin, 2007). The linkage between risk factors and treatments gives clinicians leverage in addressing issues efficaciously. An analogous process in MEBM is the manager making decisions about how to mitigate (i.e. treat) human impacts on marine ecological components. If it is known what is contributing to these impacts, management actions can be developed and implemented that targets these issues to reduce or modify the impacts (e.g. bycatch reduction devices to reduce the catch of non-target species in trawl fisheries; Dayton *et al.*, 1995; Broadhurst *et al.*, 1997). Such points of leverage underpin the effectiveness of management and research for single sector human activities.

A risk factor in marine ERA is any attribute or characteristic of an ecological component or exposure of a human activity stressor that increases the likelihood of an impact occurring (adapting the WHO definition). I surveyed ERA papers in the fields of marine ecology and ecotoxicology from 1980 to 2013 to determine the extent to which this term or similar has been used. I found 27% used terms that fit this basic definition Of these papers, 58.3% of ERA studies on marine non-native invasive species and 50% of ERA studies on marine ecosystems and biodiversity used concepts equivalent to risk factor (e.g. Hayes and Landis (2004) used "risk predictors" and "contributors to risk"). However, few of the papers reviewed directly linked these contributors to the treatment of risk, that is, specific management and research actions that could reduce, mitigate, or modify the risk to a marine ecosystem or ecological component. Rather, links were made to potential generalized management strategies, such as spatial management (e.g. Halpern *et al.*, 2007, 2009). Specific management actions in response to factors contributing to risk levels, that is, risk treatments, were mainly identified with respect to a single type of human pressure (HP), such as commercial fishing (e.g. Pitcher, 2014), or similar types of stressors on particular marine organisms and habitats, such as contaminants in marine sediments (e.g. Brown *et al.*, 2013).

This paper describes how different types of risk factors are extracted from an ERA method for marine biodiversity and how specific risk treatments can be developed to address these factors applicable to regional and small spatial scales. For the purposes of this paper, marine biodiversity was defined as the variety of species, assemblages, habitats, and ecosystems in marine and estuarine waters. Ecological components are the individual components that make up this diversity such as a type of habitat or species.

Estimating ecological risk to marine biodiversity

A complete description of the ERA method used for marine biodiversity is given in Astles (2010) and illustrated in Figure 1. It is a quantitative development of the method used for assessing the risk from commercial fisheries to fish species and habitats (Astles *et al.*, 2006, 2009). For this paper, only the risk characterization step will be described in detail in the interests of keeping the length of the paper manageable. The risk context was determined by a subset of the management goals of one of the local governments with oversight for the Hawkesbury estuary, New South Wales on the east coast of Australia. Their goal was to conserve, protect, and enhance sustainable economic, recreational, and social issues without compromising the high-quality and functional estuarine ecosystems upon which they rely (Haines *et al.*, 2008). Therefore, the risk that was being assessed for the Hawkesbury estuary was the likelihood that current human activities in the estuary will lead to estuarine habitats becoming degraded such that the biodiversity they support is unable to sustain its current abundance and distribution in the estuary in the next 20 years. The time frame was specified by the council's estuary management plan (Haines *et al.*, 2008).

The level of risk was determined as the likelihood that an interaction between a human activity and an ecosystem component will result in the undesirable outcome, i.e. consequence, that the goals of the management plan was seeking to avoid. For example, the risk to seagrass from recreational boating is the likelihood that seagrass will not be able to maintain its current abundance and distribution within an estuary for the next 20 years as a result of its interactions with recreational boating. The likelihood was estimated by determining the pressure being exerted by a human activity on an ecological component and the capacity of an ecological component to respond to that pressure. Therefore, two sets of information were

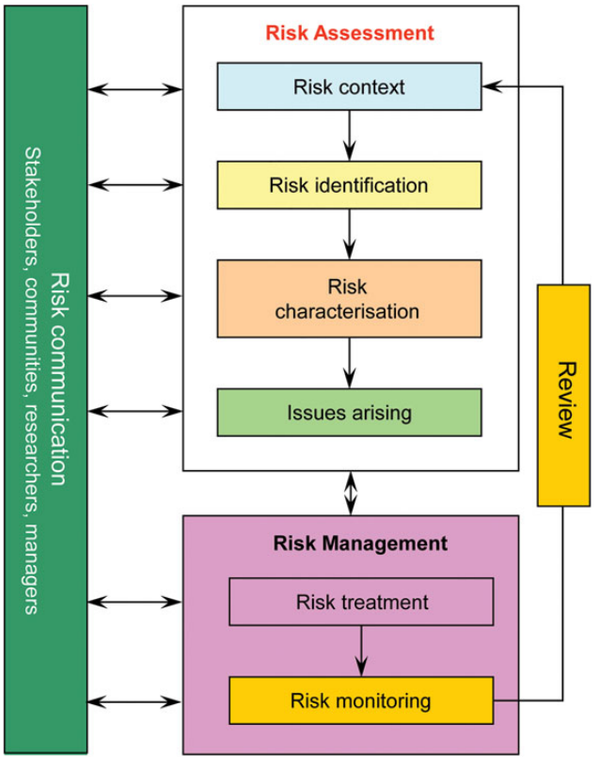

Figure 1. Framework for an ecological risk analysis of marine biodiversity consistent with AS/NZS ISO 31000:2009. Standards Australia (2009).

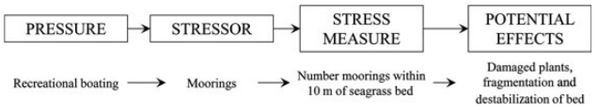

Figure 2. Relationship between pressure, stressor, stress measure and potential outcome and an example for recreational boating on seagrass habitat.

used to estimate the risk to an ecological component of marine biodiversity—HP and capacity to respond (CTR).

HP was an activity that directly or indirectly interacts with an ecological component. The level of HP was determined by examining the relationship between its stressors and their potential effects on an ecological component (Hughes and Connell, 1999; Scanes *et al.*, 2007) (Figure 2). These relationships were sourced from the scientific literature (Table 1) so that the basis of the choices made was on documented scientific studies open to scrutiny and critique. Stressors are the agents of a human activity that, when they reach a critical range, can result in a change in the structure and/or function of an ecological component. These included both extant stressors and historical legacies still operating (e.g. Knott *et al.*, 2009). Each stressor had a measure of magnitude, duration, distribution, and/or frequency (which maybe unknown for many stressors). Potential effects were the changes to the structure and function of an ecosystem component as a result of its interaction with the stressors (e.g. Hallac *et al.*, 2012).

Each HP usually had more than one stressor. To avoid correlations among stressors and overestimating the level of pressure one stressor for each type of potential impact was used (see Table 1). Therefore, each stressor was treated as additive. Stress measures were quantitative or qualitative information, such as presence or absence. Measures were either direct (e.g. the number of boats travelling over a shallow seagrass bed within an estuary over a year) or indirect (e.g. the number of boat ramps within 50 m of a seagrass habitat and proportion of boating visitors to the area). The choice of which measure to use depended on the data available, the resources (time, money and expertise) required to obtain data and to what extent it directly or indirectly measured the stressor. Every stressor had a measure, even if there was no information available for a particular measure. All measures were standardized to the spatial scale of the assessment area. Measures that were unknown were extracted later as knowledge gaps.

CTR to a HP by an ecological component is its ability to maintain, recover, or adapt its structure and/or function as a result of its interaction with a HP. Ecological components can be affected by an interaction with a HP in three ways—inert (no change in structure or function), natural (change but within current spatial and temporal variability), or impacted (change outside its current spatial and temporal variability) (Underwood, 1989). The CTR of an ecological component is governed by the type and magnitude of the impact, the spatial and temporal scale of the impact, the inherent characteristics of the ecological component, its current condition, and the spatial and temporal scales of its recovery (Underwood, 1989; Glasby and Underwood, 1996). Therefore, the CTR was assessed using three aspects: the characteristics of an ecological component that would enable it to maintain, recover, or adapt its structure and function, its current condition, management effectiveness, and a specified spatial and temporal scale of recovery. Importantly, the CTR was not solely based on inherent ecological or biological characteristics, as has been critiqued in other studies (e.g. Pitcher, 2014), but included the local context (condition and management) in which the ecological component operates.

The contribution of these three aspects made to its CTR was assessed relative to a magnitude and the spatial and temporal scales of a specified natural disturbance. Therefore, CTR was a measure of an ecological component's response to a hypothetical natural disturbance of a specified magnitude (Minchinton, 2007), allowing a standardized level of impact to be applied to each ecological component being assessed. In the Hawkesbury estuary, a hypothetical natural disturbance was defined as an event or series of events (e.g. storm, flood, natural dieback, and ecological interactions) that resulted in a $\geq$50% depletion in a component's abundance, distribution, and/or function within a 12-month period at the spatial scale of sub-catchments within the estuary.

The level of CTR of an ecological component was determined by examining the relationships between its functions, characteristics, and potential contribution to its ability to return to its prior variability in abundance and distribution and/or function (Figure 3). These were sourced from the scientific literature so that the basis of the choices made was on documented scientific studies open to scrutiny and critique (Table 2). Functions are the biological, geomorphological, hydrological, and/or biogeochemical processes of an ecological component. Characteristics are the individual attributes of a function that contribute to an ecological component's CTR in time and space. For example, the function of growth for seagrass includes the characteristics of leaf extension, rhizome extension, and above-ground biomass. Functions and characteristics of

Table 1. An example of stressors, stress measures, and rationale for foreshore development for an estuary.

Disturbance category	Stressor	Stress measure	Rationale
Inputs	Intensity—urban/industrial	Proportion of urbanized/industrialize catchment per surface area of estuary/sub-catchment per area of habitat	A collective measure of the amount of potential stress from urban and industrial development, including changes to shoreline.
Biomass	Mooring damage	Proportion of moorings within 10 m of vegetated habitat	A measure of the stress that can occur from increased human activity and direct damage from mooring chains on soft sediment habitats, including seagrass (Demers *et al.* , 2013).
Physical structure	Change of hardness and slope of shoreline	Proportion of artificial shoreline per total perimeter of estuary/region within 10 m of a habitat	A measure of the stress that can occur from changed slope and hardness of foreshore such as increased water turbulence (Bulleri, 2005).
	Seawall type	Proportion of habitat friendly seawalls per length of artificial shoreline	A measure of the amount of artificial habitat that is suitable for marine biodiversity (Clynick *et al.*, 2009).
	Infrastructure maintenance	Frequency of maintenance of instream infrastructure	A measure of the stress from maintenance activities on instream infrastructure.
Ecosystem/ecological function	Groundwater pressure—regional	Regional groundwater level per area of habitat	Groundwater levels affect below ground processes important for maintaining below group biomass of saltmarsh and mangrove habitat. Increased human extraction can affect levels (New South Wales Government, 2010).
	Groundwater pressure—local	Local groundwater level per area of habitat	Groundwater levels affect below ground processes important for maintaining below ground biomass of saltmarsh and mangrove habitat. Increased human extraction can affect levels (New South Wales Government, 2010).
	Groundwater pressure—aquifer	Aquifer pressure structure per area of habitat	Aquifers feeding groundwater support below ground processes for maintaining below ground biomass of saltmarsh and mangrove habitat. In urbanised or mining catchments these can become degraded resulting in contaminants being transported to these habitats (New South Wales Government, 2010).
	Water extraction[1]	Volume of groundwater or surface water extraction per surface area of estuary/sub-catchment per area of habitat	An alternative measure of the combined effects of groundwater extraction. Could be used if more specific information is not available.
	Invasive species	Number of artificial habitats including pilings, wharves, jetties, and pontoons per area of estuary/region	A measure of the artificial habitat available that could be colonized by invasive species. This includes oyster lease infrastructure (Glasby *et al.*, 2007).
	Changes to connectivity	Proportion of perimeter of habitat adjacent non-native or disturbed areas (urban, industrial, agriculture, instream structures, and disturbed habitat)	A measure of the extent to which foreshore development has disconnected habitats (Meynecke *et al.*, 2008).
	Change of flow and tidal regimes, fish passage	Number of dams, weirs or flood gates within the tidal range of creeks/rivers per surface area of estuary plus the proportion of species potentially using these creeks	A measure of inhibition to fish movement into tributaries of estuaries and the number of fish spp. in an estuary that use these habitats (Boys *et al.*, 2012).
Protected spp.	Urbanisation	Number of intertidal wetlands within 100 m of an urbanised area	A measure of the disturbance from urban areas to shorebird foraging areas, such as artificial illumination to nocturnal birds (Santos *et al.*, 2010).
Climate change	Sea level rise mitigation	Projected percentage increase in shoreline artificial structures per area of estuary	A measure of the increased stress from armouring of foreshore for flood and sea level rise mitigation (Clynick *et al.*, 2009).
	Increased water extraction during droughts	Projected percentage increase in groundwater extraction per area of estuary	A measure of increased stress on below ground processes affecting below ground biomass and surface elevation (Koehn *et al.*, 2011).
	Increased land clearing or back burning for bush fire control	Projected percentage increase in land clearing or area of back burning within the catchment per area of estuary	A measure of stress from increased run-off and sedimentation from the catchment (Gilman *et al.*, 2008).

Note: Alternative stress measure.

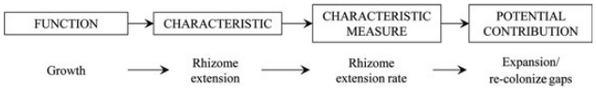

Figure 3. Relationship between a function, characteristic, its measure and potential contribution to a component's response to natural disturbance and an example for seagrass.

an ecological component were determined over the range of biological organization relevant to the management context. For example, the spatial scales of organization for seagrass include individual plants, individual patches, or beds and multiple patches across a whole estuary. Characteristics included the condition of ecological components (e.g. areal extent, proportion of habitat loss over 10 years Astles, 2010) and management effectiveness (e.g. proportion of habitat in protected areas) at the relevant spatial and temporal scales.

Each function of an ecological component will usually have more than one characteristic that contributes to its CTR. As for stressors, correlations among characteristics were eliminated so as to not over or underestimate the CTR of an ecological component by selecting one characteristic per type of contribution (see Table 2). Each characteristic may have several ways they can be measured, so only one measure per characteristic was chosen. As for stressors, every characteristic was given a measure even if information was unavailable and measures that were unknown were extracted later as knowledge gaps. All measures were standardized to the spatial scale of the assessment area.

Determining the level of HP, CTR and risk

The contribution each stressor and characteristic made to the HP and CTR, respectively, was determined using decision criteria. These criteria can be determined in two ways—absolutely and relatively. Absolute criteria are derived from the scientific literature, standards and guidelines, and government reports. Absolute criteria are independent of the specific context of the ERA (e.g. Samhouri and Levin, 2012; Pitcher, 2014). However, such information is often unavailable or not applicable to a particular region (e.g. Scanes *et al.*, 2007) and relative criteria are used instead. Relative criteria are set by determining the range of the values of a stress or characteristic measure within the region of the assessment and are benchmarked to levels of human disturbance (for stressors) and/or levels of capacity (for ecological characteristics). For example, within the Hawkesbury estuary, the Pittwater sub-catchment had the highest level of human disturbance based on the number of human activities present and Mangrove and Mullet sub-catchments the lowest level. Similarly, the largest proportional area of undisturbed seagrass habitat occurred in the Patonga sub-catchment and was therefore benchmarked as having the largest potential CTR for seagrass within the Hawkesbury estuary. When no independent data were available decision criteria were set by taking a conservative value of 40% of the range of each stress measure and 60% of the range of each characteristic measure that occurred within the estuary (Tables 3 and 4). Thus, a stressor >40% was considered contributing to the pressure being exerted and a characteristic <60% was considered contributing to making an ecological component less CTR. Forty and 60% was used for the Hawkesbury ERA to bias the decision criteria towards detecting an interaction and minimize type II errors (not detecting an interaction when one has occurred). Alternatively, relative criteria can be determined by using the range of values for measures in other estuaries or coastal regions benchmarked to levels of human disturbance, for example, estuaries or coastal regions next to national parks compared with estuaries or coastal regions with high human population densities or agricultural development.

All the information used to determine decision criteria, i.e. scientific literature and/or government reports and data, were documented and made explicit. This made the criteria upon which decisions were made transparent and open to scrutiny (Astles, 2008; Samhouri and Levin, 2012; Goble and Bier, 2013) in contrast to methods that rely on group deliberation to determine levels of risk (e.g. see discussion of issues in Drescher *et al.*, 2013).

A binomial score of either 0 or 1 was given to a measure that did not or did exceed, respectively, the decision criterion. A binomial structure was used to eliminate one form of linguistic uncertainty, ambiguity (Regan *et al.*, 2002; Hayes, 2011), that often occurs in descriptive criteria of risk level components (e.g. Fletcher, 2005; Halpern *et al.*, 2007). Any measures for which there was no data were allocated a "U". These unknown measures were included in estimating the level of HP or CTR. Lack of information contributes to the analytical uncertainty (Suter *et al.*, 1987) in estimating the level of risk and needs to be incorporated to account for the possibility of not detecting an effect when one has occurred.

The level of *HP* for an activity was calculated as a proportion as:

$$\mathrm{HP} = \frac{\sum_{n=1}^{s} S_i + \sum_{h=1}^{u} S_u}{N_s}$$

where S_i is a stressor that exceeded the decision criteria, s is the total number of stressors that exceeded the criteria, S_u are the stressors with unknown values, u is the total number of stressors with unknown values, and N_s is the total number of stressors evaluated for the HP.

The level of *CTR* for an ecological component was calculated as a proportion as:

$$\mathrm{CTR} = \frac{\sum_{k=1}^{c} C_i + \sum_{l=1}^{v} C_u}{N_c}$$

where C_i is a characteristic that exceeded the decision criteria, c is the total number of characteristics that exceeded the criteria, C_u is a characteristic with unknown values, v is the total number of characteristics with unknown values, and N_c is the total number of characteristics evaluated for the ecological component.

The risk level (R) for each human activity and ecological component interaction was calculated as the Euclidean distance from the origin (0,0) in a space defined by *HP* and *CTR* values:

$$R = \sqrt{(\mathrm{HP} - 0)^2 + (\mathrm{CTR} - 0)^2}$$

The risk to an ecological component increases with increasing distance from the origin which was categorized using a 5 × 5 matrix (Figure 4). The design of the matrix conforms to the mathematical rules as set out by Cox (2008).

Identifying risk factors for ecological components at highest levels of risk

For those combinations of ecological components and human activity at high levels of risk, the factors contributing to that risk were determined. Risk factors were of three types. First, stressor

Table 2. An example of functions, characteristics, measures, and rationale for mangrove habitats for an estuary.

Organizational scale	Function	Characteristic	Measure	Rationale
Plants	Reproduction	Recruitment/ establishment	Initial shoot growth in relation to salinity	A measure of a mangrove's ability to establish and colonize in a suitable habitat. Shoot initiation is a stronger predictor of mangrove establishment than life history traits such as dispersal ability (Clarke *et al.*, 2001).
Individual stands	Composition	Diversity—species	Number of species present in estuary/ region compared to expected	Different species provide different structural and biogeochemical properties for marine biodiversity (Melville and Burchett, 2002; Melville *et al.*, 2004).
		Diversity regional—genetic	Genetic diversity of mangrove species increases with distance between estuaries	A measure of how restricted the gene flow is between mangrove stands and assemblages. The more restricted the less resilient to estuary wide impacts on the population (Melville and Burchett, 2002; Melville *et al.*, 2004).
		Diversity local—genetic	Genetic diversity of mangrove species within estuaries	A measure of whether there are multiple sources of genetic material within the estuary. The more sources the greater potential for adaption and CTR (Melville and Burchett, 2002; Melville *et al.*, 2004).
Multiple stands	Biomass	Abundance	Total area of mangrove per total area of intertidal habitats available	A measure of the amount of mangrove habitat available for marine biodiversity to use.
			Percentage change in mangrove area over 5 years	A measure of the amount of mangrove lost or gained outside expected levels of change. A gain in mangrove habitat may indicate a corresponding loss of saltmarsh habitat (Williams and Thiebaud, 2007).
	Growth	Sediment processes	Proportion of total area of mangrove spp with aerial roots (e.g. pneumatophores)	A measure of the surface structure available to accrete sediment and support above and below ground biomass production (Rogers *et al.*, 2006).
			Density of trees	An alternative measure of the surface structure available to accrete sediment.
		Erosion	Proportion of mangrove area eroded	A measure of the loss of suitable habitat for mangrove to occupy
	Survival	Surface elevation maintenance	Percentage change in surface elevation over last 5 years	A measure of the trend of mangrove stand to remain within a suitable tidal range (Rogers *et al.*, 2006).
	Connectivity	Mangrove-saltmarsh	Proportion of length of connected edge between saltmarsh and mangrove over the total length of terrestrial edge of mangrove	A measure of the potential of flow of energy, organic matter and other biological material between habitat types. The longer the connected edge the more resilient (Meynecke *et al.*, 2008; Beger *et al.*, 2010).
		Mangrove—water	Proportion of length of connected edge between mangrove and seagrass habitat edge within 50 m over the total length of water edge of mangrove	A measure of the potential of flow of energy, organic matter and other biological material between habitat types. The longer the connected edge the more resilient (Meynecke *et al.*, 2008; Beger *et al.*, 2010).
	Climate change response	Increased air temperature	Projected or actual percentage change in mangrove dieback per area of estuary	A measure of the effect of increasing dry conditions from higher than average temperatures. Boon *et al.* (2010).
		Salinity changes	Apical shoot initiation salinity	A measure of the sensitivity of reproductive propagules to changes in salinity (Clarke *et al.*, 2001).

Continued

Table 2. *Continued*

Organizational scale	Function	Characteristic	Measure	Rationale
		Freshwater flow/inputs	Percentage change in proportion of tidal river length occupied by mangroves	An alternative to changes in salinity as a measure of the extension upstream of mangrove habitat due to changes in freshwater flow and tidal intrusion.
		Sea level rise	Proportion of terrestrial edge of mangrove with a natural barrier within 10 m of terrestrial interface. A natural barrier is any change in slope >5 degrees for >=50% of mangrove edge	A measure of the potential for mangroves to move upslope in response to sea level rise (Gilman *et al.*, 2008).
		Species interactions	Number of species with biological characteristics potentially able to adapt to changes in climatic conditions	A measure of the potential for a change in the dominance of species occupying space due to more favourable conditions as a result of climate change (Ruiz *et al.*, 1997).

Table 3. Example of decision criteria used for the Pittwater sub-catchment of the Hawkesbury estuary and results for three different habitats.

			Seagrass		Mangroves		Mudflats	
Stressor	**Stress measure**	**Decision criteria**	**Data**	**Score**	**Data**	**Score**	**Data**	**Score**
Intensity—urban/industrial	Proportion of unsewered foreshore housing per surface area of sub-catchment per area of habitat	>0.1	0.06	0	0	0	0	0
Mooring damage	Proportion of moorings within 10 m of vegetated habitat	>0.1	0.69	1	0	0	NA	
Change of hardness and slope of shoreline	Proportion of artificial shoreline per total perimeter of sub-catchment within 10 m of a habitat	>0.1	0.40	1	0.32	1	0	0
Seawall type	Proportion of habitat friendly seawalls per length of artificial shoreline within 10 m of a habitat	<0.4	0	1	0	0	0	0
Infrastructure maintenance	Frequency and duration of maintenance of instream infrastructure in sub-catchment per year per area of habitat	4 per year of 0.5 day duration	U	U	U	U	U	U
Groundwater pressure—regional	Regional groundwater pressure per area of habitat	>66% LTAAEL[1]	33%	0	33%	0	33%	0
Groundwater pressure—local	Local groundwater pressure per area of habitat	>66% LTAAEL[1]	66%	1	66%	1	66%	1
Groundwater pressure—aquifer	Aquifer structure pressure per area of habitat	>66% LTAAEL[1]	125%	1	125%	1	125%	1
Invasive species	Proportion of artificial habitats including pilings, wharves, jetties, and pontoons per area of sub-catchment within 10 m of a habitat	>0.1	0.51	1	0.03	0	0	0
Changes to connectivity	Proportion of perimeter of habitat within 50 m of non-native or disturbed areas (urban, industrial, agriculture, instream structures, and disturbed habitat)	>0.02	0.04	1	0.006	0	0.04	1
Change of flow and tidal regimes, fish passage	Number of dams, weirs or flood gates within the tidal range of creeks/rivers per surface area of sub-catchment plus the proportion of species within estuary potentially using these creeks	>0.4	U	U	U	U	U	U
Urbanization	Proportion of intertidal habitat within 100 m of an urbanised area	>0.1	0.04	0	0.06	0	0.04	0
Sea level rise mitigation	Projected percentage increase in artificial shoreline per area of habitat	>10	U	U	U	U	U	U
Increased water extraction during droughts	Projected percentage increase in groundwater extraction per area of habitat	>10	U	U	U	U	U	U
Increased land clearing or back burning for bush fire control	Projected percentage increase in land clearing or area of back burning within the catchment per area of sub-catchment per area of habitat	>5	U	U	U	U	U	U
	Total stress measures used			15		15		14
	Total stress measures > criteria			7		3		3
	Total unknown stress measures			5		5		5
	Proportion stress measures > criteria			0.47		0.20		0.21
	Proportion unknown stress measures			0.33		0.33		0.36
	Total pressure (HP)			0.80		0.53		0.57

NA—not applicable, 0—does not exceed criteria, 1—exceeds criteria, U—unknown, no data available.
Note: LTAAEL—Long term annual average extraction limit vs. entitlement. Source: New South Wales Government (2010).

Table 4. Example of decision criteria and results used for mangrove habitat in the Pittwater sub-catchment of the Hawkesbury estuary for two species of mangrove.

Characteristic	Measure	Decision criteria	Mangrove, all species		*Avicennia marina*		*Aegiceras corniculatum*	
			Data	Score	Data	Score	Data	Score
Recruitment/ establishment	Initial shoot growth in relation to salinity	<60% initiation at 100% seawater	NA	NA	90.0	0	0.01	1
Diversity—species	Number of species present in estuary/region compared to expected	<2	2	0	NA	NA	NA	NA
Diversity regional—genetic	Genetic diversity of mangrove species increases with distance between estuaries	U	NA	NA	U	U	U	U
Diversity local—genetic	Genetic diversity of mangrove species within estuaries	U	NA	NA	U	U	U	U
Abundance	Total area of mangrove per total area of intertidal habitats available	>0.29	0.16	1	NA	NA	NA	NA
	Percentage change in mangrove area over 5 years	>10 ± %	−3%	0	NA	NA	NA	NA
Sediment processes	Tree density per surface area of sub-catchment	<0.6	U	U	NA	NA	NA	NA
Erosion	Proportion of mangrove area eroded	>0.4	U	U	NA	NA	NA	NA
Surface elevation maintenance	Rate of surface elevation over last 5 years	<3 mm yr^{-1}	5.64	0	NA	NA	NA	NA
Mangrove—saltmarsh	Proportion of length of connected edge between saltmarsh and mangrove over the total length of terrestrial edge of mangrove	<0.6	U	U	NA	NA	NA	NA
Mangrove—water	Proportion of length of connected edge between mangrove and seagrass habitat edge within 50 m over the total length of water edge of mangrove	<0.6	U	U	NA	NA	NA	NA
Increased air temperature	Projected or actual percentage change in mangrove dieback per area of estuary	U	U	U	NA	NA	NA	NA
Salinity changes	Optimum apical shoot initiation salinity	<60% sw	NA	NA	All salinities	0	5% sw	1
Freshwater flow/ inputs	Percentage change in proportion of tidal river length occupied by mangroves	>0.4	U	U		NA		NA
Sea level rise	Proportion of terrestrial edge of mangrove with a natural barrier within 10 m of terrestrial interface. A natural barrier is any change in slope >5 degrees for ≥50% of mangrove edge	>0.4	U	U		NA		NA
Species interactions	Number of species with biological characteristics potentially able to adapt to changes in climatic conditions.	U	U	U		NA		NA
	Total characteristic measures used			12		4		4
	Total characteristic measures > criteria			1		0		2
	Total unknown characteristic measures			8		2		2
	Proportion characteristic measures > criteria			0.08		0.06		0.13
	Proportion unknown characteristic measures			0.67		0.63		0.63
	Total Measures			0.75		0.69		0.75
	Total CTR (1 − total measures)			0.25		0.31		0.25

If measure exceeds criteria it is less resilient. NA—not applicable, 0—does not exceed criteria, 1—exceeds criteria, U—unknown, no data available, sw - seawater.

risk factors are the stressors of human activities that are exerting pressure on an ecological component that increases the likelihood of an impact. These risk factors were identified by extracting all stressors for a human activity that exceeded their decision criteria for a particular ecological component, prioritized by the level of risk (i.e. ecological components at the highest levels of risk first) (Table 3).

Second, ecological risk factors are the characteristics of an ecological component that decrease its CTR to a particular HP thereby increasing the likelihood of being unable to maintain its current structure and function as a result of the pressure from the human activity. As for stressor risk factors, these ecological risk factors were identified by extracting all characteristics that exceed their decision criteria (Table 4).

Third, knowledge gap risk factors are the stressors and characteristics for which information or data are lacking. These risk factors contribute to the likelihood of an impact occurring because the effects of some aspects of the interaction between a human activity and ecological component are unknown. This lack of knowledge contributes to what Game *et al.* (2013) call the risk of failure of MEBM objectives at regional and local scales. Knowledge gap risk factors were identified by extracting all characteristics and stressors that were marked as unknown (Tables 3 and 4). All three types of risk factors were collated and summarized in the issues arising stage of

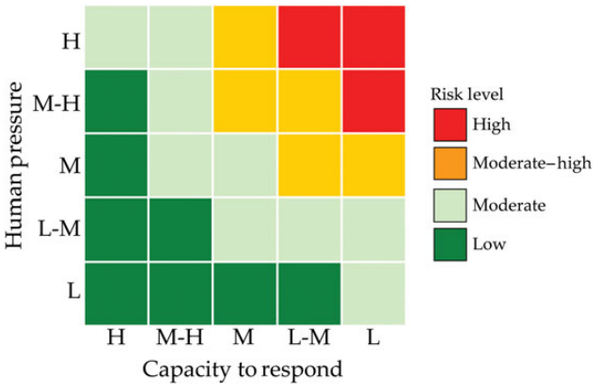

Figure 4. Risk matrix used to determine level of risk. H, high; H-M, high moderate; M, moderate; L-H, low moderate; L—low.

the ERA method (Figure 1; Astles, 2010). An example is shown in Table 5 for a sub-catchment of the Hawkesbury estuary.

Linking risk factors to risk treatments

Risk treatments are the specific management and research actions that are designed to modify, mitigate, or reduce the level of risk to ecological components (e.g. Sutherland *et al.*, 2014) which in turn lowers the risk of not achieving management objectives (Astles, 2012). The different types of risk factors enable development of effective risk treatments. As a minimum, to be effective risk treatments need to do three things: (i) match the most appropriate set of management and research tools to the risk factors (Halpern *et al.*, 2010); (ii) identify and resource the groups (government, non-government, and communities) best able to implement those tools (Stocker *et al.*, 2012); and (iii) monitor, assess, and learn from the outcomes of the management and research actions on the ecological components and/or ecosystem (Underwood, 1995; Allan *et al.*, 2013). It must be emphasized to note that implementation and prioritization of management and research tools to treat risk factors are context dependent, especially at small and regional spatial scales. The examples below of how risk factors can be used to determine what, where, and how to implement management and research tools are not intended to allude to definitive rules. Rather, they illustrate how stressor and risk factors could be linked to risk treatments.

Matching management and research tools to risk factors

Stressor risk factors assist in identifying an appropriate range of management tools to address issues for ecological components at high levels of risk. For example, in the Cowan sub-catchment of the Hawkesbury estuary seagrass habitat was at high risk from the HP of foreshore development from three stressor risk factors that were within 10 m of seagrass beds—artificial rockwalls, unsewered foreshore housing and proportion of wharves and jetties. Table 6 lists the range of management tools that could be used to treat these risk factors and the research tools that could be applied to monitor and assess their effectiveness in the Cowan sub-catchment. In another sub-catchment of the Hawkesbury (Berowra) under an adjacent local government jurisdiction, different stressor risk factors were acting on different habitats. Intertidal mudflats and rocky reef were at high risk from recreational fishing activity from the stressor of bait collection. This requires developing a different set of management and research tools targeted at assessing the impact of bait collection on these habitats. In this manner, the stressor risk factors for each ecological component identified at the highest levels of risk can be systematically worked through to match them with appropriate management and research actions (e.g. solution scanning Sutherland *et al.*, 2014). Studies like Sutherland *et al.* (2014) can be used as a source of ideas and broaden the perspective of managers and scientists to a wider range of possibilities than they would otherwise. However, the objective is not to develop an exhaustive list of possible tools. Rather it is to match appropriate tools to a specific risk factor instead of assuming one type of tool (e.g. spatial management) will address all issues (Halpern *et al.*, 2010).

Ecological risk factors provide guidance in both prioritizing and implementation of management and research actions. For example, an ecological risk factor for seagrass beds is the proportion of area occurring in water depths shallower than 2 m. Therefore, modifications to artificial seawalls to reduce their vertical slope, to reduce turbulence, could be prioritized for those walls with adjacent seagrass in water depth shallower than 2 m. The proportion of connectivity between habitat next to natural habitats is another ecological risk factor that could be used to guide implementation. For example, seagrass habitat next to natural stands of mangrove habitat have been shown to be more highly productive than seagrasses without such connectivity (Jelbart *et al.*, 2007; Meynecke *et al.*, 2008), which potentially increases their CTR to a HP. Consequently, management actions directed toward treating a single stressor for seagrass habitat with intact connectivity might be more effective than implementing management actions at multiple stressors in seagrass habitat with highly fragmented connectivity and in poor condition. Conversely, restoration of fragmented connectivity between natural habitats might be given a higher priority within a sub-catchment or estuary where there are few or no well-connected habitats.

Knowledge gap risk factors identify research actions that would improve the assessment of risk and elucidate the stressor and ecological risk factors contributing to risks of ecological components. The outcomes of these research actions reduce epistemic uncertainty (Hayes, 2011) and enable the development of effective management actions. In this way, research actions are ultimately linked to risk treatments. For example, in the Cowan sub-catchment of the Hawkesbury estuary, there were three interrelated key knowledge gap risk factors: effective total nitrogen loads from non-point source pollutants (Roper *et al.*, 2010), flushing time of bays, and recreational boating activity. Quantifying the magnitude, duration, frequency and spatial and temporal patterns of nitrogen loads from foreshore housing, and boating activity within the sub-catchment along with the flushing time of bays enables two things to be evaluated. First, whether nitrogen loads from these non-point sources exceeds the decision criteria in the risk characterization step. If it does not then the risk level for each ecological component can be refined. Second, if it does exceed the decision criteria the contribution of this nearshore source of increased nitrogen makes to the sub-catchment compared with whole catchment sources can be determined (Eyre and Pepperell, 1999). Filling these knowledge gaps would help determine whether targeting management actions to reduce total effective nitrogen loads at the sub-catchment scale would be more efficacious than targeting management actions only at the estuary wide scale (Eyre and Pepperell, 1999).

Identifying and resourcing management implementation groups

Once all risk factors for ecological components at high levels of risk have been matched with a range of potential management tools and

Table 5. Example of risk factors for seagrass habitat at high risk in the Pittwater sub-catchment.

Human activity	Risk factor (type)	Issues arising
1. Recreational fishing	Intensity (stressor) Proportion of habitat (ecological)	Large potential for interaction between recreational fishers and seagrass habitat from both boat and shore-based fishing. Annual recreational fishing from boats exceeded 50 h per hectare of shallow water area (<5 m) in Pittwater and the estimated proportion of seagrass habitat in these shallow areas was 30%. Annual recreational fishing from the shoreline exceeded 200 h per km of shoreline in Pittwater. Estimated proportion of seagrass habitat along the shoreline was 26%.
	Invasive species (stressor)	Known vector for the introduction of non-native invasive species in seagrass beds (West *et al.*, 2007). Potential for *Caulerpa taxifolia* (list pest species) to spread via fragments on anchors and from trailers if not properly cleaned.
2. Foreshore development	Artificial rock wall (stressor)	Large proportion of artificial rock walls are within 10 m of a seagrass bed. Change in hardness and slope of shore can increase the intensity and frequency of water turbulence around seagrass beds potentially destabilizing them.
	Depth of water of seagrass (ecological)	Seagrasses in shallower depths are more vulnerable to being affected by such increased turbulence.
	Wharves and jetties Depth of water of seagrass (ecological)	Large proportion of private and public wharves and jetties are within 10 m of seagrass (>58%). The level of potential stress will depend on the depth in which these seagrasses occupy. Wharves and jetties increase boat activity If surrounding seagrass are in shallow depths they may be stressed by such activity.
	Invasive species (stressor)	Foreshore developments can be a vector for non-native invasives by providing a substrate for attachment. The potential for some of these species to spread into seagrass habitats is increased by the proximity of foreshore developments to seagrass.
3. Stormwater and catchment run-off	Catchment run-off (stressor)	Large proportion of stormwater outlets (>30%) are within 10 m of a seagrass bed. Increased turbidity, water turbulence, and water quality could be having localized but cumulative effects on seagrass condition and bed stabilization. In addition, the proportion of stormwater catchment to the surface area of Pittwater exceeds 50% potentially affecting water quality and hence seagrass condition in the bay.
	Gross pollutants (stressor)	Effectiveness of removal of gross pollutants from stormwater is low (<50%). Gross pollutants may sink onto seagrass resulting in damage, epiphytic growth and smothering.
	Effective total nitrogen load (knowledge gap)	There are a substantial number of stormwater outlets that are in proximity to a number of estuarine habitats within Pittwater. Information on the total effective nitrogen loads from these outlets will enable better assessment of the risk to these habitats to nutrient enrichment from these outlets.
4. Commercial vessels	Proximity of vessels to habitat (stressor)	Frequency of ferry services that are within 10 m of seagrass habitats during their routes exceeds 8 times a day and potentially interacts with 10 different seagrass beds. Especially prevalent around Scotland Island where surrounding seagrasses have declined over the last 10 years and ferries dock at four different locations around the island. Frequency of interaction with ferries may cause increased turbulence and turbidity affecting growth of seagrass depending on their depth.
	Intensity (knowledge gap)	Water taxis are known to be used by both residents and visitors to the bay. Information on their routes with respect to habitats, particularly in shallow areas, the frequency of their use, and method of operation (e.g. drop-offs and pick-ups from beaches or wharves) would enable assessment of their potential level of interaction with estuarine habitats. There are also an unknown number of mooring contractors, rubbish barges, and maintenance vessels operating in Pittwater. Information on their number and where they operate in relation to habitats especially in shallow areas is needed.
5. Recreational boating (non-fishing)	Intensity (knowledge gap)	Recreational boating (non-fishing) is a major activity in Pittwater but there is little information on the number of boats participating in these activities, where they go and how many people they carry. Recreational boats are able to move virtually anywhere in the bay, depending on their size, and so can potentially interact with all types of estuarine habitats (Bell *et al.*, 2002). Information is needed on the magnitude of activity (e.g. number of boats, number of people per boat, number of hours of recreational activity that is boat-based), location and size of boats (smaller day boats compared to larger overnight vessels) participating in recreational activities. Such information should be collected to ensure differences in activity between seasons, week days and weekends, and school and non-school holiday periods can be assessed.
6. Dredging	Intensity (knowledge gap)	Dredging and foreshore development has occurred in many places in Pittwater particularly in its southern most sections. These activities result in changes to the bathymetry of the bay over time which can lead to erosion and/or accretion of sediments around subtidal habitats, potentially destabilizing them. Erosion can be seen along the foreshore at or above the waterline. The extent of any such erosion and/or sediment accretion subtidally is poorly known. Declines in habitat patches, such as seagrasses, over time may be partly caused by such subtidal sedimentation processes

Continued

Table 5. *Continued*

Human activity	Risk factor (type)	Issues arising
	Contaminated sediments (knowledge gap)	There are contaminated sediments in Pittwater (Lawson and Treloar, 2003). Information on the proportion of sediments contaminated and the distribution of these sediments with respect to other estuarine habitats (e.g. seagrass, mangroves, mudflats, and saltmarsh) would enable a better assessment of whether these habitats are at risk of being affected by these contaminated sediments.

Type of risk factor in brackets. See text for explanation.

Table 6. An example of potential management and research tools that could be applied to stressor risk factors of foreshore development on seagrass habitat.

Stressor risk factor	Potential management tools	Potential research tools
Proportion of wharves and jetties	Limit further development of new jetties	Beyond BACI monitoring programme to detect impacts of boat activity on the condition of seagrass beds within the sub-catchment
	Regulate boat activity around jetties and wharves with seagrass within 10 m of structure	Monitoring programme that measures the magnitude, frequency, and duration of boating activity around wharves and jetties within 10 m of seagrass beds in the sub-catchment
	Remove disused jetties	Monitoring programme to detect the introduction of non-native invasive species into seagrass beds via the vectors of wharves and jetties
Unsewered housing	Improve on-site sewage treatment	Beyond BACI monitoring programme to detect impacts of increased nutrients from ineffective sewage treatment on the condition of seagrass beds within the sub-catchment and improvement in condition as a result of management actions
	Limit further foreshore housing development within the sub-catchment	
	Increase frequency of on-site sewage collection from septic tanks	
Artificial rockwalls	Modify rockwalls to decrease slop and hardness	Beyond BACI monitoring programme to detect impacts of artificial rockwalls on the condition of seagrass beds within the sub-catchment
	Replace artificial rockwalls with environmentally friendly walls	Monitoring programmes to measure changes in the condition of seagrass beds within the catchment as a result of management actions
	Implement no wash zones for boat activity in areas with artificial rockwalls with adjacent seagrass habitat	Monitoring programme that measures the magnitude, frequency and duration of turbulence and suspended sediments around artificial rockwalls within 10 m of seagrass beds in the sub-catchment
	Removal of rockwalls and re-vegetate with natural habitat	Monitoring programme to detect the introduction of non-native invasive species into seagrass beds via the vector of artificial rockwalls
	Limit further development of artificial rockwalls within the sub-catchment	

research actions MNRM now has a platform for obtaining and allocating resources, allocating responsibilities, and engaging partnerships to implement them. Mechanisms such as cost–benefit analysis can be used for prioritizing resources within a single jurisdiction or human activity sector (but see Wegner and Pascual, 2011). However, many risk factors will require risk treatments that extend beyond the boundary of local government or sector responsibilities (Rosenberg and Sandifer, 2009). The risk factor-treatment platform provides a practical way these boundaries can be opened up as illustrated in Table 7. There are at least four ways this platform can be used.

First, it can be used to engage communities and stakeholders in discussion and negotiation of the risk factors needing to be addressed. It provides them with a concrete way of understanding what is at risk and why. It gives them the opportunity to contribute their own ideas for management actions to address the risk factors. Importantly, it provides a basis for negotiating which risk factors to address and when, given limited resources. For example, communities and stakeholders may prefer accepting constraints on one type of human stressor than another or be willing to accept the consequences of not addressing an issue to maintain the social and/or economic benefits from a human activity in an area, although ecologically this may be undesirable. The platform enables communities to weigh up the social, economic, and ecological costs of different management actions in a tangible way.

Second, the platform can be used to identify and negotiate with those government and non-government groups responsible for implementing particular management tools. For example, a local council responsible for a sub-catchment may identify that moorings within seagrass habitats are a risk factor for their sustainability and should be replaced with less damaging types or removed (Demers *et al.*, 2013). However, responsibility for moorings is a state government agency. Bringing local and state government agencies together around the risk factor-treatment platform enables these groups to understand and discuss how different management tools address the range of risk factors and work together to implement risk treatments that lowers the risk of not achieving the MEBM objectives for an area.

Third, the platform can be used to address multiple and cumulative risk factors and design and coordinate risk treatments across jurisdictions, sectors, and communities. For example, in the Pittwater sub-catchment of the Hawkesbury estuary, seagrass habitats were at high levels of risk from four different human activities governed by two levels of government and involving a range of different community groups. To effectively reduce the risk level to this habitat may require risk treatments for the stressor risk

Table 7. An example of a hypothetical risk factor treatment linkage platform of foreshore development for seagrass habitat for a sub-catchment of an estuary.

		Risk factors (stressor)		
Management group	**Management tools**	**Proportion of wharves and jetties**	**Unsewered housing**	**Artificial rockwalls**
Maritime	Seawall construction			Modify rockwalls to decrease slop and hardness; Replace artificial rockwalls with environmentally friendly walls
	No wash zones	Instigate no wash zones for boat activity around jetties and wharves close to seagrass		Implement no wash zones for boat activity in areas with artificial rockwalls close to seagrass
Local council	Planning laws	Limit further development of new jetties	Limit further foreshore housing development within the sub-catchment	Limit further development of artificial rockwalls within the sub-catchment
	Sewage collection		Improve on-site sewage treatment; Increase frequency of on-site sewage collection from septic tanks	
Community groups	Education	Education programme on effects of boating activity on seagrass ecology		Education programme on benefits of environmentally friendly seawalls
	Clean-up campaigns	Remove disused jetties		
	Bush care			Removal of rockwalls and re-vegetate with natural habit
Research provider	**Research tools**			
Government agency	Monitoring and manipulative experiments	Beyond BACI monitoring programme to detect impacts of boat activity on the condition of seagrass beds within the sub-catchment; Monitoring programme to detect the introduction of non-native invasive species into seagrass beds via the vectors of wharves and jetties	Beyond BACI monitoring programme to detect impacts of increased nutrients from ineffective sewage treatment on the condition of seagrass beds within the sub-catchment and improvement in condition as a result of management actions	Monitoring programmes to measure changes in the condition of seagrass beds within the sub-catchment as a result of management actions
	Manipulative experiments	Test hypotheses to determine casual relationships between seagrass condition and disturbances due wharves, jetties unsewered foreshore housing, and artificial seawalls		
University	Monitoring and manipulative experiments	Monitoring programme that measures the magnitude, frequency, and duration of boating activity around wharves and jetties within 10 m of seagrass beds in the sub-catchment		Beyond BACI monitoring programme to detect impacts of artificial rockwalls on the condition of seagrass beds within the sub-catchment
Private consultants	Monitoring			Monitoring programme that measures the magnitude, frequency and duration of turbulence and suspended sediments around artificial rockwalls within 10 m of seagrass beds in the sub-catchment.
Citizen science	Underwater diver surveys	Monitoring condition of seagrass habitats and associated fish assemblages		

factors from all four human activities. Potential cumulative interacting risk factors can also be evaluated (Crain *et al*, 2008) and common pressure pathways identified (Knights *et al*., 2013). This can lead to developing research actions and management tools that more effectively assesses and addresses the risks.

Fourth, by linking knowledge gap risk factors with potential research actions, the platform can used to evaluate the consequences of not filling certain knowledge gaps compared with filling others, given limited resources. From a practical perspective it is unlikely that all knowledge gaps can be filled. The platform provides a concrete means to engage all sectors, jurisdictions, community and stakeholder groups, and research providers in discussing which knowledge gaps, if they were filled, could bring the greatest benefit in managing risks and achieving MEBM objectives. Once this has been determined the platform can be used to identify and engage appropriate research providers to work collaboratively and interdisciplinarily to address multiple facets of knowledge gaps to fill them. The platform can then be used to justify funding sought to resource those research actions.

Monitoring, assessing, and learning from the outcomes of management actions

An integral part of an adaptive management framework is learning from the outcomes of management actions (Smith *et al*., 2009). This learning can only occur through monitoring and assessing the effectiveness of management implementation. Proposed management actions lead to testable hypotheses which enable research providers to work with management, stakeholders, and communities to design and assess management responses effectively (Underwood, 1995). Risk treatments should result in changes to the stressor risk factors and corresponding changes in the condition (structural and/or functional) of ecological components (Allan *et al*., 2013). Therefore, clear predictions of what should change both in the stressors and the ecological components are determined by the management actions. Research actions are then designed to focus on monitoring that detects changes in these stressors and characteristics of the ecological components, at appropriate spatial and temporal scales. Determining what to monitor on this basis results in developing measures (i.e. indicators) that are relevant, have an expected response to management action, and are measurable and interpretable (Rochet and Trenkel, 2003).

Assessment evaluates the outcomes of management actions in terms of MEBM objectives for the ecosystem in focus. Have the outcomes lowered the risk of not achieving the management objectives for a marine ecosystem to an acceptable level? If so, what has worked, why and how can this be sustained through improved policies and management? If it has not lowered the risk, what has been learned about the relationship between risk factors and the structure and function of marine biodiversity components and about the design and implementation of risk treatments? (Underwood, 1997; Smith *et al*., 2009). Therefore, assessment using the linkage between risk factors and risk treatments can track, in tangible ways, what has improved and what has not in achieving MEBM at regional and small spatial scales.

Discussion

Recently, there have been calls for practical ways of implementing MEBM objectives, particularly at smaller spatial scales (Cook *et al*., 2013; Game *et al*., 2013). As part of the solution to this, greater attention in marine ERAs has been given to identifying specific characteristics or attributes that contribute to risks (e.g. Sethi, 2010; Cormier *et al*., 2013). For example, Samhouri and Levin (2012) used spatial and temporal management factors to evaluate exposure and resistance and recovery factors to evaluate sensitivity in their ERA of a coastal ecosystem. They then propose these factors could be used to explore "*how* human activities *influence* risk to ecosystem properties" (emphasis added). The method described above takes this idea a step further and identifies different types of risk factors that can then be used to direct specific management and research actions to help achieve MEBM objectives.

There are three key features of the risk factor-treatment linkage that makes ERA for marine biodiversity more efficacious than simply using it to prioritize issues (e.g. Levin *et al*., 2009). First, it provides a scientifically based and transparent process to engage all actors who need to be involved in addressing the issues raised by an ERA, including researchers, managers, stakeholders, communities, and government advisors. One of the impediments to implementing MNRM to achieve MEBM objectives is the lack of consensus and ownership of what human activities need to be managed and why (e.g. Griffin, 2009). This can result in poor compliance to some management actions (e.g. Kritzer, 2004). Furthermore, grasping the complexity of marine ecosystems can be significantly challenging for different actors (de Jonge *et al*., 2012). To meet these challenges, the risk factor-treatment linkage engages all actors by helping explain what ecological components are at high risk and why and breaking down the complexity of human–ecosystem interactions into manageable parts. Consequently, it provides a basis for more open and honest discussions among all actors about priorities, preferences and the effects of trade-offs on achieving MEBM objectives of addressing some issues and not others given limited resources. Thus, it is a mechanism for determining where limited resources can be best invested to maximize the achievement of management objectives for marine biodiversity within a local context.

Second, it provides a means by which management and research actions can be integrated. The risk factor-treatment linkage means that research and management can be focused on the same issues such that management actions are coupled or underpinned with research. Research supports management actions by (i) helping design interventions so that their effects can be measured and detected, (ii) monitoring the effectiveness of management actions to provide insights for improvement and track progress, and (iii) filling knowledge gap risk factors which improve the assessments of risks and understanding of ecoystems. Such integration has been achieved in some single sector MEBM approaches, such as fisheries (e.g. Fletcher *et al*., 2012). But the method described in this paper provides a tangible way integration between management and research could be achieved in a multi-sector complex marine ecosystem.

Third, it provides a more comprehensive and complete assessment of the risks to ecological components (Hayes, 2011). Listing all stressors exerted by human activities revealed interactions where potential impacts could occur that would have been missed if only one stressor was used. For example, including bait collection as a stressor of recreational fishing identified the potential risk to intertidal mudflats and rocky reefs in sub-catchments of the Hawkesbury estuary. This would have been missed if only the intensity of recreational fishing (number of angler hours) was used. Furthermore, management actions to address the intensity of recreational fishing and not also bait collection may not adequately conserve all components of marine biodiversity in the estuary because bait collection can occur independently of active fishing effort (Wynberg and Branch, 1997; Lewin *et al*., 2006). Similarly, listing

all factors that contribute to the CTR of ecological components reveals different aspects that may be impacted by different stressors. For example, including characteristics for above and below ground processes for mangrove habitats indicates their vulnerability to catchment impacts (such as changes in sediment inputs) and to groundwater impacts (such as increased drawn downs during drought conditions) both of which could negatively affect their CTR to sea level rise via surface elevation maintenance (Rogers *et al.*, 2006). Such comprehensive lists of stressors and characteristics are necessary when undertaking an ERA of marine biodiversity in the context of multiple human activities to provide a more complete analysis (Hayes, 2011; Aven, 2012).

Making connections between contributors to risk and ecosystem states has been investigated by Cook *et al.* (2013). They developed a risk assessment model of the linkage between pressures, states, and ecosystem services for a regional coastal ecosystem. From this they identified the relative impacts of different ecosystem pressures on multiple ecosystem services, such as changed freshwater delivery on existence of natural systems. Similarly, Hayes and Landis (2004) used a regional risk assessment on an estuarine system that identified major contributors to risk to the ecosystem of vessel traffic, upland urban, and agricultural land use and shoreline recreational activities. However, neither study drilled down to identify the specific stressors that these HPs exert on ecological components. MNRM and research at regional and small spatial scales can usually only have an effect on stressors not the pressures. For example, increases in recreational boating and foreshore activities are unlikely to be stopped at these spatial scales but the stressors from such activities at particular times and places can be influenced to reduce their potential impacts, such as the number and placement of boat ramps and jetties. The distinctive feature of the risk factor-treatment linkage of this paper is that risk factors are identified at this finer scale giving management and research greater leverage in addressing issues.

Three challenges of the ERA method for linking risk factors to risk treatments need to be addressed in the future. First, comprehensive lists of stressors and ecological characters have the potential for generating false-positive and false-negative ERA results. False positives identify high risk when it is actually low. Including multiple stressors could over inflate the measure of pressure being exerted on an ecological component and hence increase the perceived level of risk. This could result in the investment of resources to issues where it is not needed. False negatives identify low risk when it is actually high. Including many ecological characters of a component could assess it as having a greater CTR than it actually has, underestimating the risk level, a potentially more serious problem. The ERA method was designed to be bias towards detecting false positives in two ways. The precautionary principle was applied by assuming there will be an interaction between a stressor and an ecological component in the absence of contrary information. Then conservative estimates were used in the decision criteria that were biased towards detecting a contribution to a HP or CTR. These biases have been applied in other ERA methods for fisheries (e.g. Hobday *et al.*, 2011). Addressing the challenge of false positives and negatives in the future will require undertaking sensitivity analyses that varies the number of stressors and characteristics used to assess risk levels and the rates of false-positive and -negative results generated.

Second, the extent to which the relationship between stressors, stress measures, and outcomes is correlative or causal is unknown for many human activities. Likewise, the nature of the relationships between characteristics, measures, and contribution to CTR is unknown for many ecological components. This has implications for developing effective management actions that address stressor risk factors. If stressor risk factors are correlative, then they may not respond to management actions or produce unexpected outcomes. This would become evident in well-designed management, monitoring, and research action that tested hypotheses about these relationships and is part of the learning process. But future research should also aim to test some of these relationships in advance to provide more robust information on linkages.

Third, incorporating epistemic uncertainty and weighting into the measures of stressors and characteristics needs to be developed. Epistemic uncertainty (Hayes, 2011) in the ERA of marine biodiversity in the Hawkesbury occurred in at least five places: (i) in models of the relationships between human activities, their stressors and potential outcomes and models of the relationships between ecological components, functions, characteristics and outcomes; (ii) decision criteria used; (iii) choice of measures used for each stressor and ecological characteristic; (iv) the values of the measures of each stressor and characteristic; (v) unknown interactions between stressors, ecological components, or multiple HPs. These uncertainties can contribute to generation of false negatives and positives. By applying the precautionary principle and using conservative values, however, the method has erred on the side of false positives rather than false negatives.

Incorporating these areas of uncertainty in future analyses would require some or all the following:

(i) *Model uncertainty*: all relationships were based on those documented in the scientific literature and were assumed to be realistic. However, even published relationships can turn out to be incorrect or not be very strong. Running sensitivity tests on these assumptions would assess how the level of HP would change if these relationships were false or weak. In addition, measures of stressor or characteristics could be multiplied by the strength of the relationships reported in published studies to account for model uncertainty.

(ii) *Uncertainty in the decision criteria used*: This was addressed by using conservative estimates that were biased toward detecting a contribution to a HP or CTR.

(iii) *Uncertainty in the choice of measures used for each stressor and ecological characteristic*: This was due to some measures being used that were indirect rather direct measures. This uncertainty can be incorporated by applying an error term to the total pressure or CTR for the proportion of measures that were indirect. Error terms could be derived using direct measures of stressors or characteristics for which data are available then substituting these with indirect measures and evaluate to what extent it changes the level of risk.

Uncertainty in the choice of measures may also be due to the combination of measures used. Some stressors or characteristics may contribute to HP or CTR, respectively, more than others (e.g. Suter and Cormier, 2011). For example, shoot density of a seagrass bed may be a more important contributor to its CTR to stressors than areal extent of the bed (e.g. Worm and Reusch, 2000). When it is known that particular stressors or characteristics do contribute more than others a weighting can be applied to such measures. However, such weighting needs be justified with adequate, independent empirical evidence (Linkov *et al.*, 2009). In the Hawkesbury estuary, there was not sufficient independent information to

determine levels of weighting to measures and therefore, all measures were considered as having equal weight. The effect of this is that the level of risk might be under or overestimated.

(iv) *Uncertainty in the values of the measures of each stressor and characteristic*: This uncertainty can be addressed by applying an error term to each value from the study from which it was derived (e.g. standard error in the abundance of seagrass) if it is available. For the Hawkesbury estuary, no such error terms were available and deriving a qualitative level of uncertainty based on expert judgment was not considered robust. Therefore, it was assumed that all measures had the same level of error. Again the effect of this is that the level of risk might be under or overestimated. Future applications of the method should run sensitivity tests on the range of errors for each measure and the effect on the levels of HP and CTR. The results can then be included in the advice to managers about the level of risks.

(v) *Uncertainty about interactions among stressors, characteristics and HPs*: The method for capturing unknowns for measures with no information could be extended to identify and extract unknowns in potential interactions based on a literature review. Sensitivity tests could then be done to assess the effect of assuming strong or weak levels of interactions.

Despite these challenges, linking risk factors to risk treatment in ERA for marine biodiversity are a promising mechanism. The method described here provides a tangible way management and research can address specific issues using the different types of risk factors. The systematic approach enables the dual complexities of marine ecosystems and multiple HPs to be broken down to identify and target issues effectively. The risk factor-treatment linkage provides a platform to negotiate and develop effective management and research actions across jurisdictional, disciplinary and community and stakeholder boundaries. Using this mechanism could provide a practical means to achieve MEBM objectives at regional and small spatial scales.

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References

Allan, J. D., McIntyre, P. B., Smith, S. D. P., Halpern, B. S., Boyer, G. L., Buchsbaum, A., Burton, G. A., *et al.* 2013. Joint analysis of stressors and ecosystem services to enhance restoration effectiveness. Proceedings of the National Academy of Sciences, 110: 372–377.

Astles, K. L. 2008. A systematic approach to estimating ecological risk in marine fisheries. CAB Reviews, 3: 16.

Astles, K. L. 2010. Qualitative ecological risk assessment of human disturbances on estuarine habitats. *In* Estuarine habitat mapping and geomorphic characterisation of the lower Hawkesbury River and Pittwater estuaries, Fisheries Final Report Series No. 117, pp. 33–104. Ed. by K. L. Astles, G. West, and R. Creese.

Astles, K. L. 2012. Qualitative ecological risk assessment in fisheries—challenges for scientists and managers. *In* Advances in Environmental Research, 6, pp. 222–251. Ed. by J. A. Daniels. Nova Science Publishers, New York.

Astles, K. L., Gibbs, P. J., Steffe, A. S., and Green, M. 2009. A qualitative risk-based assessment of impacts on marine habitats and harvested species for a data deficient wild capture fishery. Biological Conservation, 142: 2759–2773.

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. Fisheries Research, 82: 290–303.

Aven, T. 2012. Foundational issues in risk assessment and risk management. Risk Analysis, 32: 1647–1656.

Beger, M., Grantham, H. S., Pressey, R. L., Wilson, K. A., Peterson, E. L., Dorfman, D., Mumby, P. J., *et al.* 2010. Conservation planning for connectivity across marine, freshwater, and terrestrial realms. Biological Conservation, 143: 565–575.

Bell, S. S., Hall, M. O., Soffian, S., and Madley, K. 2002. Assessing impacts of boat propeller scars on shrimp and fish utilization of seagrass beds. Ecological Applications, 12: 206–217.

Boon, P. I., White, M., and Sinclair, S. 2010. Climate change impacts on Victoria's coastal vegetation (mangroves and saltmarsh): Western port case study. Climate Change 2010: Practical Responses to Climate Change. 21.

Boys, C. A., Kroon, F. J., Glasby, T. M., and Wilkinson, K. 2012. Improved fish and crustacean passage in tidal creeks following floodgate remediation. Journal of Applied Ecology, 49: 223–233.

Broadhurst, M. K., Kennelly, S. J., Watson, J. W., and Workman, I. K. 1997. Evaluations of the Nordmore grid and secondary bycatch-reducing devices (BRD's) in the Hunter River prawn-trawl fishery, Australia. Fishery Bulletin, 95: 209–218.

Brown, T. M., Kuzyk, Z. Z. A., Stow, J. P., Burgess, N. M., Solomon, S. M., Sheldon, T. A., and Reimer, K. J. 2013. Effects-based marine ecological risk assessment at a polychlorinated biphenyl-contaminated site in Saglek, Labrador, Canada. Environmental Toxicology and Chemistry, 32: 453–467.

Bulleri, F. 2005. The introduction of artificial structures on marine soft-and hard-bottoms: ecological implications of epibiota. Environmental Conservation, 32: 101–102.

Clarke, B., Stocker, L., Coffey, B., Leith, P., Harvey, N., Baldwin, C., Baxter, T., *et al.* 2013. Enhancing the knowledge–governance interface: coasts, climate and collaboration. Ocean and Coastal Management, 86: 88–99.

Clarke, P. J., Kerrigan, R. A., and Westphal, C. J. 2001. Dispersal potential and early growth in 14 tropical mangroves: do early life history traits correlate with patterns of adult distribution? Journal of Ecology, 89: 648–659.

Clynick, B. G. 2008. Characteristics of an urban fish assemblage: distribution of fish associated with coastal marinas. Marine Environmental Research, 65: 18–33.

Clynick, B. G., Blockley, D., and Chapman, M. G. 2009. Anthropogenic changes in patterns of diversity on hard substrata: an overview. *In* Marine Hard Bottom Communities, 206, pp. 247–256. Ed. by M. Wahl. Springer-Verlag, Berlin.

Cook, G. S., Fletcher, P. J., and Kelble, C. R. 2013. Towards marine ecosystem based management in South Florida: investigating the connections among ecosystem pressures, states and services in a complex coastal system. Ecological Indicators, doi: 10.1016/j.ecolind.2013.10.026.

Cormier, R., Kannen, A., Elliot, M., Hall, P., and Davies, I. M. 2013. Marine and coastal ecosystem-based risk management handbook. ICES Cooperative Research Report No. 317. 60 pp.

Cox, L. A. 2008. What's wrong with risk matrices? Risk Analysis, 28: 497–512.

Crain, C. M., Kroeker, K., and Halpern, B. S. 2008. Interactive and cumulative effects of multiple human stressors in marine systems. Ecology Letters, 11: 1304–1315.

Dayton, P. K., Thrush, S. F., Agardy, M. T., and Hofman, R. J. 1995. Environmental effects of marine fishing. Aquatic Conservation: Marine and Freshwater Ecosystems, 5: 205–532.

de Jonge, V. N., Pinto, R., and Turner, R. K. 2012. Integrating ecological, economic and social aspects to generate useful management information under EU Directives' "ecosystem approach". Ocean and Coastal Management, 68: 169–188.

Demers, M. A., Davis, A. R., and Knott, N. A. 2013. A comparison of the impact of "seagrass-friendly" boat mooring systems on *Posidonia australis*. Marine Environmental Research, 83: 54–62.

Di Franco, A., Graziano, M., Franzitta, G., Felline, S., Chemello, R., and Milazzo, M. 2011. Do small marinas drive habitat specific impacts? A case study from Mediterranean Sea. Marine Pollution Bulletin, 62: 926–933.

Drescher, M., Perera, A. H., Johnson, C. J., Buse, L. J., Drew, C. A., and Burgman, M. A. 2013. Toward rigorous use of expert knowledge in ecological research. Ecosphere, 4: art83.

Eyre, B. D., and Pepperell, P. 1999. A spatially intensive approach to water quality monitoring in the Rous River catchment, NSW, Australia. Journal of Environmental Management, 56: 97–118.

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

Fletcher, W. J., Gaughan, D. J., Metcalf, S. J., and Shaw, J. 2012. Using a regional level, risk-based framework to cost effectively implement ecosystem-based fisheries management. *In* Global progress in ecosystem based fisheries management, pp. 129–146. Ed. by G. H. Kruse, H. I. Browman, K. L. Cochrane, D. Evans, G. S. Jamieson, P. A. Livingston, D. Woodby, and C. I. Zhang. Alaska Sea Grant, University of Alaska Fairbanks.

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

Gilman, E. L., Ellison, J., Duke, N. C., and Field, C. 2008. Threats to mangroves from climate change and adaptation options: a review. Aquatic Botany, 89: 237–250.

Glasby, T. M., Connell, S. D., Holloway, M. G., and Hewitt, C. L. 2007. Non-indigenous biota on artificial structures: could habitat creation facilitate biological invasions? Marine Biology, 151: 887–895.

Glasby, T. M., and Underwood, A. J. 1996. Sampling to differentiate between pulse and press perturbations. Environmental Monitoring and Assessment, 42: 241–252.

Goble, R., and Bier, V. M. 2013. Risk assessment can be a game-changing information technology—but too often it isn't. Risk Analysis, 33: 1942–1951.

Griffin, L. 2009. Scales of knowledge: North Sea fisheries governance, the local fisherman and the European scientist. Environmental Politics, 18: 557–575.

Haines, P., Fletcher, M., Rollason, V., and Coad, P. 2008. Lower Hawkesbury Estuary Management Plan. BMT WBM Pty Ltd, Sydney.

Hallac, D. E., Sadle, J., Pearlstine, L., Herling, F., and Shinde, D. 2012. Boating impacts to seagrass in Florida Bay, Everglades National Park, Florida, USA: links with physical and visitor-use factors and implications for management. Marine and Freshwater Research, 63: 1117–1128.

Halpern, B. S., Kappel, C. V., Selkoe, K. A., Micheli, F., Ebert, C. M., Kontgis, C., and Crain, C. M., *et al.* 2009. Mapping cumulative human impacts to California current marine ecosystems. Conservation Letters, 2: 138–148.

Halpern, B. S., Lester, S. E., and McLeod, K. L. 2010. Placing marine protected areas onto the ecosystem-based management seascape. Proceedings of the National Academy of Sciences, 107: 18312–18317.

Halpern, B. S., Selkoe, K. A., Micheli, F., and Kappel, C. V. 2007. Evaluating and ranking the vulnerability of global marine ecosystems to anthropogenic threats. Conservation Biology, 21: 1301–1315.

Hayes, E. H., and Landis, W. G. 2004. Regional ecological risk assessment of a near shore marine environment: Cherry Point, WA. Human and Ecological Risk Assessment, 10: 299–325.

Hayes, K. R. 2011. Uncertainty and uncertainty analysis methods. Australian Centre for Excellence in Risk Analysis, Project 0705, Report EP102467.

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.

Hughes, T. P., and Connell, J. H. 1999. Multiple stressors on coral reefs: a long-term perspective. Limnology and Oceanography, 44: 932–940.

Jelbart, J. E., Ross, P. M., and Connolly, R. M. 2007. Fish assemblages in seagrass beds are influenced by the proximity of mangrove forests. Marine Biology, 150: 993–1002.

Kazdin, A. E. 2007. Mediators and mechanisms of change in psychotherapy research. Annual Review Clinical Psychology, 3: 1–27.

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.

Knott, N. A., Aulbury, J. P., Brown, T. H., and Johnston, E. L. 2009. Contemporary ecological threats from historical pollution sources: impacts of large-scale resuspension of contaminated sediments on sessile invertebrate recruitment. Journal of Applied Ecology, 46: 770–781.

Koehn, J. D., Alistair, J., Hobday, A. J., Morgan, S., Pratchett, M. S., and Gillanders, B. M. 2011. Climate change and Australian marine and freshwater environments, fishes and fisheries: synthesis and options for adaptation. Marine and Freshwater Research, 62: 1148–1164.

Kraemer, H. C., Kazdin, A., Offord, D., Kessler, R. C., Jensen, P. S., and Kupfer, D. 1997. Coming to terms with the terms of risk. Archives of General Psychiatry, 54: 337–343.

Kraemer, H. C., Stice, E., Kazdin, A., Offord, D., and Kupfer, D. 2001. How do risk factors work together? Mediators, moderators and independent, overlapping and proxy risk factors. American Journal of Psychiatry, 158: 848–856.

Kritzer, J. P. 2004. Effects of Noncompliance on the success of alternative designs of marine protected-area networks for conservation and fisheries management. Conservation Biology, 18: 1021–1031.

Lawson and Trelor Pty Ltd. 2003. Pittwater Estuary Processes Study. Lawson and Treloar Pty Ltd, Sydney, Australia. 150 pp.

Lazarow, N., Souter, R., Fearon, R., and Dovers, S. 2006. Coastal management in Australia: Key institutional and governance issues for coastal natural resource management and planning. p. 136. Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management (Coastal CRC) Indooroopilly Sciences Centre.

Levin, P. S., Fogarty, M. J., Murawski, S. A., and Fluharty, D. 2009. Integrated ecosystem assessments: developing the scientific basis for ecosystem-based management of the ocean. PLOS Biology, 7: e1000014.

Lewin, W. C., Arlinghaus, R., and Mehner, T. 2006. Documented and potential biological impacts of recreational fishing: insights for management and conservation. Reviews in Fisheries Science, 14: 305–367.

Linkov, I., Loney, D., Cormier, S., Satterstrom, F. K., and Bridges, T. 2009. Weight-of-evidence evaluation in environmental assessment: review of qualitative and quantitative approaches. Science of the Total Environment, 407: 5199–5205.

McNie, E. C. 2007. Reconciling the supply of scientific information with user demands: an analysis of the problem and review of the literature. Environmental Science and Policy, 10: 17–38.

Melville, F., and Burchett, M. 2002. Genetic variation in *Avicennia marina* in three estuaries of Sydney (Australia) and implications for rehabilitation and management. Marine Pollution Bulletin, 44: 469–479.

Melville, F., Burchett, M., and Pulkownik, A. 2004. Genetic variation among age-classes of the mangrove *Avicennia marina* in clean and contaminated sediments. Marine Pollution Bulletin, 49: 695–703.

Meynecke, J. O., Lee, S. Y., and Duke, N. C. 2008. Linking spatial metrics and fish catch reveals the importance of coastal wetland connectivity to inshore fisheries in Queensland, Australia. Biological Conservation, 141: 981–996.

Minchinton, T. E. 2007. Natural disturbance and regeneration of marine benthic communities. *In* Marine Ecology, pp. 138–174. Ed. by S. D. Connell, and B. M. Gillanders. Oxford University Press, South Melbourne, Australia.

New South Wales Government. 2010. Groundwater, Technical report series of the NSW Monitoring, Evaluation and Reporting Program. NSW Office of Water, Sydney, NSW.

Pitcher, C. R. 2014. Quantitative indicators of environmental sustainability risk for a tropical shelf trawl fishery. Fisheries Research, 151: 136–147.

Regan, H. M., Colyvan, M., and Burgman, M. A. 2002. A taxomony and treatment of uncertainty for ecology and conservation biology. Ecological Applications, 12: 618–628.

Rochet, M., and Trenkel, V. M. 2003. Which community indicators can measure the impact of fishing? A review and proposals. Canadian Journal of Fisheries and Aquatic Sciences, 60: 86–99.

Rogers, K., Wilton, K. M., and Saintilan, N. 2006. Vegetation change and surface elevation dynamics in estuarine wetlands of southeast Australia. Estuarine, Coastal and Shelf Science, 66: 559–569.

Roper, A., Creese, R., Scanes, P., Stephens, K., Williams, R., Dela-Cruz, J., and Coade, G., *et al.* 2010. NSW Estuaries – understanding condition and influencing management. Department of Environment, Climate Change and Water, NSW, Sydney.

Rosenberg, A. A., and Sandifer, P. A. 2009. What do managers need. Ecosystem-based management for the oceans. Island Press, Washington, DC. 13–30 p.

Ruiz, G. M., Carlton, J. T., Grosholz, E. D., and Hines, A. H. 1997. Global invasions of marine and estuarine habitats by non-indigenous species: mechanisms, extent, and consequences. American Zoologist, 37: 621–632.

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

Santos, C. D., Miranda, A. C., Granadeiro, J. P., Lourenço, P. M., Saraiva, S., and Palmeirim, J. M. 2010. Effects of artificial illumination on the nocturnal foraging of waders. Acta Oecologica, 36: 166–172.

Scanes, P., Coade, G., Doherty, M., and Hill, R. 2007. Evaluation of the utility of water quality based indicators of estuarine lagoon condition in NSW, Australia. Estuarine, Coastal and Shelf Science, 74: 306–319.

Sethi, S. A. 2010. Risk management for fisheries. Fish and Fisheries, 11: 341–365.

Smith, T. F., Carter, R. W., Thomsen, D. C., Mayes, G., Nursey-Bray, M., Whisson, G., Jones, R., *et al.* 2009. Enhancing science impact in the coastal zone through adaptive learning. Journal of Coastal Research, SI 56: 1306–1310.

Standards Australia. 2009. AS/NZS ISO 31000:2009 Risk management—principles and guidelines. Joint Technical Committee OB-007, Risk Management. Councils of Standards Australia and Standards New Zealand.

Stocker, L., Kennedy, D., Kenchington, R., and Merrick, K. 2012. Sustainable coastal management. *In* Sustainable Coastal Management and Climate Adaptation, Global Lessons from Regional Approaches in Australia, pp. 29–56. Ed. by R. A. Kenchington, L. Stocker, and D. Wood. Boca Raton, FL: CRC Press.

Suter, G. W., II, Barnthouse, L. W., and O'Neill, R. V. 1987. Treatment of risk in environmental impact assessment. Environmental Management, 11: 295–303.

Suter, G. W., II, and Cormier, S. M. 2011. Why and how to combine evidence in environmental assessments: weighing evidence and building cases. Science of the Total Environment, 409: 1406–1417.

Sutherland, W. J., Gardner, T., Bogich, T. L., Bradbury, R. B., Clothier, B., Jonsson, M., Kapos, V., *et al.* 2014. Solution scanning as a key policy tool: identifying management interventions to help maintain and enhance regulating ecosystem services. Ecology and Society, 19: 3.

Underwood, A. J. 1989. The analysis of stress in natural populations. Biological Journal of the Linnean Society, 37: 51–78.

Underwood, A. J. 1995. Ecological research and (and research into) environmental management. Ecological Applications, 5: 232–247.

Underwood, A. J. 1997. Experiments in ecology—their logical design and interpretation using analysis of variance. Cambridge University Press, Cambridge. 504 pp.

Vinebrooke, R. D., Cottingham, K. L., Scheffer, M., Dodson, S. I., Maberly, S. C., and Sommer, U. 2004. Impacts of multiple stressors on biodiversity and ecosystem functioning: role of species co-tolerance. Oikos, 104: 451–457.

Voyer, M., Gladstone, W., and Goodall, H. 2012. Methods of social assessment in marine protected area planning: Is public participation enough? Marine Policy, 36: 432–439.

Wegner, G., and Pascual, U. 2011. Cost-benefit analysis in the context of ecosystem services for human well-being: A multidisciplinary critique. Global Environmental Change, 21: 492–504.

West, E. J., Barnes, P. B., Wright, J. T., and Davis, A. R. 2007. Anchors aweigh: fragment generation of invasive *Caulerpa taxifolia* by boat anchors and its resistance to desiccation. Aquatic Botany, 87: 196–202.

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.

Williams, R. J., and Thiebaud, I. 2007. An analysis of changes to aquatic habitats and adjacent land-use in the downstream portion of the Hawkesbury Nepean River over the past sixty years. NSW Department of Primary Industries, Fisheries Final Report Series No. 91. 97 pp.

Wilson, K. A., Underwood, E. C., Morrison, S. A., Klausmeyer, K. R., Murdoch, W. W., Reyers, B., Wardell-Johnson, G., *et al.* 2007. Conserving biodiversity efficiently: what to do, where, and when. PLOS Biology, 5: e223.

Worm, B., and Reusch, T. B. 2000. Do nutrient availability and plant density limit seagrass colonization in the Baltic Sea? Marine Ecology Progress Series, 200: 159–166.

Wynberg, R. P., and Branch, G. M. 1997. Trampling associated with bait-collection for sandprawns *Callaianassa kraussi* Stebbing: effects on the biota of intertidal sandflat. Environmental Conservation, 24: 139–148.

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