



A decision-support tool to facilitate discussion of no-take boundaries for Marine Protected Areas during stakeholder consultation processes



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ABSTRACT

Marine Protected Areas (MPAs) are proposed to help conserve marine biodiversity and ecological integrity. There is much guidance on the optimal design of MPAs but once potential MPAs are identified there is little guidance on defining the final no-take boundaries. This is especially problematic in temperate zones where ecological boundaries are “fuzzy”, which can be quite complicated during a consultation process involving the government and divergent stakeholder groups. More decision-support tools are needed to help stakeholders and government agencies objectively compare conservation and socio-economic trade-offs among proposed boundary options. To that end, we developed a method to identify which boundary minimizes spatial overlap of highly vulnerable species and a dominant stressor. We used the recently proposed boundary options of a candidate MPA in Atlantic Canada to illustrate our method. We evaluated the vulnerability of 23 key species to bottom trawling, the most prevalent stressor in the area. We then compared the spatial overlap of the most vulnerable species and the 2002–2011 footprint of bottom trawling among boundary options. The best boundary option was identified as that which minimized spatial overlap and total area. This approach identifies boundary options which provide the greatest protection of vulnerable species from their most significant stressor, at limited socio-economic cost. It is an objective decision-support tool to help stakeholders agree on final boundaries for MPAs.

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Introduction

For generations, the ocean was thought to have a virtually limitless amount of living resources. It is now evident that this is not true. Large predatory fish have declined by approximately 90% in biomass since pre-industrial times (Myers & Worm, 2003). Exploitation and pollution, and now global climate change, are causing changes in the ocean that may not benefit humankind. This is acknowledged at a global scale, and, consequently, there is growing interest in Marine Protected Areas (MPAs) (CBD, 2012). In fact, 193 countries have signed in the Convention on Biological Diversity (CBD). As signatories to this convention, these countries have committed to increasing the number of MPAs in their waters so that, by 2020, 10% of the world's managed coastal and marine areas will be protected within nationally governed MPAs (CBD, 2012).

An MPA is a “clearly defined [marine] space, recognized, dedicated and managed, through legal or other effective means, to

achieve the long-term conservation of nature with associated ecosystem services and cultural values” (Day et al., 2012).

When designed appropriately, MPAs positively effect the biomass, diversity, and productivity of organisms, especially previously exploited populations (Halpern & Warner, 2002; Pérez-Ruzafa et al., 2008; Vandeperre et al., 2011). This in turn can increase biodiversity and productivity throughout ecosystems. These effects can occur rapidly and can spread beyond MPA boundaries (Halpern & Warner, 2002; Higgins et al., 2008; Lester & Halpern, 2008; Lubchenco et al., 2003).

There is a wealth of guidance on the optimal design of MPAs (Agardy et al., 2011; Agardy, 1994; Claudet et al., 2008; Hastings & Botsford, 2003; Higgins et al., 2008; Horsman et al., 2011; Pérez-Ruzafa et al., 2008; Vandeperre et al., 2011). Spatial optimization tools have been developed to help identify potential MPAs and MPA networks that capture all necessary qualities (Ball et al., 2009; Horsman et al., 2011). However, once potential MPAs are identified there is little guidance on defining the final no-take boundaries (Kendall et al., 2008; Young et al., 2013). Guidance is especially needed when the ecologically-defined boundaries are fuzzy as in temperate oceans, and when stakeholders have divergent expectations.

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The starting point for government-led processes towards the CBD goals is often the identification of a potential MPA based on conservation and fisheries objectives. This is followed by a public consultation process involving the responsible government agencies, industry and non-governmental organization stakeholders. During this phase, the fuzzy ecologically-based boundaries surrounding the potential MPA are discussed. Zoning and final boundary discussions can be time-consuming and contentious because of differing perspectives. The problem is so prevalent that there is an entire branch of literature devoted to understanding and reducing stakeholder conflict/distrust (Abecasis et al., 2013; Gleason et al., 2010; Hattam et al., 2014; Mangi and Austen, 2008; Stevenson & Tissot, 2013; Young et al., 2013).

Our goal was to develop and test an objective decision-support tool to compare candidate MPA boundary options proposed by stakeholders. We use the Scotian Shelf, Canada, a temperate region, to demonstrate use of this decision-support tool. As a signatory of the CBD, Canada has committed to establishing an MPA network (Government of Canada, 2011) and has identified potential constituent MPAs on the Scotian Shelf through a spatial optimization program (Horsman et al., 2011). A potential no-take MPA was identified in the St Anns Bank area on the Eastern Scotian Shelf (ESS), where ecological boundaries are “fuzzy”. Following a lengthy public consultation process, Fisheries and Oceans Canada (DFO) began planning for this MPA as the first in the network in 2011. During the stakeholder-led boundary discussion phase, industry stakeholders proposed seven different boundary options (Fig. 1) as variations of the initial boundary design (Horsman et al., 2011). Each boundary option was proposed to minimize the extent to which the MPA would encroach on some popular fishing area, but conservation benefits also vary among them. Bottom trawling can be destructive to bottom habitat (Dayton et al., 1995; Jones, 1992) and is the most prevalent stressor in the St Anns Bank area (DFO, unpublished data). Since an overarching goal of MPAs is to protect ecosystem health (Agardy, 1994; Halpern & Warner, 2002; Pérez-Ruzafa et al., 2008; Vandeperre et al., 2011), we assume protecting key species from such a significant stressor would be a priority. We develop a method to compare spatial overlap of highly vulnerable key species with bottom trawling, as well as area cost, among boundary options proposed for any MPA. We also provide a method with which marine managers can use this information to objectively identify the best option.

We demonstrate use of these methods by identifying the best boundary option for the St Anns Bank MPA.

Methods

Vulnerability assessment

A vulnerability assessment (VA) translates qualitative information about a species or system into a score or rank depicting its vulnerability to (a) significant stressor(s). VAs have been used to identify vulnerable areas in need of protection (Eno et al., 2013; Stelzenmüller et al., 2010), as well as to triage species by their vulnerability to key stressors (Chin et al., 2010; Patrick et al., 2010; Stobutzki et al., 2001). We adapted a vulnerability assessment that follows a widely accepted logic framework (Füssel & Klein, 2006; Patrick et al., 2010; Stobutzki et al., 2001). Herein, vulnerability to a given stressor, in this case, bottom trawling, is a function of sensitivity (susceptibility of a species or population to be negatively impacted by the stressor) and adaptive capacity (potential of a species or population to cope with stress, recover from adverse effects, or migrate to more favourable habitat). A species or population can only be highly vulnerable to bottom trawling if it is both highly sensitive to this activity and cannot adapt to its effects.

Table 1
Species listed by scientific and common names.

Scientific name	Common name
<i>Hemitripterus americanus</i>	Sea Raven
<i>Sebastes mentella</i> and <i>Sebastes fasciatus</i>	Redfish species
<i>Lamna nasus</i>	Porbeagle Shark
<i>Squalus acanthias</i>	Spiny Dogfish
<i>Lophius americanus</i>	Monkfish
<i>Hippoglossus hippoglossus</i>	Atlantic Halibut
<i>Leucoraja ocellata</i>	Winter Skate
<i>Gadus morhua</i>	Atlantic Cod
<i>Pollachius virens</i>	Pollock
<i>Melanogrammus aeglefinus</i>	Haddock
<i>Hippoglossoides platessoides</i>	American Plaice
<i>Anarhichas lupus</i>	Atlantic Wolffish
<i>Glyptocephalus cynoglossus</i>	Witch Flounder
<i>Malacoraja senta</i>	Smooth Skate
<i>Pandalus borealis</i>	Northern Shrimp
<i>Clupea herengus</i>	Atlantic Herring
<i>Mallotus villosus</i>	Capelin
<i>Scomber scombrus</i>	Atlantic Mackerel
<i>Placopecten magellanicus</i>	Sea Scallop
<i>Illex illecebrosus</i>	Northern Shortfin Squid
<i>Chionoecetes opilio</i>	Snow Crab
<i>Homarus americanus</i>	American Lobster

Sensitivity to bottom trawling is a function of diet, reproductive strategy and affinity to the sea bed, and natural disturbance. Adaptive capacity is a function of habitat scope for growth, population status, and life history resilience. For each factor making up sensitivity and adaptive capacity to bottom trawling, scores range from least sensitive or most adaptive (1) to most sensitive or least adaptive (5). Each factor was weighted by overall importance (*I*) to sensitivity or adaptive capacity (1 = least important, 2 = important, 3 = very important, based on the literature (Chin et al., 2010; Patrick et al., 2010; Stelzenmüller et al., 2010; Stobutzki et al., 2001). We acknowledge there is an element of subjectivity, but as in most VAs, it is essential that not all factors are treated as equally important to vulnerability (e.g. Downing et al., 2005; Hiddink et al., 2007). Justification of factors, factor weighting, and scoring is described in more detail below.

We used our VA on 23 key and/or depleted species (Table 1), many of which are conservation targets for the Scotian Shelf MPA network (Horsman et al., 2011). We then examined conservation benefits, i.e. the overlap of highly vulnerable key species and bottom trawling activity, given the spatial footprint of bottom trawl fishery activity from 2002 to 2011 (DFO, 2011), in each boundary option. We also measure the area contained within each boundary option to contrast conservation benefits and socio-economic costs.

Sensitivity factors

Factor S1: Diet (*I* = 1)

Diet information was obtained via Fishbase.com, Garrison and Link (2000), Shackell et al. (2012), and NOAA (National Oceanic and Atmospheric Administration) essential fish habitat source documents (NOAA Fisheries Science Center, 2011). Scores were based on the flexibility of a species' diet, and whether or not their prey could be directly or indirectly impacted by the fishery. This factor was weighted less heavily than others because marine animals are often food generalists. As a result, diet may not have as large of an impact on their sensitivity to bottom trawling as reproductive strategy or habitat. Scores were as follows: 1 = planktivores, 2 = pelagic piscivores, 3 = scavengers, 4 = indiscriminate benthivores, 5 = selective benthivores/benthic piscivores (majority of diet made up of one functional group).

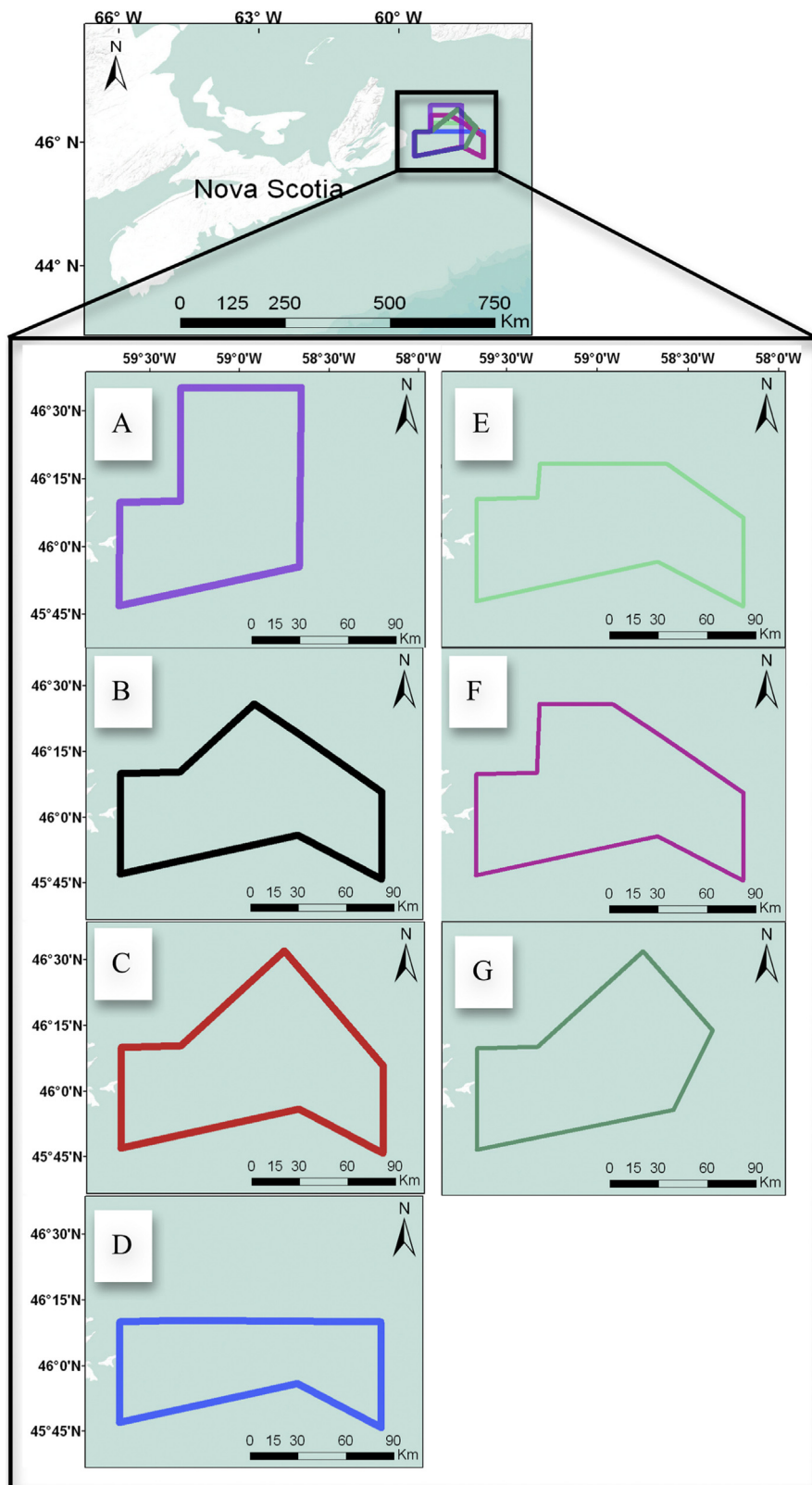


Fig. 1. Seven boundary options for the St Anns Bank MPA, Scotian Shelf, Canada: Boundary options (A) is the initial boundary proposed for the MPA. (B–G) were proposed by stakeholders.

Factor S2: Reproductive strategy ($I=2$)

This factor considers the sensitivity of early life history stages to bottom trawling in terms of both location and abundance. Trawl nets are designed to catch adult fish, but they can cause damage

to eggs or larvae that occur on the bottom. However, exploitation of adults is widely accepted as the primary cause of population decline and loss of resilience (Hutchings & Reynolds, 2004; Shelton et al., 2006; Zhou et al., 2010). For this reason, this factor was

weighted less heavily than adult habitat factors (see below). Information was obtained from Fishbase.org (Froese & Pauly, 2000). Scores for this factor were adopted from Stelzenmüller et al. (2010), who effectively assessed the sensitivity of early life stages to aggregate extraction, a bottom-intensive activity with effects similar to bottom trawling. We calculated species scores as the sum of the following: Eggs laid on sea floor = +2, eggs carried by parent = +1, or eggs planktonic = 0; Post-larvae attached to seabed = +2, post-larvae associated with seabed = +1, or planktonic post-larvae = 0; Low fecundity (<1000 eggs/yr.) = +1, or high fecundity (>1000 eggs/yr.) = 0.

Factor S3: Affinity to seabed ($I = 3$)

This factor considered the general habitat, with respect to depth, of the adult stage and how this may increase or decrease the likelihood of encountering trawl gear. This factor was weighted as very important to species sensitivity as it determines whether a species habitat coincides with the target depth of bottom trawlers, the sea floor. Information was obtained from Fishbase.org and COSEWIC.gc.ca. Scores were given as follows: 1 = pelagic, 2 = benthopelagic, 3 = demersal (near the sea bed, mobile), 4 = epibenthic (lives on the sea bed, slow-moving), and 5 = epibenthic and sedentary (or sessile).

Factor S4: Habitat natural disturbance ($I = 3$)

Kostylev and Hannah (2007) characterized habitat vulnerability on the SS using the self-defined determinants, “natural disturbance” and “scope for growth”. Natural disturbance (ND), i.e. wave disturbance and turbidity, may be an important determinant of species sensitivity to unnatural disturbances, such as bottom trawling (methods for evaluating habitat natural disturbance can be found in Kostylev & Hannah, 2007). Bottom trawlers drag gear along the sea floor, damaging habitat for many benthic species (Jones, 1992). Bottom trawling will likely cause little additional damage in areas where habitat state and food availability are continually in flux as a result of high ND (Bergman & Hup, 1992; Drabsch et al., 2001; Jones, 1992; Kaiser & Spencer, 1996; Kaiser et al., 2003). Animals that are adapted to naturally disturbed habitats are likely less sensitive to the effects of a bottom trawl fishery. Similarly, such activities can cause destruction of long-standing, stable habitat in areas with low ND (e.g. Jones, 1992; Kaiser et al., 2003). Animals living in low-disturbance habitats may depend on stable environments and will likely be more sensitive to trawling. This factor was weighted as very important to species sensitivity.

ND was divided into 5 classes (Horsman et al., 2011): very high, high, medium, low, and very low. Areas not characterized by ND were not included in the analyses. Using the Intersect tool in ArcGIS (ESRI, 2011), species distribution were assigned to an average ND classification in the St Anns Bank area. The average ND category was then translated into a sensitivity score: where 1 = very high ND, 2 = high ND, 3 = medium ND, 4 = low ND, and 5 = very low ND.

Adaptive capacity factors

Factor A1: Habitat scope for growth ($I = 2$)

Scope for growth (SG), is defined by Kostylev and Hannah (2007) as a habitat's potential to enhance growth and reproduction. SG is a function of food availability, water temperature, oxygen availability, and salinity (methods for evaluating habitat scope for growth can be found in Kostylev & Hannah, 2007). Species occurring mostly in areas with low SG may be more vulnerable to overexploitation (Fisher et al., 2011). Species that occur in more productive, high scope for growth areas are more likely to recover (Ludsin et al., 2001; Norse et al., 2012; Williams et al., 2011). This factor was weighted as important to species adaptive capacity, but less important than populations status for reasons outlined below. SG

was divided into 5 classes (Horsman et al., 2011): very high, high, medium, low, and very low. Using the “Intercept” tool in ArcGIS, species were assigned an average SG. Each species' average SG category translated into a sensitivity score: 1 = very high SG, 2 = high SG, 3 = medium SG, 4 = low SG, and 5 = very low SG.

Factor A2: Population status ($I = 3$)

We used the Committee on the Status of Endangered Wildlife in Canada, COSEWIC (www.cosewic.gc.ca), listings to score species for this factor. Note that the same scoring logic could be used with the IUCN Red List (www.iucnredlist.org) in other regions. Overexploited species are less resilient to perturbation due to decreased average size of individuals, decreased population size, restricted geographic range and limited reproductive potential due to lowered genetic diversity and changes in life history (Bundy, 2005; Fu et al., 2001; Hutchings & Reynolds, 2004; Hutchings, 2000; McCusker & Bentzen, 2010; Mora et al., 2006). Given the importance of exploitation history to resilience and survival, this factor was weighted heavily in the VA.

When a species' status could not be found through the COSEWIC species search (COSEWIC, 2013), a “working status” was used (www.wildspecies.ca; Wild Species, 2012). Scores for this factor were given as follows: 1 = Not At Risk (NAR) or “Secure”, 2 = Data Deficient (DD) or “Sensitive”, 3 = Special Concern (SC) or “May be at Risk”, 4 = Threatened (T), and 5 = Endangered (E).

Factor A3: Inherent resilience ($I = 2$)

Species with shorter generation times reproduce more quickly and so can recover from disturbances or adapt to environmental changes more quickly than species with longer generation times (Dulvy et al., 2003; Jennings et al., 1998; Musick, 1999; Reynolds, 2003). Resilience can be estimated from age at maturity, generation time, maximum size, and/or fecundity (Musick, 1999). However, these traits can change with population status. Recordings of age at maturity, generation time, maximum size, and fecundity used to inform species scores for this factor could be outdated. Species that were once quite resilient may now, due to over-exploitation, have shorter generation times, smaller maximum size, etc., and are likely less resilient (Kuparinen & Hutchings, 2012). For this reason, this factor was weighted less heavily than population status. Following guidelines from Musick (1999), species were given resilience ranks, which were then translated into factors scores as follows: 1 = high resilience, 2.3 = medium resilience, 3.6 = low resilience, and 5 = very low resilience.

Calculating vulnerability scores

A species' vulnerability was determined as the product of its sensitivity (importance-weighted sum of sensitivity factor scores) and adaptive capacity (importance-weighted sum of adaptive capacity factor scores) (Eq. (1)). Scores were rescaled from 0 to 1 for easier comprehension. Multiplication of the components ensured that our method was conservative; lower scores are more likely than higher scores (Fig. 2). This is because it is necessary to have both high sensitivity and low adaptive capacity in order to be highly vulnerable.

A Monte Carlo simulation of our VA, with randomly generated factor scores (1, 2, 3, 4, or 5), produced a distribution of all possible vulnerability scores (Fig. 2). Tertiles of this distribution allowed us to group our 23 key species into low, medium, and high vulnerability categories based on their vulnerability scores (Figs. 2 and 3). Those species falling within the high vulnerability category were considered to be of high conservation priority for the St Anns Bank MPA. Overlap with bottom trawling was compared among boundary options for these species only.

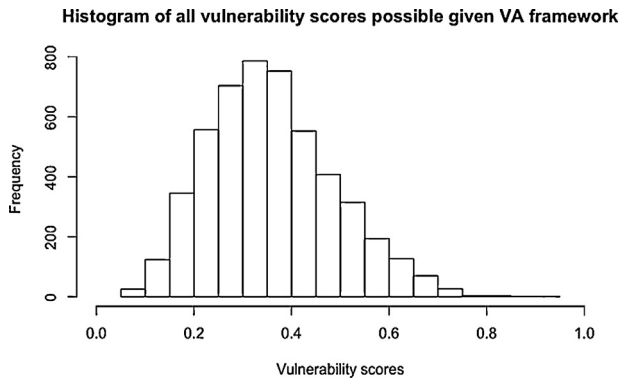


Fig. 2. Histogram of all vulnerability scores possible given the VA framework. This distribution was achieved via a Monte Carlo simulation of the VA ($n = 5000$).

We used ArcGIS (ESRI, 2011) to quantify the spatial overlap of highly vulnerable species with the spatial footprint of the redfish trawl fishery (DFO, 2011; Fig. 3). We calculated the proportion of each species' total Eastern Scotian Shelf (ESS) biomass which overlapped with the trawl footprint within each boundary option. We then calculated the vulnerability-weighted mean of all species' overlap (hereafter referred to as VWMO) for each boundary option. We also measured the area within each boundary option. We subsequently ranked boundary options from 1–7 with regards to both overlap and area (1 represents the highest level of overlap or the smallest area, and 7 represents the lowest level of overlap or the largest area). Effectively, lower ranks were more desirable than higher ranks. From a conservation perspective, limitation of overlap of highly vulnerable species and their most significant stressor is desirable; the MPA boundary option that captures the most overlap would limit overlap most effectively when designated. From a socio-economic perspective, limiting the area enclosed in an MPA is desirable; less area to monitor and enforce, and less area taken away from industry (Agardy et al., 2011; Fox et al., 2012; Mascia et al., 2010; Smith et al., 2010). The boundary option(s) that had low ranks for both overlap and area was/were considered the best

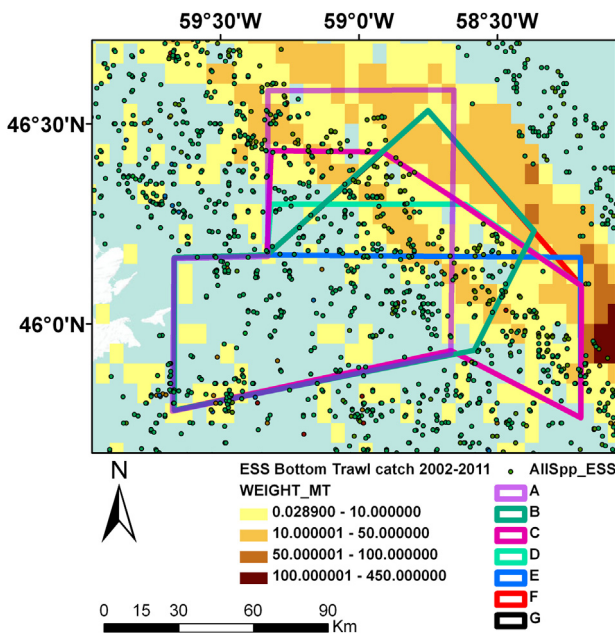


Fig. 3. Foot print of the eastern Scotian Shelf (ESS) redfish bottom trawl fishery (catch biomass in metric tonnes from 2002 to 2011) in and around the St Anns Bank boundary options (A–G). Distribution of key species is shown as green dots (AllSpp_ESS). The green dots are where key species have been found at any biomass.

boundary option(s). This/these option(s) would most successfully protect vulnerable key species from a significant stressor, and limit area cost, relative to the other options.

Spatial data

Species distribution data were collected via DFO summer Research Vessel (RV) surveys. These surveys are conducted annually, between May and September, to characterize fish distribution and abundance, and oceanographic trends over the Scotian Shelf. They follow a stratified random sampling design and record geographic coordinates, depth, temperature, and biomass and number of individuals of each species observed at each sample point (Clark et al., 2010). These data were retrieved via a publically accessible database, the Ocean Biogeographic Information System (obis.org).

Aggregated fishery landings data (sum of all observed landings from 2002–2011 measured in metric tonnes, MT) were obtained from the DFO's Maritime Fishery Information System (MARFIS) (Fig. 3). Greater than 97% of these landings are from the redfish bottom trawl fishery and include bycatch of other depleted ground-fish species (DFO, 2011). These data were geographically referenced and provide a recent 9-year snapshot of the general spatial footprint of the bottom trawl fishery on the ESS. This fishery operates in the St Anns Bank area from May to October and the annual RV survey occurs mostly from May to September. This temporal overlap between stressor and RV survey data was appropriate for our analyses.

Equation 1. Vulnerability as the product of sensitivity and adaptive capacity (both importance weighted sum of factor scores).

$$\text{Vulnerability} = \frac{(S \times A) - V_{\min}}{V_{\max} - V_{\min}} \quad (1)$$

$$S = ((I) \times \text{Diet}) + ((I) \times \text{Natural disturbance}) + ((I) \times \text{Affinity to seabed}) + ((I) \times \text{Larval dispersal}) \quad (2)$$

$$A = ((I) \times \text{Scope for growth}) + ((I) \times \text{Population status}) + ((I) \times \text{Resilience}) + ((I) \times \text{Adult mobility}) \quad (3)$$

where, " $V_{\min}$ " is the minimum possible vulnerability score when all factor scores are 1 (or 0 where appropriate) and " $V_{\max}$ " is the maximum possible vulnerability score obtained when all factors scores are 5 (1).

I is each factor's rating of importance to overall vulnerability, which ranges from 1 to 3; 3 being the most important and 1 being the least important (2 and 3). Each factor's importance was determined via the literature-supported opinion of the authors.

S is the importance-weighted sum of all factors scores for sensitivity (1 and 2).

A is the importance-weighted sum of all factor scores for adaptive capacity (1 and 3).

Results

Species vulnerability

Vulnerability scores among species range from 0.11 to 0.81 (Fig. 4). The average score is 0.38. 47.8% of species are in the low vulnerability category, 13% in the medium vulnerability category, and 39.1% are in the high vulnerability category. These results reflect the distribution of vulnerability scores expected given the Monte Carlo simulation (Fig. 2). Highly vulnerable species are: redfishes (*Sebastes mentella* and *Sebastes fasciatus*), Winter Skate (*Leucoraja ocellata*), Atlantic Wolffish (*Anarhichas lupus*), American Plaice (*Hippoglossoides platessoides*), Thorny Skate (*Amblyraja radiata*), Smooth

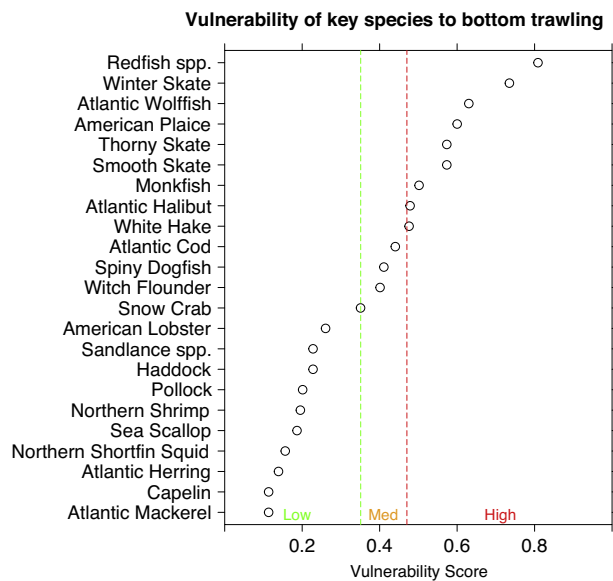


Fig. 4. Vulnerability scores for 23 species. The most vulnerable species are those with scores to the right of the red line. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Skate (*Malacoraja senta*), Monkfish (*Lophius americanus*), Atlantic Halibut (*Hippoglossus hippoglossus*), and White Hake (*Urophycis tenuis*) (Fig. 4). The majority of these species are highly vulnerable to overexploitation due to their life history traits (Musick, 1999), current depleted state (McCusker & Bentzen, 2010), and because they occupy demersal habitat, where bottom trawling has the greatest impact (Jones, 1992).

Ranking boundary options by overlap

The boundary options were ranked from highest to lowest VWMO as such: F, C, B, E, A, D, G (Fig. 5). We therefore gave a rank of 1 to boundary option F, a rank of 2 to boundary option C, a rank of 3 to boundary option B, and so on (Table 2), where a rank of 1 has the highest exposure value and a rank of 7 has the least.

Ranking boundary options by area cost

Boundary options can be ranked from smallest to largest as such: D, G, E, B, A, F, C (Fig. 5). We therefore assigned a rank of 1

Table 2

Exposure and area cost rankings of boundary options based on GIS analyses.

Boundary option	Overlap rank	Area cost rank
A	5	5
B	3	4
C	2	7
D	6	1
E	4	3
F	1	6
G	7	2

to boundary option D, a rank of 2 to boundary option G, and so on (Table 2), where a rank of 1 implies the lowest area cost and a rank of 7 implies the highest area cost.

Finding a balance between cost and benefit

Relative to the other boundary options, B and E offer the best trade-off between protection of highly vulnerable species and limitation of MPA size (Table 2). These boundary options have scores of 3 and 4 (B), and 4 and 3 (E) for overlap and area respectively. Other factors have lower scores for either overlap or area, but high scores for the other. Such boundary options do not offer a balanced trade-off. Boundary options B and E should please both conservation and socio-economic-focused stakeholders. For limitation of overlap, option B is the better option of the two (Table 2). For limitation of area, option E is the better option. However, the difference between these option for either overlap or area is minute (only one rank either way).

Discussion

We created a decision-support tool to facilitate discussion of no-take boundaries for potential Marine Protected Areas during government/stakeholder consultation processes. We developed a flexible vulnerability assessment and demonstrated its use by assessing the vulnerability of 23 key species to their most significant stressor in the St Anns Bank area, bottom trawling. The combination of this vulnerability assessment and area/spatial overlap analyses allows stakeholders to discuss compromises for each proposed boundary option. As a common goal of MPAs is to protect key and overexploited species (Agardy, 1994; Fox et al., 2012), it is prudent to explore an MPAs potential for protecting highly vulnerable key and overexploited species from their most influential stressor. Indeed, assessing species-stressor interactions for the

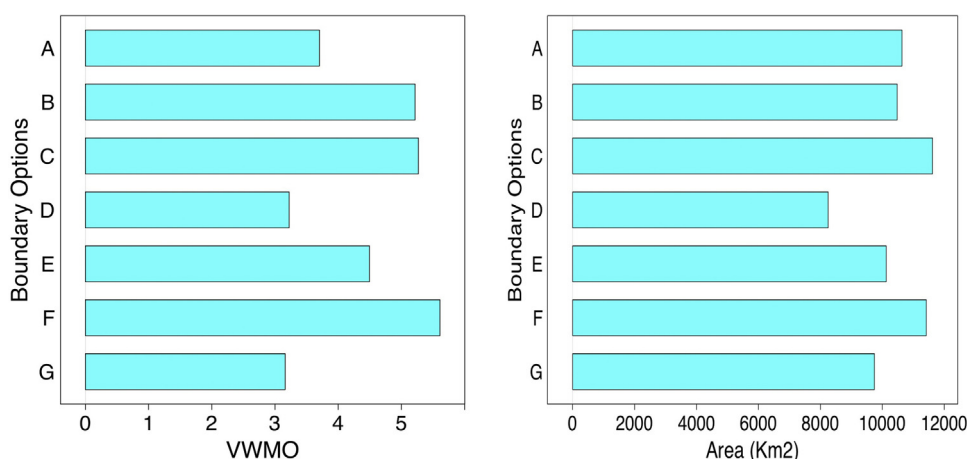


Fig. 5. Left: Vulnerability-weighted mean of highly vulnerable species overlap with bottom trawling (VWMO, measured as percent of total ESS biomass) within each boundary option; Right: Area (km²) contained within each boundary option.

most influential and dominant stressor can be just as effective as assessing all species-stressor interactions (Foden et al., 2011).

The ultimate goal of MPAs is to improve ecosystem health and productivity, which can benefit fishers in the long term (Halpern & Warner, 2002; Hastings & Botsford, 2003; Higgins et al., 2008; Lester & Halpern, 2008; Lubchenco et al., 2003). For this reason, it is important that the smallest boundary option not be chosen simply to reduce socio-economic cost. In the case of the St Anns Bank MPA, it is also important to limit the impact of the most significant stressor, bottom trawling, on highly vulnerable key species. The best compromise is to choose the boundary option that would best serve vulnerable key species, and therefore the ecosystem, at the smallest size possible. A ranking system as provided here (Table 2) can be used to objectively identify this/these boundary option(s). For the St Anns Bank MPA, boundary option B or boundary option E would offer the best compromise (Fig. 5).

Conclusion

Stakeholder consultation in the MPA boundary design stage, although necessary for building trust and ensuring compliance (Gleason et al., 2010; Mangi and Austen, 2008; Young et al., 2013) can be a lengthy process. When multiple boundary options are proposed as a result of differing stakeholder expectations, and when ecological boundaries are “fuzzy”, it can be helpful to have a decision-support tool to objectively evaluate trade-offs among those options (Kendall et al., 2008). To date, very few tools have been proposed for this purpose. Our flexible methodology can provide an objective perspective to help facilitate discussion and foster agreement among stakeholders. This method adds to the resources available to managers in the final stages of MPA planning. The greater the number of tools available, the faster managers and stakeholders can agree on final boundaries for MPAs (e.g. Kendall et al., 2008). This will increase our ability to meet CBD goals for 2020.

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