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## Ecological risk assessment and its application to elasmobranch conservation and management

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Ecological risk assessments (ERAs) are employed to quantify and predict the vulnerability of a particular species, stock or population to a specific stressor, *e.g.* pollution, harvesting, climate change, by-catch. Data generated from ERAs are used to identify and prioritize species for implementation of effective conservation and management strategies. At this time, ERAs are of particular importance to elasmobranchs, given the ecological importance and documented global population declines of some elasmobranch species. Here, ERAs as a tool for elasmobranch conservation and management are reviewed and a theoretical roadmap provided for future studies. To achieve these goals, a brief history of ERAs and approaches used within them (in the context of elasmobranchs) are given, and a comprehensive review conducted of all ERA studies associated with elasmobranchs published between 1998 and 2011. The hazards assessed, species evaluated and methodological approaches taken are recorded. Chronological and geographical patterns suggest that this tool has grown in popularity as a commercial fishery management instrument, while also signalling a recent precautionary approach to elasmobranch management in commercial fisheries globally. The analysis demonstrates that the predominant parameters incorporated in previous ERAs are largely based on life-history characteristics, and sharks have received the majority of attention; batoids (including skates) have received less attention. Recreational fishing and habitat degradation are discussed as hazards which warrant future investigation through ERA. Lastly, suggestions are made for incorporating descriptive ecological data to aid in the continued development and evolution of this management tool as it applies to future elasmobranch conservation.

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### INTRODUCTION

The Earth's ecosystems, habitats and species are being challenged by a growing collection of anthropogenic stressors such as pollution, overharvest, species invasion and climate change (including the projected changes in sea level, weather patterns and ocean acidification), to name a few. In light of the global scope of these threats, the linkage between environmental scientists and policy makers is becoming increasingly

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important, highlighting the need for effective and well-informed management. As such, substantial research effort has sought to investigate the risks of these global hazards both individually and collectively, where risk is defined as the likelihood of something happening that will have an undesirable outcome within a certain time (Burgman, 2005; Fletcher, 2005).

Understanding the ecological risk of a specific hazard, threat or stressor (used interchangeably to describe a source of potential harm or adverse effect) on an ecosystem, habitat or species (hereafter only species are referred to) is a common theoretical approach to assessing or guiding the solutions of large-scale environmental problems. Ecological risk assessment (ERA) is a useful and flexible tool that informs decision-making by defining a hazard and assessing, qualitatively or quantitatively, a species' response to its exposure (Lackey, 1994; Suter, 2000; Hope, 2006). In many cases, this method is used to estimate the influence of human actions on natural biological systems and processes (Burgman, 2005). Broadly, this tool seeks to identify a critical issue and determine how an ecological asset will be affected. Throughout this process, a greater understanding (*i.e.* the risk analysis) of the relationship between a hazard and its consequences can be reached. The ERAs integrative model and approach generally involves several steps: (1) defining the assessment context, (2) assessing components of vulnerability and (3) integrating vulnerability components to derive the predicted vulnerability of each species to a stressor.

Overall, this type of approach has been employed to quantitatively and qualitatively rank the relative vulnerabilities of species in response to a particular stressor, thereby providing a framework of information for management prioritization (Burgman, 2005; Fletcher, 2005). Such generated rankings can vary in scope from a comprehensive assessment of a single species, to a group of species (ecological or trophic group), to an entire ecosystem (Crain *et al.*, 2009).

The issue of species overexploitation continually ranks among the top environmental issues globally (Kappel, 2005; Venter *et al.*, 2006). In particular, this threat's impact on marine species and ecosystems is highlighted by the increased prevalence of overexploited wild fish stocks (Myers & Worm, 2003; Worm *et al.*, 2006), decreased overall landings despite increased effort (Christensen *et al.*, 2007) and the local and global loss of species (Dulvy & Forrest, 2010). Moreover, non-target species can be profoundly affected through incidental catch (*i.e.* by-catch) of non-selective fishing methods (Crowder & Murawski, 1998; Lewison & Crowder, 2003). Overharvest can be especially detrimental to marine systems when the targeted species plays a keystone role in the ecosystem (Paine, 1966), such is the case with top oceanic predators (Stevens *et al.*, 2000; Myers *et al.*, 2007; Baum & Worm, 2009).

While ERA is still a relatively new research and management strategy (two to three decades old; Burgman, 2005), it is gradually becoming employed as a fisheries management tool. For example, ERAs have been applied to marine species such as the southern bluefin tuna *Thunnus maccoyii* (Castelnaud 1872) (Matsuda *et al.*, 1998; Matsuda, 2003) and sharks (Chin *et al.*, 2010; Cortés *et al.*, 2010). ERAs may be particularly effective in the case of data-poor fishes, such as when elasmobranchs are primarily components of by-catch (see 'Approaches used in ERAs' section below).

Increased exploitation and by-catch rates coupled with low resilience to fishing pressure have resulted in global population declines of several elasmobranch species (Lucifora *et al.*, 2011). A comparison of 26 shark and 151 teleost populations found that sharks exhibit twice the fishing extinction risk of teleosts (Myers & Worm, 2005).

Recent estimates suggest that some populations of large sharks have declined by up to 90% or more in areas where they were once abundant (Myers *et al.*, 2007; Ferretti *et al.*, 2008). While the magnitude of these estimates has been questioned (Burgess *et al.*, 2005), there is mounting concern among the public, conservationists, scientists and policy makers to understand what these population declines and subsequent impacts on ecosystem health will mean from ecological, economic and public health perspectives, and how a changing climate may exacerbate the situation (Dulvy *et al.*, 2008; White & Kyne, 2010; Gallagher & Hammerschlag, 2011; Simpfendorfer *et al.*, 2011).

In this paper, a summary of ecological risk assessment as a conservation and management tool for elasmobranchs is presented. By describing the scope of ERA techniques, approaches, as well as a comprehensive review of past elasmobranch ERA studies, a framework is provided for future elasmobranch ERAs. Also, additional hazards to elasmobranch populations, which should be considered within ERAs, are identified and described and provide a platform for future investigation by including other sources of ecological data to fit within an ERA framework.

## MATERIALS AND METHODS

Peer-reviewed publications were selected from the Science Citation Index Database (Web of Science; <http://isi5.newisiknowledge.com/>) using the title and keyword searches: 'ecological risk assessment', 'elasmobranch' and 'risk analysis elasmobranch'. The search generated papers that fell between the earliest records of the database (1945) up to June 2011. The more than 100 publications was filtered on two criteria: (1) the study must have been published in the primary literature and (2) the purpose of the study was to generate/analyse the sensitivity of an elasmobranch to an identified hazard (*e.g.* vulnerability to fishing). For example, several studies have generated life-history characteristics for elasmobranchs that aid in assessing risk, but were excluded from the analysis because they did not generate their own risk assessment. Additional papers were added from authors' personal libraries and literature cited sections from relevant papers were surveyed until no further publications arose.

The final list of publications were analysed for the following variables: (1) major ocean basin of study (Atlantic, Pacific, Indian Ocean; following FAO Major Fishing Areas; FAO, 2011), (2) year of publication, (3) classification of elasmobranch assessed (shark, skate, non-skate batoid), (4) total number of species assessed, (5) major hazard assessed (*e.g.* commercial fishing and climate change) and (6) methodological approach. To categorize the methodological approach used in each study, Dulvy *et al.*'s (2004) three-pronged classification system for assessing extinction risk in marine fishes was followed: (1) life-history characteristics (*i.e.* age at maturity, natural mortality and the von Bertalanffy growth parameter  $k$ ), (2) time-scale data (population growth rate  $\lambda$  and maximum population growth rate  $r$ , spawner–recruit parameters) and (3) demographic modelling (fishing mortality rate  $F$ , rebound potential estimates). If any study metrics were not obvious in the manuscript, available supplementary material was investigated and contact was made with the authors.

## CHRONOLOGY AND GEOGRAPHY

Fifteen peer-reviewed published ERA studies were collated, the first of which was published in 1998 (Punt & Walker, 1998). The search identified only two additional studies that were conducted up until 2006, with a gap between 2003 and 2005 (Table I). From 2006 to 2011, the search generated 12 more studies (80% of

TABLE I. Results from literature review of all previously published ERA studies for (or including) elasmobranchs. Information on year, ocean basin, number of species, major hazard assessed and methodological approach (life history, time-series, demographic) for each study is presented (Y, 'Yes' for the inclusion of a given methodological approach)

Reference	Ocean basin	No of species assessed	Hazard	Life history	Time-series	Demographic
Aires-da-Silva & Gallucci (2007)	Atlantic	1	Commercial fishing: longline			Y
Astles <i>et al.</i> (2009)	Pacific	7	Commercial fishing: trawl		Y	
Arribabalaga <i>et al.</i> (2011)	Atlantic	27	Commercial fishing: purse-seine and longline	Y		Y
Braccini <i>et al.</i> (2006)	Pacific and Indian	1	Commercial fishing: various	Y		
Chin <i>et al.</i> (2010)	Pacific	133	Climate change	Y		
Cortés <i>et al.</i> (2010)	Atlantic	11	Commercial fishing: longline	Y		Y
García <i>et al.</i> (2008)	Global	105	Meta-analysis	Y		Y
Griffiths <i>et al.</i> (2006)	Indian	56	Commercial fishing: trawl	Y	Y	
Hobday <i>et al.</i> (2011)	Pacific and Indian	139	Commercial fishing: various	Y	Y	Y
Patrick <i>et al.</i> (2010)	Pacific and Atlantic	51	Commercial fishing: various	Y		
Punt & Walker (1998)	Pacific and Indian	1	Commercial fishing: gillnet		Y	
Simpfendorfer <i>et al.</i> (2000)	Indian	1	Commercial fishing: gillnet	Y	Y	
Stobutzki <i>et al.</i> (2002)	Pacific and Indian	56	Commercial fishing: trawl	Y		
Tovar-Ávila <i>et al.</i> (2010)	Pacific and Indian	1	Commercial fishing: various	Y		Y
Zhou & Griffiths (2008)	Pacific and Indian	51	Commercial fishing: trawl		Y	Y

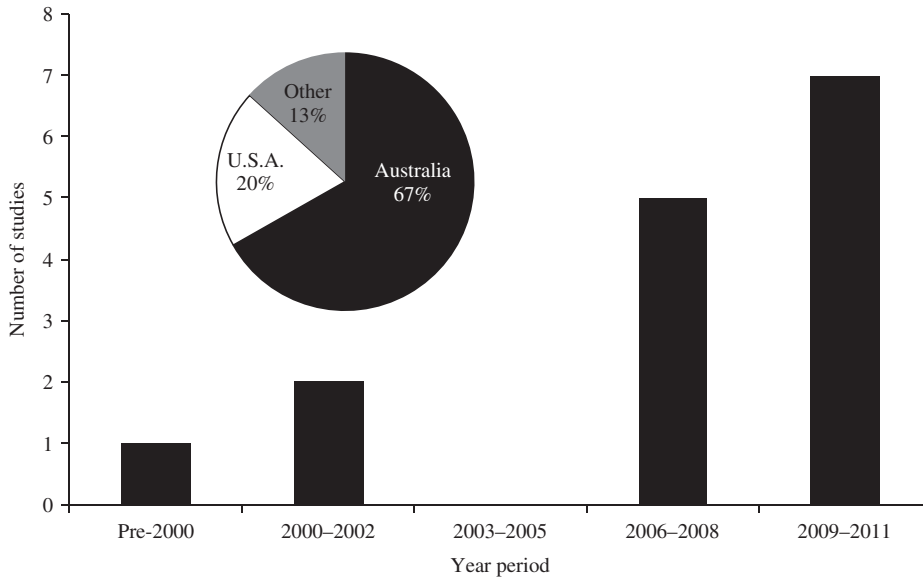


FIG. 1. The number of elasmobranch ERA studies from 1998 to 2011 and their country of origin.

elasmobranch ERAs; Fig. 1), a trend which probably reflects: (1) a growing attention to elasmobranch conservation in the last decade, (2) an increase in the application of ERAs as a method of precautionary management in light of stock crashes of major fishes (Rosenberg & Brualt, 1993; Astles *et al.*, 2006) and (3) governmental response to the criticisms of the traditional assessment of wild fish stocks stemming from uncertainty (Astles *et al.*, 2006). Furthermore, between 2003 and 2005 there were some prominent (and sometimes controversial) publications with high exposure that suggested previously undocumented, large population declines for elasmobranchs in the Atlantic Ocean (Baum *et al.*, 2003; Baum & Myers, 2004; Burgess *et al.*, 2005), and these may have directed more attention at elasmobranchs within fisheries management arenas (including their incorporation into ERAs).

In terms of ocean basin of assessment, 46.6% of studies considered the Indo-Pacific Oceans, 20.0% considered the Atlantic Ocean alone, 13.3% the Pacific Ocean alone and 6.7% each of the Pacific and Atlantic Oceans combined and the Indian Ocean alone, and globally (Table I). Studies that considered two ocean basins were those that looked at a commercial fishery operating in more than one ocean basin. For example, Australia's Northern Prawn Fishery (NPF), which has been the subject and case study for the development fisheries-related ERA methodology in Australia (Milton, 2001; Stobutzki *et al.*, 2001, 2002; Griffiths *et al.*, 2006; Zhou & Griffiths, 2008), extends across an expansive area of northern Australia, encompassing waters of the western central Pacific and the eastern Indian Ocean. At a finer scale, the geographical spread of published ERAs is skewed towards studies originating primarily from just two countries, Australia and the U.S.A. (Table I). This result may be explained by long-term fisheries management policy and current fisheries initiatives.

Scandol *et al.* (2009) considered Australia to be 'at the forefront of risk-based fisheries management', and Australian fishery managers at both the Commonwealth

and the State and Territory level have utilized ERAs as a methodology for supporting ecosystem-based fisheries management (EBFM) (Smith *et al.*, 2007; Scandol *et al.*, 2009; Hobday *et al.*, 2011). Smith *et al.* (2007) outline the evolution of EBFM in Australia, which stems from a number of policy approaches, including a national approach to ecologically sustainable development. The use of ERAs in Australia has been widely implemented as a result of directed investment into their application. For example, a series of Commonwealth-funded research projects into developing an ERA framework (Hobday *et al.*, 2011) have progressed to the development of national guidelines for the application of risk-based methods for fisheries management (Scandol *et al.*, 2009). In addition, the application of the Ecological Risk Assessment for the Effects of Fishing (ERAEF) framework to all Australian Commonwealth-managed fisheries is now well advanced (AFMA, 2011), and elasmobranchs are incorporated into many assessments as either target species, by-product and by-catch species, or threatened, endangered and protected (TEP) species. In the eastern Hemisphere, the use of ERAs in federally managed U.S. fisheries has its foundation in the need to assess the vulnerability of data-poor stocks to overfishing (Patrick *et al.*, 2009, 2010). While the term vulnerability is widely used in U.S. fishery management guidelines, there is a need to find appropriate methodology to assess it, given its importance in defining the structure and nature of management for U.S. fish stocks (Patrick *et al.*, 2009, 2010). The ERA approach has also been recommended for use by the U.S. National Oceanographic and Atmospheric Administration and by at least one regional fisheries management authority, the International Commission for the Conservation of Atlantic Tunas (Cortés *et al.*, 2010). This has seen the technique applied directly to pelagic sharks (Cortés *et al.*, 2010), and to pelagic fisheries for which elasmobranchs, particularly carcharhiniforms (ground sharks), are by-catch (Arrizabalaga *et al.*, 2011).

While ERA usage in the context of marine and elasmobranch fisheries has been thus far dominated by Australia and the U.S.A., risk analysis techniques have also been tailored to the fisheries management requirements of other nations, *e.g.* New Zealand (Campbell & Gallagher, 2007); it is hoped that this will see the publishing of ERAs as applied to either target or by-catch elasmobranch fisheries (including recreational, as discussed later) in a wider group of countries.

## APPROACHES USED IN ERAS

A variety of techniques and analyses are commonly used in ERAs. Here, these methods are summarized and how and where they have been used in previous studies is described.

The qualitative approach has been suggested as a flexible and repeatable paradigm, particularly when species information for a given fishery or habitat is lacking or deficient (Astles *et al.*, 2009). More common is the semi-quantitative ERA, susceptibility-recovery analysis (SRA), later modified and termed productivity-susceptibility analysis (PSA; PSA will be used for references to both techniques), which was initially directed at data-poor by-catch communities (Stobutzki *et al.*, 2001), including sea snakes (Milton, 2001) and elasmobranchs (Stobutzki *et al.*, 2001). This technique was concurrently developed by Milton (2001) and Stobutzki *et al.* (2001, 2002) for use in the Australian NPF, and subsequently taken up by others (Griffiths *et al.*, 2006;

Patrick *et al.*, 2010). It is considered an effective assessment tool for these by-catch species for which there is often little information available (Stobutzki *et al.*, 2001, 2002; Griffiths *et al.*, 2006; Zhou & Griffiths, 2008), and has been readily applied to elasmobranchs.

The PSA method considers (1) the susceptibility of a species to a hazard (*e.g.* a commercial fishery) and (2) the recovery potential or productivity of a species after (real or potential) depletion by the hazard (*i.e.* fishing activities). The susceptibility of a species is dependent on the level of interaction of the species with the fishery; the species' recovery or productivity is dependent on its ecological and biological attributes, these attributes determine its productivity and hence its ability to recover after depletion. The possible attributes that comprise each of the components susceptibility and recovery or productivity are numerous (Patrick *et al.*, 2010). Examples of productivity attributes include age at maturity, maximum age, fecundity, maximum size, size at maturity, reproductive strategy (*i.e.* live bearer, demersal egg layer or broadcast spawner) and trophic level (Hobday *et al.*, 2011). Examples of susceptibility attributes include overlap of species range with hazard, global distribution, habitat overlap with hazard, depth overlap with hazard, selectivity of hazard (*i.e.* fishing gear) and post-capture mortality (Hobday *et al.*, 2011). The analysis assigns a susceptibility rank and a recovery or productivity rank to each species, the combination of which highlights those species most at risk to the hazard (Milton, 2001; Stobutzki *et al.*, 2001, 2002; Griffiths *et al.*, 2006; Hobday *et al.*, 2011).

The Australian ERAEF framework provides a method of screening ecosystems, communities, habitats or species of interest and subsequently subjecting them to risk assessments of varying complexity, *i.e.* a level 1 (qualitative) risk assessment, a level 2 (semi-quantitative) risk assessment and a level 3 (quantitative) risk assessment (Smith *et al.*, 2007; Hobday *et al.*, 2011). The potential vulnerability of elasmobranchs to population depletion, which stems from their generally conservative life histories and often substantial contribution to by-catch (Camhi *et al.*, 1998; Stevens *et al.*, 2000), makes level 2 semi-quantitative approaches (*e.g.* PSA) particularly relevant to this group. Although it is generally accepted that chondrichthyans are at risk of overexploitation and subsequent population depletion, and once depleted may have a limited ability to recover from reduced population levels (Pratt & Casey, 1990; Camhi *et al.*, 1998), it is also recognized that there is a range of biological productivities amongst the sharks and batoids (Smith *et al.*, 1998). Consequently, fishing activities may impact different species to differing degrees; PSA allows information on the productivities of individual species to be incorporated into an assessment of fishing impacts (Fig. 2). Level 3 risk assessments are more difficult for elasmobranchs as the higher data requirements of a fully quantitative approach often restrict their use, particularly for by-catch species. It should be noted that while data requirements and cost increase with each level of assessment, uncertainty decreases (Hobday *et al.*, 2011). This is a key attribute of the hierarchical framework (Hobday *et al.*, 2011), and should encourage the collection of necessary data to facilitate level 3 risk assessments, when appropriate.

In data-poor situations within the PSA method, where species-specific data are not available, information can be drawn from similar species or a precautionary approach can be taken. The latter uses the assumption of high risk in the absence of information (Hobday *et al.*, 2011). While this approach encourages the collection of information to reduce the uncertainty associated with this assumption, ranking unknown attributes

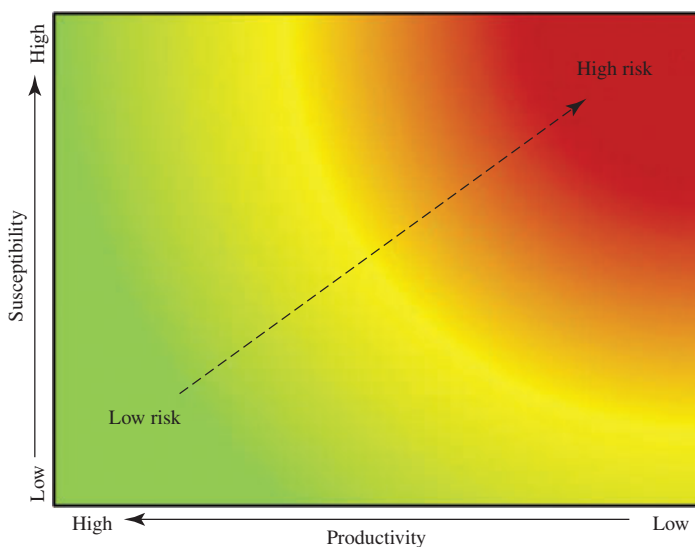


FIG. 2. Schematic of a common productivity-susceptibility analysis (PSA) plot used widely in semi-quantitative ecological risk assessments.

of a species within an ERA framework as high risk can overestimate risk (unpubl. data). Hobday *et al.* (2011), however, argue that a bias towards these false positives (erroneously assessing a species at higher risk than would occur if assessed with more data) is better than one towards false negatives (erroneously assessing at lower risk). Within the ERAEF framework, false positives can be screened at a higher assessment level, whereas false negatives could see the incorrect elimination of at-risk species (Hobday *et al.*, 2011).

Chin *et al.* (2010) integrated a fisheries-based ERA methodology with a climate change vulnerability assessment framework (Johnson & Marshall, 2007). The climate change vulnerability framework considered the exposure of a species to climate change stressors (hazards), its sensitivity to these hazards and its adaptive capacity, *i.e.* its ability to respond to change. In Chin *et al.* (2010), the component exposure was analogous to susceptibility and the components sensitivity and adaptive capacity (termed rigidity in Chin *et al.*, 2010) were analogous to recovery or productivity within the above-mentioned PSA risk assessment framework. The Integrated Risk Assessment for Climate Change (IRACC) developed by Chin *et al.* (2010) also had much the same aim as semi-quantitative fisheries ERAs, that is to provide 'a simple and transparent mechanism to assess the vulnerability of individual species to climate change (the hazard) even when there are few data available' (Chin *et al.*, 2010). Species ranked by the IRACC as having a high vulnerability to the impacts of the hazard (climate change) were those species most at risk from the hazard; their identification is a critical first step in defining necessary management, mitigation or conservation actions. Integrated and novel ERAs such as that developed by Chin *et al.* (2010) can be easily transferable to other elasmobranch contexts, whether different species groups, alternative geographic regions or different stressors or hazards.

Considering the ERAEF framework (Hobday *et al.*, 2011), the majority of published studies incorporating elasmobranchs (60%) have involved level 2 (semi-quantitative)



risk assessments. Studies using only a semi-quantitative assessment were the SRAs of Stobutzki *et al.* (2002) and Griffiths *et al.* (2006), the PSAs of Arrizabalaga *et al.* (2011) and Patrick *et al.* (2010) and the IRACC of Chin *et al.* (2010). Encouragingly, 60% of studies have also incorporated a level 3 risk assessment. Studies using *only* a quantitative assessment were the demographic modelling approaches of Punt & Walker (1998), Simpfendorfer *et al.* (2000) and Aires-da-Silva & Gallucci (2007); the level 3 PSA of Cortés *et al.* (2010) and the SAFE (Sustainability Assessment for Fishing Effects) technique of Zhou & Griffiths (2008). García *et al.* (2008) and Tovar-Ávila *et al.* (2010) incorporated semi-quantitative and quantitative approaches. In the case of Tovar-Ávila *et al.* (2010), who worked within the ERAEF framework, a level 2 risk assessment (PSA) was augmented with a level 3 risk assessment that included demographic modelling. Braccini *et al.* (2006) developed the only ERA for an elasmobranch to employ the full hierarchical approach of the ERAEF framework. Braccini *et al.* (2006) systematically moved through level 1, 2 and 3 risk assessments to determine the risk that the piked spurdog *Squalus megalops* (Macleay 1881) faces from the cumulative effects of several fishing methods. Hobday *et al.* (2011) used an Australian trawl fishery, in which several elasmobranchs are a by-catch, to illustrate the ERAEF approach.

Dulvy *et al.* (2004) describe three different approaches to assessing the extinction risk of a species; here this classification is used to categorize methodological patterns in elasmobranch ERAs and the level 2 risk assessments, including the SRA and PSA methods, correspond to correlative approaches based on knowledge of life histories and ecology or what Dulvy *et al.* (2004) termed 'life history and rule of thumb approaches'. The fact that more studies fall into this category (73%) rather than either 'time-series approaches that examine changes in abundance' (40%) or 'demographic approaches based on age- or stage-based schedules of vital rates and fisheries reference points' (46%) again highlights the use of ERAs in data-poor situations (the general case of elasmobranchs as by-catch in commercial fisheries). Time-series and demographic approaches are considerably more data intensive than life-history approaches, which can provide a 'rapid' assessment of risk.

Qualitative and semi-quantitative risk assessments can highlight elasmobranch species most at risk from a particular hazard, *e.g.* a particular commercial or recreational fishery, the cumulative impacts of several fisheries, habitat loss or the projected impacts of climate change. Methods such as the PSA are relative, *i.e.* species A is more at risk than species B, and this suggests where management and conservation measures need to be applied. To that end, risk assessment results can inform assessments for the IUCN Red List of Threatened Species, and highlight the susceptibility of a species to a hazard (or the risk posed to a species by a hazard). Level 2 risk assessments do not, however, provide the quantitative data required to directly assess a species against IUCN criteria for threatened categories. In contrast, level 3 risk assessments can assist the Red List process.

## SPECIES ASSESSED

The collection of previous ERA studies that have assessed elasmobranchs generally fall into two categories: (1) larger, community- or ecosystem-wide investigations across multiple taxa (*i.e.* mammals, bony fishes, elasmobranchs and birds) or (2) more

narrowly focused studies on one or more species of elasmobranch. Among studies which were the former, elasmobranchs were consistently ranked as the most at-risk and sensitive grouping of animals (Patrick *et al.*, 2010; Arrizabalaga *et al.*, 2011; Hobday *et al.*, 2011). This review revealed that, to date, a total of 291 different elasmobranch species have received some degree of assessment through a published ERA, with carcharhiniform sharks [predominantly the family Carcharhinidae (requiem sharks)] and several members of the batoid family Dasyatidae (whip-tail stingrays) receiving the most attention. Sharks were assessed in 100% of the reviewed ERA studies (153 species), followed by non-skate batoids 47% of the time (79 species) and skates 22% (51 species). Pelagic species such as the blue shark *Prionace glauca* (L. 1758), shortfin mako *Isurus oxyrinchus* Rafinesque 1810 and common thresher shark *Alopias vulpinus* (Bonnaterre 1788) were assessed across multiple studies, a reflection on their prominence in longline fisheries. The ERA of U.S. Atlantic longline fishing impacts on pelagic sharks conducted by Cortés *et al.* (2010), for example, concluded that many species of pelagic shark are vulnerable to fishing pressure due to their much lower biological productivities, with the silky shark *Carcharhinus falciformis* (Bibron, in Müller & Henle 1839), *I. oxyrinchus* and the pelagic thresher shark *Alopias pelagicus* Nakamura 1935, showing the highest risk levels. Conversely, Arrizabalaga *et al.* (2011) suggested that coastal elasmobranch species (mainly sharks) tended to rank higher than their pelagic counterparts across the E.U. and U.S. longline, purse-seine and gillnet fisheries. Such a variation in risk rankings between eco-groupings, while probably context (fishery)-dependent, provides the basis for future ERAs that focus on coastal species.

At the finer scale, ERAs have been able to predict the most influential determinants of an elasmobranch species' overall vulnerability to fishing. For example, Aires-da-Silva & Gallucci (2007) concluded that through demographic modelling, the higher biological productivity of the *P. glauca* decreased its overall risk of over-exploitation to commercial fishing in the Atlantic Ocean, yet that the juveniles should receive protection from harvest. Similarly, Tovar-Ávila *et al.* (2010) cited the low discard mortality rate of the Port Jackson shark *Heterodontus portusjacksoni* (Meyer 1793) in Australian fisheries by-catch as driving its low vulnerability to exploitation. Conversely, other studies have found certain species to be highly vulnerable to by-catch due to low productivity, decreased mobility and schooling behaviour, such as *S. megalops* (Braccini *et al.*, 2006), the school shark *Galeorhinus galeus* (L. 1758) (Punt & Walker, 1998), the whiskery shark *Furgaleus macki* (Whitley 1943) (Simpfendorfer *et al.*, 2000) and various benthic non-skate batoids (Astles *et al.*, 2009).

In comparison with sharks, non-skate batoids and skates probably receive less scientific investigation in the literature (including for ERAs) because most are not as commercially important. Stobutzki *et al.* (2001) described this group of elasmobranchs as highly sensitive to non-selective fishing practices, due to their small size at time of capture and high rates of in-net mortality. Perception may also influence the choice of focal species included in ERA studies. For example, Dulvy & Forrest (2010) demonstrated that conservation and research interests of elasmobranch biologists were not evenly distributed across species, but skewed towards the larger, more charismatic species. They found that relative to the proportion of extant elasmobranchs, sharks of the orders Carcharhiniformes and Lamniformes (mackerel sharks)

were over-represented in conservation and research interests, with batoids (including skates) under-represented (Dulvy & Forrest, 2010).

## HAZARDS ASSESSED

Given the resource constraints of modern fisheries managers, coupled with the progression to EBFM, it is not surprising that the ERA framework has been so readily applied to the management of commercial fisheries. In fact, with the exception of one study, all elasmobranch ERA papers were based on commercial fisheries (Table I). Chin *et al.* (2010) is the only published elasmobranch ERA that looked at a non-fisheries hazard (climate change).

While fisheries applications have considered trawl, gillnet, longline and purse-seine fisheries, ERAs (and risk analysis studies) have generally been applied where elasmobranchs are by-catch, with few applications to targeted species. This again highlights the utility of ERAs for by-catch species and, given the propensity of applications to data-poor situations, the utility of its application where traditional stock assessment and demographic techniques are unsuitable due to data requirements. With regard to its application to targeted species, Punt & Walker (1998) and Simpfendorfer *et al.* (2000) use risk analysis on single target species, and these are within a demographic analysis (quantitative) approach, while Cortés *et al.* (2010) used a species' biological productivity and susceptibility to rank the vulnerabilities to longline fishing for 11 elasmobranch species.

## FUTURE ERAS

### RECREATIONAL FISHING

This review failed to identify any ERAs for elasmobranchs assessing their vulnerability to recreational fishing. While often overshadowed by commercial fishing, there is mounting evidence suggesting that recreational fisheries play a significant factor in the exploitation of marine fishes (Pauly *et al.*, 2002; Cooke & Cowx, 2004). It has been estimated that recreational fishing accounts for *c.* 10% of the total global fishing harvest, with an estimated 47 million fish landed globally (Coleman *et al.*, 2004; Cooke & Cowx, 2004). While large-scale impacts of this industry are often less understood and harder to detect, Post *et al.* (2002) reported on the collapse of four inland fisheries in Canada as a direct result of recreational fishing. In addition, it has been suggested that some recreational fisheries actually exhibit a higher harvest than their commercial counterpart (Schroeder & Love, 2002).

Catch and release practices are increasingly popular in recreational fisheries, yet the sustainability of this practice relies upon the major assumption that landed fishes survive and remain healthy after release, and that fished stocks are continually reproducing and well maintained (Cooke & Schramm, 2007). As such, a growing number of studies have sought to investigate and predict the post-release behaviour of landed fishes, a metric which has direct implications for survival and fitness (Cooke & Suski, 2005). When a fish suffers post-release mortality, the overall effect is analogous to a discard or by-catch mortality encountered through commercial fishing

(Alverson, 1997). Due to the seemingly nebulous impacts of recreational fisheries on elasmobranchs (Lynch *et al.*, 2010), coastal management would benefit significantly from integrated ERAs taking recreational elasmobranch fishing into consideration (for both harvest and catch and release), particularly in areas of distinct overlap between elasmobranch nursery and pupping grounds and sustained recreational fishing pressure.

## HABITAT DEGRADATION

This review failed to identify any ERAs for elasmobranchs assessing their vulnerability to habitat degradation. Estuarine and coastal transformation has, however, accelerated significantly in the past 100–300 years (Lotze *et al.*, 2006), and reductions in habitat quality and concomitant decreases in species diversity (Courrat *et al.*, 2009) have been observed in a variety of marine habitats including seagrass (Pihl *et al.*, 2006), mangroves (Mumby *et al.*, 2004; Faunce & Serafy, 2006), estuaries (Van Dyke & Wasson, 2005) and coral reefs (Mumby *et al.*, 2007; Ward-Paige *et al.*, 2010). Moreover, the quality of habitat is a major biotic driver of ecosystem health and production, having direct correlative influences on the recruitment, growth and survival of juvenile fishes (Peterson *et al.*, 2000; Thrush *et al.*, 2008).

Numerous species of elasmobranchs globally use inshore coastal and estuarine habitats for feeding, mating, parturition and early-stage somatic growth, where they are disproportionately vulnerable to habitat modification. Inshore coastal habitats throughout southern Florida, U.S.A., alone provide critical habitat for a large number of carcharhinids (Heupel *et al.*, 2006; Yeiser *et al.*, 2008). As such, maintaining essential habitat for elasmobranchs has been documented as a management priority in light of a growing human presence (Jennings *et al.*, 2008; Knip *et al.*, 2010); however, understanding how habitat degradation affects elasmobranch species has only just begun. Jennings *et al.* (2008) conducted a long-term study on the effects of development on juvenile lemon shark *Negaprion brevirostris* (Poey 1868) growth and survival throughout Bahamian nursery grounds and found a *c.* 25% decline in juvenile survival after dredging activity. Thus, ERAs for coastal species will benefit by including parameters (*e.g.* fecundity and dispersal) that can influence a population's vulnerability to habitat degradation (Akçakaya, 2001; Baldwin & Demaynadier, 2009).

## INCORPORATING ECOLOGICAL SPECIALIZATION

The degree of ecological specialization can be a major determinant of a species' total vulnerability to environmental and anthropogenic hazards (Dulvy *et al.*, 2003; Hobbs *et al.*, 2010). While many of the previous ERAs have had the foresight to include some level of ecological specializations into their analyses for elasmobranchs (Arrizabalaga *et al.*, 2011), there is room for continued development. Risk assessment studies on seabirds, for example, have included more complex variables such as the susceptibility to hooking (ability to dive), the propensity to switch diet and even bioenergetic models (Furness & Tasker, 2000; Waugh *et al.*, 2008). In the following sections, three types of ecological specialization data are suggested for inclusion in elasmobranch ERAs.

## COMPARATIVE ECO-PHYSIOLOGY

Investigations into the acute changes in an individual's physiology can provide detailed information on the impact of potential disturbance factors, whereby these data can function as a type of conservation physiology (Wikelski & Cooke, 2006). In some cases, it has been shown that the physiological disturbances resulting from fishing activities on released by-catch teleost species have the potential to influence population declines (Davis & Olla, 2001; Wikelski & Cooke, 2006; Davis, 2007).

An example from the avian literature best exemplifies the utility of physiological data in risk assessments. Based in the Falkland Islands, Bevan *et al.* (1995) assessed the potential risks of longline fishing on population declines of the black-browed albatross *Thalassarche melanophris*. In their study, the authors combined heart rate monitors and internal temperature probes in conjunction with animal-borne satellite tags to determine overlap between fishing pressure and bird foraging. Using these data, Bevan *et al.* (1995) further demonstrated how overfishing changed albatross energetic budgets and subsequently influenced their survival.

While the majority of studies assessing physiological impacts of fishing on elasmobranchs have been conducted on carcharhinids (Mandelman & Skomal, 2009), results have provided evidence for a strong degree of interspecific variation in the stress response, which in turn can lead to varying levels of mortality (Morgan & Burgess, 2007). An example from the authors' own physiological studies further illustrates this point, where hammerhead sharks *Sphyrna* spp. show extreme disruptions in physiological condition and increased post-release mortality in response to fishing pressure (fight time), whereby the tiger shark *Galeocerdo cuvier* (Peron & Lesueur in Lesueur 1822) stress response is completely the opposite (A. J. Gallagher, S. J. Cooke & N. Hammerschlag, unpubl. data).

There are some cases of quantitative physiological data (*i.e.* discard mortality) being used within existing elasmobranch ERAs (Chin *et al.*, 2010; Cortés *et al.*, 2010; Tovar-Ávila *et al.*, 2010). There is a suite of tools and evolving technologies available to researchers to conduct rapid, data-rich physiological studies on elasmobranchs in the field, such as diagnostic blood analysers for stress analyses and quick deployment of satellite transmitters and other bio-logging devices to estimate post-release survivorship and swimming behaviour (Skomal, 2007; Gallagher *et al.*, 2010). Information on the intraspecific differences in physiological disturbance across fight times and fishing gear, as well as the resulting proportions of post-release mortality by species, could be categorically ranked and inserted into an ERA for non-target and catch and release fishing.

## FEEDING AND DIET INDICES

Specialization of a species in terms of diet may increase a species' vulnerability to anthropogenic or natural habitat loss. In theory, if a given population is composed largely of generalist predators (large dietary niche breadths, consuming a diversity of prey), then they should be more resilient to any ecosystem alterations which may affect their food availability (Gaston, 1994; Kunin & Gaston, 1997). In contrast, if a given population is composed of specialists (narrow dietary niche breadth, consuming few prey types), then they should be more susceptible to anthropogenic changes which affect their food supply (Fig. 3).

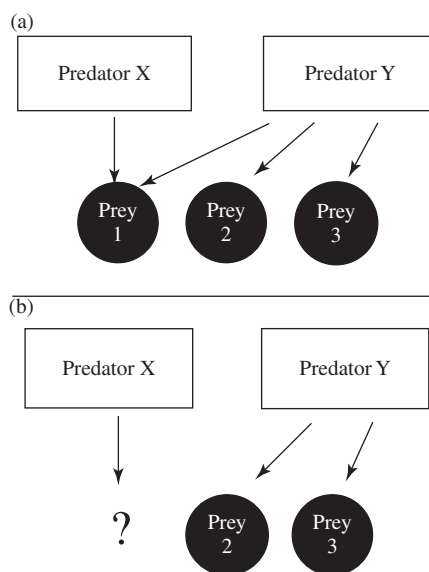


FIG. 3. Dietary breadth of a specialist (predator X) and a generalist (predator Y). (a) Two predators displaying different feeding strategies. (b) How is predator X's ability to survive affected by this situation? Its ability to recover would be useful information for ranking dietary plasticity (and survival potential) of a subset of elasmobranch species.

Information on diet and trophic level has been incorporated into several level 2 (semi-quantitative) risk assessments that have been applied to elasmobranchs. Stobutzki *et al.* (2002) and Griffiths *et al.* (2006) incorporated information on a species' diet as an attribute of its susceptibility to a prawn trawl fishery (the NPF). In this sense, they considered a species to be more susceptible to capture by trawl gear if it was 'known to, or capable of, feeding on commercial prawns or benthic organisms' (Griffiths *et al.*, 2006). Moreover, dietary breadth can be calculated by utilizing previously published data or conducting empirical studies *via* stomach content analysis or stable isotope analysis (Shiffman, 2011). In some ERAs, a species' trophic level can be considered an attribute of its productivity (Patrick *et al.*, 2010; Hobday *et al.*, 2011). Here, a lower trophic level reflects higher productivity and a higher trophic level reflects lower productivity; large predatory elasmobranchs generally fall into the latter. Similarly, the ERA conducted by Chin *et al.* (2010) considered a species' trophic specificity to reflect its level of dietary specialization. An example of how to incorporate novel dietary information in an ERA for elasmobranch vulnerability to habitat loss is provided in the 'Conclusions'.

## MOVEMENTS

It has been suggested that in order to better quantify complex ecological life-history traits, future ERAs will benefit from spatial data on the movement and migration of species, particularly outside of areas where they are encountered by fishing vessels (Aires-da-Silva & Gallucci, 2007; Cortés *et al.*, 2010). Furthermore, the paucity of such information among risk assessment studies was highlighted by

Cortés *et al.* (2010), who asserted that data on vertical distribution and habitat preferences were lacking for pelagic sharks in their risk assessment of Atlantic longline fisheries. Horizontal movements may also provide useful data on geographical range and fishing encounterability.

Recent technological advances have afforded researchers the unique chance to better understand the behavioural patterns of elasmobranchs, providing tools for tracking vertical and horizontal movements of elasmobranchs that can be analysed at relatively high resolution (Hammerschlag *et al.*, 2011). These data can then be contrasted against distribution of fishing efforts to quantify exposure and susceptibility to exploitation (Brill & Lutcavage, 2001; Luo *et al.*, 2006; Goodyear *et al.*, 2008). For example, Southall *et al.* (2006) used satellite telemetry to study basking sharks *Cetorhinus maximus* (Gunnerus 1765) migration patterns within the north-east Atlantic Ocean in relation to different boundary zones that differed in protection of the species and therefore their susceptibility to fishing. Southall *et al.* (2006) demonstrated that *C. maximus* originally tagged in waters off the U.K. regularly crossed different national boundaries where they were exposed to commercial harvest. Here, it is argued that the inclusion of high-resolution information on migratory patterns for mobile elasmobranchs, in combination with geographical patterns of fishing efforts, can provide an overlap scenario to enhance framing elasmobranch ERAs investigating the impacts of fishing.

## CONCLUSIONS

This review highlights the utility of ERAs for strengthening and prioritizing the conservation and management of elasmobranchs, while suggesting additional hazards and ecological data that could be incorporated in future ERAs. In closing, this point is illustrated with a hypothetical example for evaluating the vulnerability of elasmobranch species to habitat loss in a coastal, mangrove-dominated estuary.

In such a case, it would be necessary to quantitatively assess the degree of mangrove dependence on a species-specific basis. This can be achieved by probing multiple demographic and behavioural variables such as time-series abundance and distribution within the mangroves (Serafy *et al.*, 2007) and habitat-dependent dietary specialization (Hammerschlag-Peyer & Layman, 2010). Degree of habitat specialization and dependency could be quantified by tracking movements *via* telemetry and generating a set of semi-quantitative relative range-occupancy indices (*i.e.* percent occupancy and relative rarity) for a given region (Hockey & Curtis, 2009). Dietary breadth could be quantitatively derived by calculating the mean difference in isotopic signature (for a given isotope) between multiple tissues for isotopic analysis (Hammerschlag-Peyer & Layman, 2010). The product of these two components would generate a habitat-dependency ranking, derived *via* non-lethal methods, that can be used to rank a species' risk of loss of habitat (Hockey & Curtis, 2009), with higher risk rankings suggesting higher overall dependency (Fig. 4). By using these types of ecological data, ERAs may be able to incorporate behaviourally complex interactions that can function to identify species which would be most affected by a change in habitat, while determining which aspects of the hazard (*i.e.* loss of suitable prey refugia, decrease in home range used by a predator) will most affect a species.

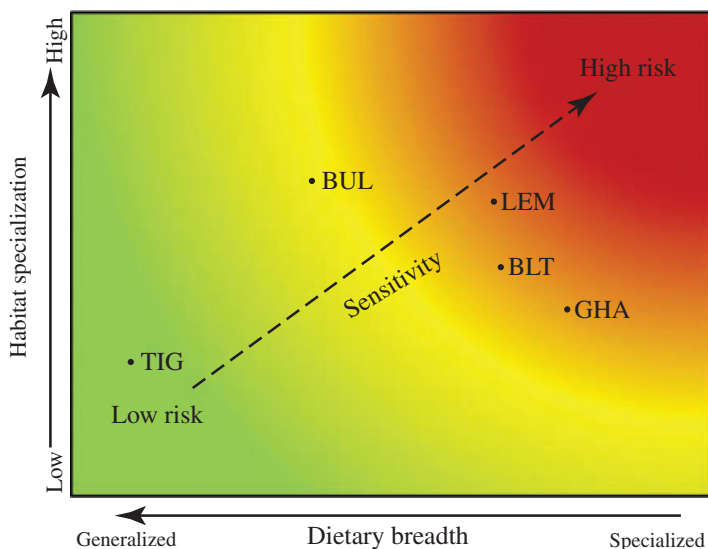


FIG. 4. Hypothetical example of an ecological risk assessment considering the vulnerabilities of five shark species commonly found in the subtropical Atlantic Ocean to loss of mangrove habitat. This figure represents an adapted form of the productivity-susceptibility model of risk that could be generated using novel ecological data for habitat specialization and dietary breadth. ---, increasing risk gradient, the product of each axis score would give a relative ranking of risk by species. Species with more generalized consumers (broad prey base) would be at lower risk, and more specialized (narrow prey base) consumers would generate a higher risk. BLT, *Carcharhinus limbatus*; BUL, *Carcharhinus leucas*; GHA, *Sphyrna mokarran*; LEM, *Negaprion brevirostris*; TIG, *Galeocerdo cuvier*.

The era of triaging species for conservation initiatives is current, and ecological risk assessment remains a powerful tool for gauging the impact of a threat while effectively linking science and policy. In addition to empirical field studies, scientists are encouraged to synthesize data from other relevant studies to maximize impact among their own ERAs, a practice that will in turn facilitate collaboration between researchers and policy makers.

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