ICES Journal of Marine Science

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

ICES Journal of Marine Science (2015), 72(3), 1116–1132. doi:10.1093/icesjms/fsu207

Contribution to the Themed Section: 'Risk Assessment'

Food for Thought

Linking risk factors to risk treatment in ecological risk assessment of marine biodiversity

K. L. Astles*

Department of Primary Industries, Fisheries NSW, PO Box 5106, Wollongong 2520, Australia

**Corresponding author: tel: +61 2 4222 8344; fax: +61 2 4222 8338; e-mail: karen.astles@dpi.nsw.gov.au*

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

Received 25 June 2014; revised 13 October 2014; accepted 26 October 2014; advance access publication 21 November 2014.

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

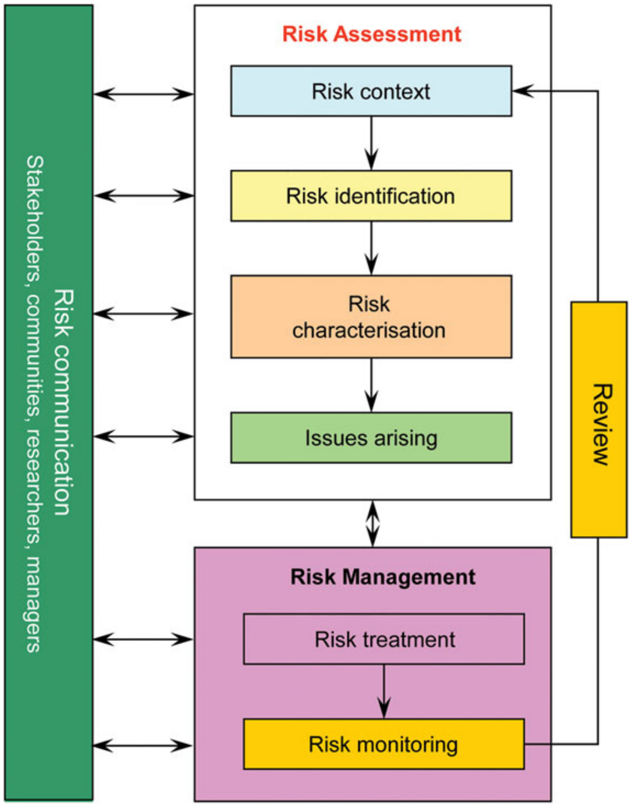

Figure 1. Framework for an ecological risk analysis of marine biodiversity consistent with AS/NZS ISO 31000:2009. Standards Australia (2009).

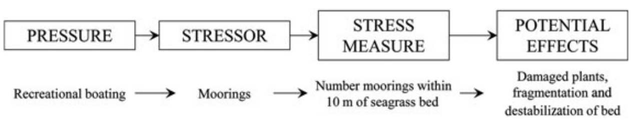

Figure 2. Relationship between pressure, stressor, stress measure and potential outcome and an example for recreational boating on seagrass habitat.

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.

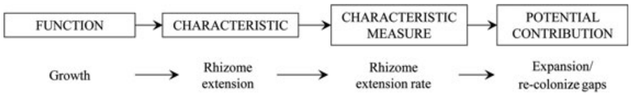

Figure 3. Relationship between a function, characteristic, its measure and potential contribution to a component's response to natural disturbance and an example for seagrass.

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.

Stressor	Stress measure	Decision criteria	Seagrass		Mangroves		Mudflats	
			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

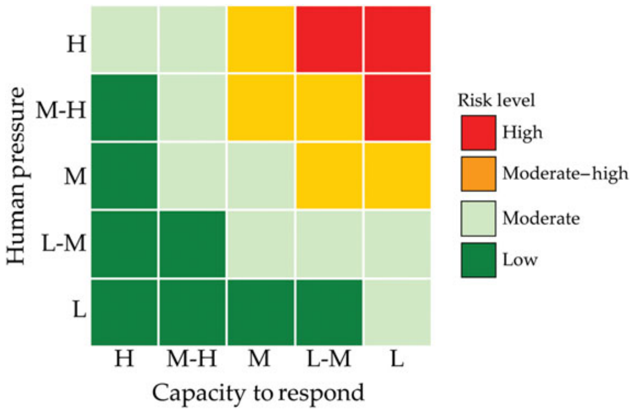

Figure 4. Risk matrix used to determine level of risk. H, high; H-M, high moderate; M, moderate; L-H, low moderate; L—low.

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 Limit further foreshore housing development within the sub-catchment Increase frequency of on-site sewage collection from septic tanks	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
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.

Acknowledgements

I thank Dr Peter Coad, Natural Resources Strategy Manager of Hornsby Shire Council, New South Wales (NSW), for his support, valued input, and opportunity to undertake the research in the Hawkesbury Estuary. I thank Greg West and Trudy Walford, GIS analysts of Fisheries NSW, Department of Primary Industries NSW, and Justin McKinnon for their expert work in providing habitat and stressor data. I thank Dr Aldo Steffe and Jeff Murphy (Fisheries NSW) for providing recreational fishing effort data and their expert help in applying it. I am appreciative of Dr M. Cryer and an anonymous reviewer whose constructive feedback improved the manuscript.

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Handling editor: Mark Gibbs