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A review of information to assist in defining thresholds for SIAs on VMEs

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A review of information to assist in defining thresholds for significant adverse impact on vulnerable marine ecosystems

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1. Purpose

The purpose of this paper is to help identify gaps and uncertainties associated with operationalising the definition of significant adverse impacts (SAIs) that can be used to improve SPRFMO's current approaches for assessing SAIs to vulnerable marine ecosystems (VMEs). This work has been undertaken as part of the Scientific Committee (SC) multiannual workplan task to "Explore thresholds for "significant" adverse impact (SAI) for VMEs at different spatial scales, and understanding [of] knowledge gaps and uncertainties.". This report is also intended to support a related task in the SC multiannual workplan to develop a multi-spatial scale risk-based approach to assess encounters with VME indicator taxa.

2. Background

The South Pacific Regional Fisheries Management Organisation (SPRFMO) Convention requires the SPRFMO Commission to ensure the long-term sustainability of fishery resources and promote the objectives of responsible utilisation, including measures to prevent significant adverse impacts (SAIs) on vulnerable marine ecosystems (VMEs). The impetus for Regional Fisheries Management Organisations/Agreements (hereafter referred to as RFMOs) to prevent SAIs on VMEs by bottom fisheries originated with United Nations General Assembly (UNGA) Resolution 61/105, which calls upon RFMOs:

> *83 (a) To assess, on the basis of the best available scientific information, whether individual bottom fishing activities would have significant adverse impacts on vulnerable marine ecosystems, and to ensure that if it is assessed that these activities would have significant adverse impacts, they are managed to prevent such impacts, or not authorized to proceed;*

This resolution (and subsequent UNGA resolutions 64/72; 66/68; 71/12; 77/118) did not define VMEs, but referred to them as *"vulnerable marine ecosystems, including seamounts, hydrothermal vents and cold-water corals"*, nor did they define SAIs. Subsequently, the Food and Agriculture Organisation (FAO) International Guidelines for the Management of Deep-sea Fisheries in the High Seas (hereafter referred to as the Deep-Sea Fisheries Guidelines), adopted in August 2008, have provided more specific guidance on the characteristics which should be used to identify VMEs, and a definition of SAI and the factors that should be considered when considering the scale and significance of an impact (numbering from the guidelines, FAO 2009):

> ***5.2 Identifying vulnerable marine ecosystems and assessing significant adverse impacts***
>
> *42. A marine ecosystem should be classified as vulnerable based on the characteristics that it possesses. The following list of characteristics should be used as criteria in the identification of VMEs:*
>
> *i. Uniqueness or rarity – an area or ecosystem that is unique or that contains rare species whose loss could not be compensated for by similar areas or ecosystems. These include:*
>
> - *habitats that contain endemic species;*

- *habitats of rare, threatened or endangered species that occur only in discrete areas; or*
- *nurseries or discrete feeding, breeding, or spawning areas.*

ii. Functional significance of the habitat – discrete areas or habitats that are necessary for the survival, function, spawning/reproduction or recovery of fish stocks, particular life-history stages (eg, nursery grounds or rearing areas), or of rare, threatened or endangered marine species.

iii. Fragility – an ecosystem that is highly susceptible to degradation by anthropogenic activities.

iv. Life-history traits of component species that make recovery difficult – ecosystems that are characterized by populations or assemblages of species with one or more of the following characteristics:

- *slow growth rates;*
- *late age of maturity;*
- *low or unpredictable recruitment; or*
- *long-lived.*

v. Structural complexity – an ecosystem that is characterized by complex physical structures created by significant concentrations of biotic and abiotic features. In these ecosystems, ecological processes are usually highly dependent on these structured systems. Further, such ecosystems often have high diversity, which is dependent on the structuring organisms.

Where site-specific information is lacking, other information that is relevant to inferring the likely presence of vulnerable populations, communities and habitats should be used.

3.3 Significant adverse impacts

17. Significant adverse impacts are those that compromise ecosystem integrity (i.e. ecosystem structure or function) in a manner that: (i) impairs the ability of affected populations to replace themselves; (ii) degrades the long-term natural productivity of habitats; or (iii) causes, on more than a temporary basis, significant loss of species richness, habitat or community types. Impacts should be evaluated individually, in combination and cumulatively.

18. When determining the scale and significance of an impact, the following six factors should be considered:

i. the intensity or severity of the impact at the specific site being affected;

ii. the spatial extent of the impact relative to the availability of the habitat type affected;

iii. the sensitivity/vulnerability of the ecosystem to the impact;

iv. the ability of an ecosystem to recover from harm, and the rate of such recovery;

v. the extent to which ecosystem functions may be altered by the impact; and

vi. the timing and duration of the impact relative to the period in which a species needs the habitat during one or more of its life-history stages.

19. Temporary impacts are those that are limited in duration and that allow the particular ecosystem to recover over an acceptable time frame. Such time frames should be decided on a case-by-case basis and should be in the order of 5-20 years, taking into account the specific features of the populations and ecosystems.

20. In determining whether an impact is temporary, both the duration and the frequency at which an impact is repeated should be considered. If the interval between the expected disturbance of a habitat is shorter than the recovery time, the impact should be considered more than temporary. In circumstances of limited information, States and RFMOs should apply the precautionary approach in their determinations regarding the nature and duration of impacts.

Despite these written guidelines, and some supporting publications (Rogers et al. 2008, Duran Munoz et al. 2012, Adron et al. 2014, FAO 2016, Morato et al. 2018), perhaps the greatest gap in understanding that impedes the management of bottom fishing impacts on VMEs is how to practically evaluate (i.e., operationalise) the definition of what constitutes a SAI. One important management component of this issue is how to determine and apply a practical measure of the threshold between an acceptable and unacceptable impacted state (Hiddink et al. 2023a). Accordingly, the 2023 Multi-Annual Workplan of SPRFMO's Scientific Committee included the task *Explore thresholds for "significant" adverse impact (SAI) for VMEs at different spatial scales, and understanding [of] knowledge gaps and uncertainties* (COMM10-Report, Annex 6). This report addresses this task through reviewing available information (primarily from RFMOs) on attempts to define thresholds between acceptable and unacceptable impacted states, i.e., a SAI, at different spatial scales, as well as attempts to operationalise the associated concept of VMEs. In so doing, this review will: (1) highlight where RFMOs have used the relationships between the abundance and spatial extent of VME indicator taxa and ecosystem function, and how these relationships relate to what constitutes a VME; and (2) identify available information to understand the biologically relevant spatial scales that can be used to identify what constitutes a SAI. This report is also intended to support a related task in the Multi-Annual Plan; *Developing multi-spatial scale risk-based approach to assess encounters with VME indicator taxa* (COMM10-Report, Annex 6). To that end, the review will identify approaches that can be used with the available information to undertake multi-spatial scale risk-based assessments. Overall, the review aims to help identify gaps and uncertainties associated with operationalising the definition of a SAI that can hopefully be used to improve SPRFMO's current approaches for assessing SAIs to VMEs.

3. Identifying Vulnerable Marine Ecosystems

Different RFMOs use different methods to identify the known or likely presence of VMEs in their convention areas. Variation among these methods is primarily related to data availability for VMEs and/or VME indicator taxa. The VME identification method used will affect how SAIs can be identified, and relatedly the determination of the threshold between an impacted and unimpacted state. Therefore, how SPRFMO, and a selection of other RFMOs, identify VMEs is first reviewed briefly.

3.1 SPRFMO

SPRFMO's method for identifying the likely presence of VMEs is based on the concept of VME indicator taxa, whereby the known or likely presence of populations of a faunal group assessed to have the characteristics of a VME (according to the criteria in the Deep-Sea Fisheries Guidelines; see section 1) is used as an indicator that a VME may exist at that location (e.g., Parker et al. 2009). There are currently 13 VME indicator taxa for the SPRFMO Convention Area (11 "Vulnerable taxa" and 2 "Habitat indicators") (CMM03-2023, Annex 5). The likely presence of a VME is identified via two methods by SPRFMO: (1) through the 'move-on encounter protocol' where the trawl bycatch of VME indicator taxa exceed particular weight thresholds in any one trawl (CMM03-2023, Annex 6A and B), and the subsequent review process; and (2) through predictive mapping of suitable habitat for VME indicator taxa based on records of such taxa from a variety of sources, most of which data are from outside of the SPRFMO area (SC11–DW01_rev1).

The habitat suitability mapping method is the principal means by which the likely presence of VMEs is identified, and thereafter included in a process to design and implement spatial management measures intended to prevent SAIs in the SPRFMO Convention Area. The full range of the modelled habitat suitability index (0-1) for VME indicator taxa is used for the design of the spatial management measures, but these values are transformed and thresholded to assess the effectiveness of the measures/SAIs (see section 3). The encounter method is used as a "back-stop" to prevent SAIs to VMEs in areas open to bottom trawling, to reflect the uncertainty in the effectiveness of spatial management measures that are based on modelled distributions of suitable habitat for VME indicator taxa (SC6-DW09). As such, the encounter threshold weights for VME indicator taxa were set by the 99th and 80th percentiles (single and multiple VME indicator taxa in a trawl tow, respectively) of the cumulative distributions of historical bottom trawl catch weights. Concerns about the catchability of VME indicator taxa in bottom trawls in relation to detecting VMEs and the potential for SAIs, subsequently led to a reduction in the threshold weight for Scleractinia (stony corals) in 2020 (COMM8-Report, CMM03-2020), and Scleractinia (stony corals), Porifera (sponges), and Actinaria (anemones) in 2021 (COMM9-Report, CMM03-2021)..

3.2 Other RFMOs

The methods used by SPRFMO to identify the known or likely presence of VMEs stem from the paucity of data for VMEs/VME indicator taxa in its convention area. In the North Atlantic Fisheries Organisation (NAFO) Convention Area, the amount and density of research trawl survey bycatch data for VME indicator taxa was deemed initially sufficient to use Kernel density analysis and simple data interpolation mapping methods to identify spatial concentrations of VME indicator taxa that were likely to reflect the presence of VMEs (to produce "VME polygons") (e.g., Kenchington et al. 2009, Murillo et al. 2010). Subsequently, species distribution modelling of VME indicator taxa was also undertaken in the NAFO area (e.g., Knudby et al. 2013a,b), which is partly based on data records from fisheries-independent seafloor camera surveys (e.g., Beazley et al. 2013). In addition, the location of likely VME communities/habitat (i.e., sponge grounds) was predicted by modelling the biomass of a VME indicator taxon (i.e., sponge *Geodia* spp*.)* above a weight threshold identified previously as representing "significant concentrations" of "sponge-dominated communities" (Knudby et al. 2013a). The congruence between the Kernel density analysis-based VME polygons and areas of predicted occurrence for VME indicator taxa derived from the species distribution modelling have been examined, and used to modify the VME polygons to eliminate areas where the taxon is not predicted to occur (NAFO 2015, 2020). For spatial management purposes, but not for direct assessments of SAIs, NAFO also identifies specific seamounts as "VME elements" when they have the potential to support VMEs (Murillo et al. 2011, NAFO 2024).

VME-related data are even more sparse in the convention area of the South Indian Ocean Fisheries Agreement (SIOFA) than in the SPRFMO area. Here habitat suitability modelling (but referred to as species distribution modelling) for VME indicator taxa has also been carried out (Ramiro-Sánchez & Leroy 2022). However, concern about data quality has apparently led SIOFA to not use these predictions for individual taxa in their assessment of potential SAIs, but instead they intend to base their assessment on bioregional classifications of VME indicator taxa (i.e., groups of VME indicator taxa) (Ramiro-Sanchez et al. 2023a) (see section 4).

Species distribution modelling has also been recently undertaken for parts of the North Pacific Fisheries Commission (NPFC) Convention Area to identify the likely presence of VME indicator taxa (Chu et al. 2019) and potential VMEs (Warawa et al. 2021, 2022). Warawa et al. (2022) identified likely coral-based VMEs using models of relative abundance for coral VME indicator taxa and a coral density threshold related to associated species riches (based on the approach of Rowden et al. (2020) for operationalising the Deep-Sea Fisheries Guidelines for identifying VMEs). Prior to these recent broad-spatial scale modelling initiatives, attempts to identify VMEs were primarily made at small spatial scales on individual seamount features in the NPFC area from seafloor imagery collected by fishery-independent surveys (e.g., Curtis et al. 2015, Miyamoto and Kiyota 2017).

3.3 ICES

Despite the successful application of the above VME identification methods as a basis to devise management measures implemented to avoid SAIs from bottom trawling, multiple-criteria assessment methods have been proposed as a more robust approach for identifying VMEs (Morato et al. 2018). That is, methods that integrate as much relevant information as possible into the process of identifying the likely occurrence of VMEs. Recently, workshops have been held by a working group of the International Council for the Exploration of the Sea (ICES) to assess and 'benchmark' these methods for application of advice in European regulatory areas, including the North-East Atlantic Fisheries Commission (NEAFC) Convention Area (ICES 2020, 2022). In brief, data for the known occurrence of VMEs ("VME Habitats") or likely occurrence of VMEs ("VME Indicators") from a variety of sources, including from video survey, VME indicator taxa bycatch, and scientific literature are the primary data used (submitted, after quality control, to the ICES VME Database). Given the non-standard data for "VME Indicators", ICES uses a "VME Index" to combine this information, which is also scored for overall likelihood of a VME being present. Data for "VME Elements" that represent important geographic features (from EMODnet dataset of geological and biogenic structures such as banks, coral mounds, mud volcanoes and seamounts) that can provide suitable habitats for VME indicator taxa (FAO 2009), despite the possible absence of sampling records in the area, are also used in combination with the “VME index” to identify VMEs (see Annex 6 of ICES 2022 for detail).

Note that habitat suitability/species distribution models are currently recommended only to be used as supporting information for the identification of the likely presence of VMEs because ICES had not (at the time of the ‘benchmark’) established standards for accepting these outputs in their advisory process. However, standards for the use of these modelling methods to identify the likely occurrence of VMEs have now been developed (ICES 2021), and will likely be included in a future ICES ‘benchmark’ for the occurrence and protection of VMEs.

3.4 Data limitations and uncertainty

It is worth noting that the ‘benchmark’ methods used for ICES advice are supported by a considerable amount of data, infrastructure, and human resource that is not possessed by most RFMOs, including SPRFMO. However, it is worth reiterating, in the context of the identification of SAIs (see section 4), that the UNGA resolutions include acknowledgment of the uncertainty associated with defining where VMEs occur, requiring RFMOs to “identify areas where VMEs are known or likely to occur” (FAO 2009). van Denderen et al. (2022) interpret this as meaning that “full knowledge of the spatial distribution of the VMEs is not a pre-requisite for management action and that there is an expectation to accept Type 1 error [false positive] over Type 2 [false negative] error.”

4. Identifying Significant Adverse Impacts

Recently, Hiddink et al. (2023a) reviewed and considered the challenge of setting thresholds, for use by decision makers, for detecting a change in ecosystem state from “good” to “degraded” or “lost” using ecological indicators (see their conceptual Figure 1; reproduced here also as Figure 1). This challenge equates with the setting and detection of thresholds for identifying SAIs to VMEs using appropriate indicators. However, it is worth noting that an ecosystem being in a “degraded” state doesn’t necessarily mean that it has been impacted by fishing, and furthermore detecting categorical states such as “good” and “degraded” may be difficult if any impacts to an ecosystem are reflected in a gradient of effects rather than in obvious step effects (which would lend themselves more readily to identifying meaningful and operationalisable thresholds). Nonetheless, Hiddink et al. (2023a) summarised ten available approaches for the selection of thresholds between a “good” and “degraded” ecosystem state (see their Table 1; reproduced here in Appendix). Only two approaches define ecologically meaningful “good” state and use data to estimate thresholds quantitatively; categorised as “Natural variation” and “Maintain function” approaches (Hiddink et al. 2023a). Given the data typically available, neither of these two approaches, nor indeed most of the other approaches summarised by Hiddink et al. (2023a) are currently used by RFMOs for setting and detecting thresholds for SAIs (see section 6 for discussion of the two ‘ideal’ approaches).

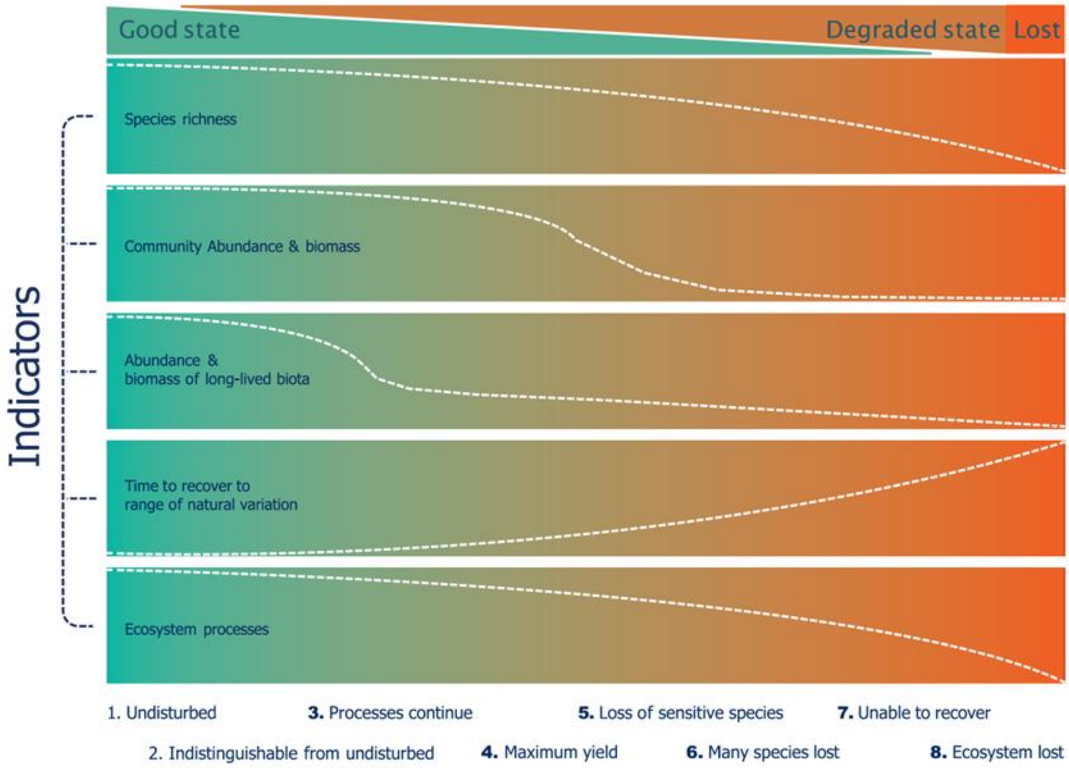

Figure 1 | An undisturbed ecosystem is expected to have many species present, with each species having a natural distribution of abundance and biomass over the different age and size classes, with ecosystem processes at high rates (stage 1). Initially, when pressure from human activity is introduced, the ecosystem is indistinguishable from undisturbed in biodiversity, structure (age, size, species) and function because any changes fall within the range of natural variation (2). When the pressure increases further, it is expected that the largest and oldest individuals in the community will be lost, but all species will be present and ecosystem processes are likely to continue at rates that are near natural (3). Sustainable human exploitation that maximises the yield of targeted species can involve intense activities and is likely to result in widespread changes in size, age, and species composition, with values generally outside the range of natural variation (4). Progressing pressure may result in the loss of the largest and most-long-lived species, resulting in large drops in the total biomass of the community, and large drops in the rates at which ecosystem processes occur (5). With further pressure, more species will be lost, and therefore overall species richness continues to drop, and all parameters are likely to be much lower than in undisturbed systems (6). At some level of pressure, the ecosystem would not be able to recover to its undisturbed state on human time-scales, even if the pressure was totally removed (7), and at the highest levels of pressure, the ecosystem can be considered lost and transformed into another ecosystem altogether (8). ***Reproduced from Hiddink et al. (2023a).***

4.1 SPRFMO

The primary method used by SPRFMO to assess SAIs is what Hiddink et al. (2023a) categorise as a "*Recovery possible*" approach. Hiddink et al. (2023a) note that the "*Recovery possible*" approach does not allow for an ecologically "good" state to be distinguished from a "degraded" state, but instead identifies what "state is already degraded at this level, and only preserves the ability to be good in the future". SPRFMO uses the Relative Benthic Status (RBS) measure (Pitcher et al. 2017), which incorporates the possibility of recovery, to estimate the ecosystem state of VMEs in its area. Although RBS is not used as a 'standard' impact assessment approach worldwide, it has been used widely to assess the impact of fishing to benthic ecosystems since its development as a simple quantitative risk assessment method applicable to data-limited fisheries (e.g., including to areas of seabed around New Zealand and Australia; Pitcher et al. 2022). Estimating RBS (derived by solving the logistic population growth equation for an equilibrium state) requires maps of fishing intensity and habitat or community type, and parameters for impact and recovery rates (which typically come from meta-analyses of experimental studies of towed- or longline-gear impacts). The status of habitats/communities in an assessed region is indicated by the distribution of RBS values within the region or sub-regions (Pitcher et al. 2017). Therefore, the determination and spatial application of RBS allow for four of the six factors from the Deep-Sea Fisheries Guidelines for assessing the scale and significance of an impact to be considered: i.e., "*the intensity or severity of the impact at the specific site being affected*", "*the sensitivity/vulnerability of the ecosystem to the impact*", "*the ability of an ecosystem to recover from harm, and the rate of such recovery*", and "*the spatial extent of the impact relative to the availability of the habitat type affected*" (FAO 2009).

In SPRFMO, historical fishing effort data (as swept area ratio; SAR), habitat suitability data layers for VME indicator taxa (transformed to better reflect the presence of a VME, and account for model uncertainty), and VME indicator taxon-specific depletion and recovery parameters are used to determine RBS, with a focus on application to Fisheries Management Areas (FMAs) (SC11-DW01_rev1). The '**quality threshold**' that is applied to RBS values to denote whether the inferred VMEs in FMAs are protected from SAIs is currently borrowed from the Marine Stewardship Council (MSC), which indicates that "B/K [B=biomass/abundance and K=carrying capacity] of 0.8 should be maintained for VMEs (MSC 2018). That is, the MSC requirement is that "habitats are not impacted beyond the point at which they could recover to 80% (or more) of their unimpacted level within 5-20 years". The MSC specifically acknowledges that "VMEs are generally habitats with slow recovery rates". The origin of the 80% threshold appears to be somewhat subjective ("MSC has nominated the 80% level as a reasonable point at which to expect most of the habitat's structure and function (including abundance and biological diversity) to have been restored, taking into consideration the likely logistic population growth of habitat-forming organisms"), although unspecified reference is made to an example study ("Off the north coast of Australia, several shelf-break VME areas have been damaged but are still there in reduced form and would recover if left undisturbed for several years.") (MSC 2018). The time frame associated with the threshold relates directly to the indication in the Deep-Sea Fisheries Guidelines of what constitutes an acceptable time frame over which an ecosystem can recover from a temporary impact (see paragraph 19 of FAO 2009). It is worth noting briefly here that the application of the MSC threshold to RBS values by SPRFMO is incomplete, and that it doesn't directly assess whether recovery is actually possible within a given timeframe; that would require some sort of additional analyses (e.g., Hiddink et al. 2023b; see also further discussion in section 6). The MSC's threshold is intended to be used, by default, for assessing habitat impacts within the areas controlled by the management regime under which their "Unit of Assessment" operates, which are referred to as "managed areas" (MSC 2018). Often these managed areas are for a particular fish stock, although the application of the habitat impact assessment over a larger or smaller managed area(s) may be justified. That is, in certain cases "it would be reasonable for the assessment [team] to scale up or scale down the "managed area" when determining the appropriate habitat range to consider". MSC further notes that expert judgement can be applied, with a provided rationale, for such a scaling (MSC 2018). In the SPRFMO context, FMAs represent a managed area and are also currently considered "the appropriate scale for assessing the performance of the spatial management areas"

(COMM11–Doc07, SC11-DW01_rev1, CMM03-2023), including using measures of RBS and the MSC threshold to assess SAIs to VMEs.

In 2020, SPRFMO used the MSC threshold for assessing potential SAIs for three spatial scales (the VME indicator taxa modelled area within the Evaluated Area, FMAs, and the Bottom Trawl Management Areas (BTMAs)) (SC8-DW07_rev), but since 2023 the RBS values are reported for the FMA scale only (which includes the BTMAs) (SC11-DW01_rev1). The calculation of the RBS values is part of SPRFMO's spatial management measure process to design and implement open and closed areas to bottom trawling. Specifically, it is used as one of the means by which to assess the "effectiveness" of the spatial closures to prevent SAIs, as part of a periodic impact risk assessment for bottom fishing (SC11-DW01_rev1). Currently, RBS values have only been determined for 7 of the 13 SPRFMO VME indicator taxa (SC11-DW01_rev1).

Since 2022, the potential for individual trawl tows within SPRFMO's BTMAs (i.e., areas open to bottom trawling) to cause SAIs to VMEs is assessed via a review of any reported trawl 'encounters' with potential VMEs. Where an 'encounter' means a fishing trawl catch of VME indicator taxa at or above particular threshold weight levels (see section 3). The encounter is reviewed using the following information: "(i) historical fishing events within 5 nm of the encounter tow, in particular, any previous encounters, and all information on benthic bycatch; (ii) model predictions for all VME indicator taxa; (iii) details of the relevant fishing activity, including the bioregion; and (iv) any other information the Scientific Committee considers relevant." (CMM03-2023). The SPRFMO member state reviewing the encounter for the Scientific Committee uses this information to apply the VME and SAI criteria in the Deep-Sea Fisheries Guidelines (see section 2) to "*verify whether a VME is likely to be present at the encounter area and/or the surrounding area, whether a significant adverse impact has occurred, and the risk of a significant adverse impact occurring in the future.*" (CMM03-2023). In effect, this risk assessment approach aligns with the "*Expert judgement*" approach of Hiddink et al. (2023a). Draft (aka proposed but not formally adopted) guidelines for carrying out this review process have been provided by New Zealand to SPRFMO (SC09-DW08), which were used for the single encounter review to date (SC09-DW09_rev1).

4.2 Other RFMOs

SIOFA has recently developed a "fishing intensity impact index" to assess the potential impact of fishing in bioregions for VME indicator taxa (Ramiro-Sanchez & Leroy 2023). However, no thresholds have currently been devised for this index that could be used to identify SAIs to VMEs represented by these bioregions (i.e., it has not been operationalised for assessing SAIs). The SIOFA report that documents this index notes the "purpose of the index is not to be interpreted as an absolute value – but rather to serve as an index of the evolution of fishing practices in the SIOFA area and explore which bioregions are the most affected by fishing impacts, according to gear type. For example, the index could be used to analyse the effect of fishing management measures, such as how limiting or fostering fishing in a particular area would affect the index of fishing impact for each bioregion." (Ramiro-Sanchez & Leroy 2023).

Like SIOFA, NPFC has also not yet developed methods with measurable objectives for determining the occurrence of SAIs to VMEs, instead it currently proposes to use a relative risk assessment that "does not explicitly refer to impacts to benthic ecosystems, but it provides a quantitative and repeatable measure of the spatially-explicit relative risk of SAIs." (Gasbarro et al. 2022). To date this relative risk assessment has focused only on assessing SAIs to VMEs from long-line fisheries in a portion of the NPFC Convention Area (i.e., the Cobb-Eickelberg seamount chain). To calculate the relative risk of SAIs to areas likely to be VMEs, data layers (1 km x 1 km grid resolution) for the probability of VME indicator occurrence (from the habitat suitability models) and the cumulative fishing impact area (from historical long-line footprint) were used to group grid cells into categories. The optimal number of categories for each data type were identified using both unsupervised analytical procedures and subjective decision making. Four categories were identified for each data type. Grid cells with both a fishing footprint and a VME indicator occurrence probability within their respective highest categories were assessed to be at 'high' risk of SAI. Grid cells with a fishing footprint and VME indicator probability within the lowest two categories were deemed 'low' risk, and all other cells 'medium' risk (Gasborro et al. 2022). This proposed relative risk assessment method, as a basis for making decisions around the design and implementation of management measures to prevent SAIs to VMEs, has not yet been adopted by NPFC.

Neither of recently developed approaches for assessing SAIs by NPFC and SIOFA are operationalised to the point that they align with one of the approaches, categorised by Hiddink et al. (2023a), for defining and determining a threshold between a "good" and "degraded" ecosystem state.

4.3 NAFO and ICES

The initial method developed by NAFO to evaluate SAIs to VMEs is what Hiddink et al. (2023a) categorise as a "*Tipping point*" approach, which often defines a threshold based on the pressure on an ecosystem rather than the state of the ecosystem. Hiddink et al. (2023a) note that the "*Tipping point*" approach doesn't always distinguish an ecologically "good" versus "degraded" state, but is an objective way to define a quantitative threshold where the "rate of harm per unit disturbance goes up". NAFO's approach to detecting SAIs (NAFO 2015) was adapted by ICES (ICES 2020) and included in the 'benchmark' methods provided by ICES for the occurrence and protection of VMEs (ICES 2022). This approach is based on the assertion that "frequently fished areas of VME will tend to support lower biomass of VME indicator taxa compared to areas of the same VME that have been fished less frequently." (NAFO 2015). Accordingly, the characteristics of a cumulative VME indicator taxa biomass plot allow the identification of the level of fishing intensity above which no new biomass is observed (i.e., the VME is likely impacted – degraded or lost). Nominally, any areas of the seabed subject to fishing intensity above a threshold value of 95% of the cumulative VME indicator biomass are likely to have experienced a SAI (Figure 2). This fishing intensity-based potential SAI threshold was determined for several VME indicator taxa in the NAFO area where trawl bycatch data for VME indicator taxa is relatively abundant. The threshold values ranged from 0.5 hours fishing km^{-2} for sea pens to 0.1 hours fishing km^{-2} for large gorgonian corals (NAFO 2016).

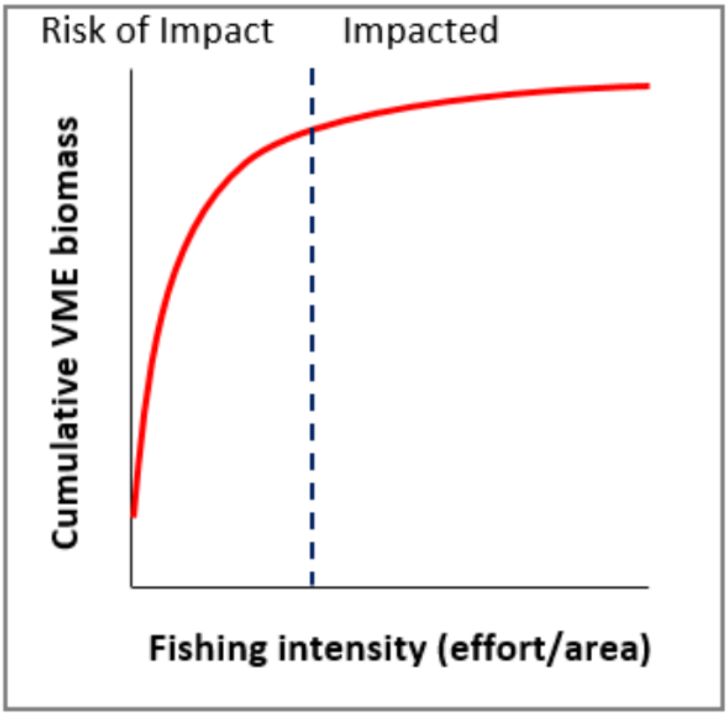

Figure 2 | Theoretical relationship between cumulative VME biomass and fishing intensity used to identify threshold of impact to VMEs. ***Reproduced from NAFO (2015).***

ICES (2020) considered that because VME indicator taxa of the same type show similar functional characteristics across the North Atlantic, it may be expected that a threshold value derived from studies in the NAFO area could be applied to European areas (including the NEAFC area). The potential SAI threshold for what is considered (at the time) the least sensitive VME indicator taxa (i.e., Pennatulacea or seapens) was conservatively adopted as appropriate for all assessments under the precautionary principle. The threshold value was converted from fishing intensity measured as hours fishing km^{-2} used by NAFO to SAR for use by ICES (i.e., 0.5 hours fishing km^{-2} to 0.43 SAR; ICES 2020). Under the ICES 'benchmark', this fishing intensity-based '**quality threshold'** can be applied to measures of the "VME Index" and "VME Habitat" to assess SAIs (see Annex 6 of ICES 2022 for detail of the procedure). The SAIs are assessed at the spatial scale of a "C-square" (0.05° x 0.05° latitude/longitude) within an "ecoregion" (see section 5) (ICES 2022). Note that currently fishing data are insufficient for the reliable use of the fishing intensity-based threshold for advice from ICES to NEAFC.

NAFO updated their SAI assessment in 2021, based on information provided by its Scientific Council Working Group on Ecosystem Science and Assessment (NAFO 2020). The fishing intensity-based impact thresholds for each VME indicator taxon, derived from all available data across the NAFO Convention Area (NAFO 2020) (Figure 3), is then applied to a mean fishing intensity data layer to classify cells (1 km x 1km grid resolution) as either "Impacted" (above the threshold) or "At Risk of Impact" (below the threshold). The total area and biomass of each VME polygon is then calculated in each impact class, across the "Fishing Footprint" polygon area (and some adjacent areas where parts of fishery closures extend beyond the footprint boundary) within the NAFO Convention Area. All cells (centre of) within fishery closures are classified as "Protected" area.

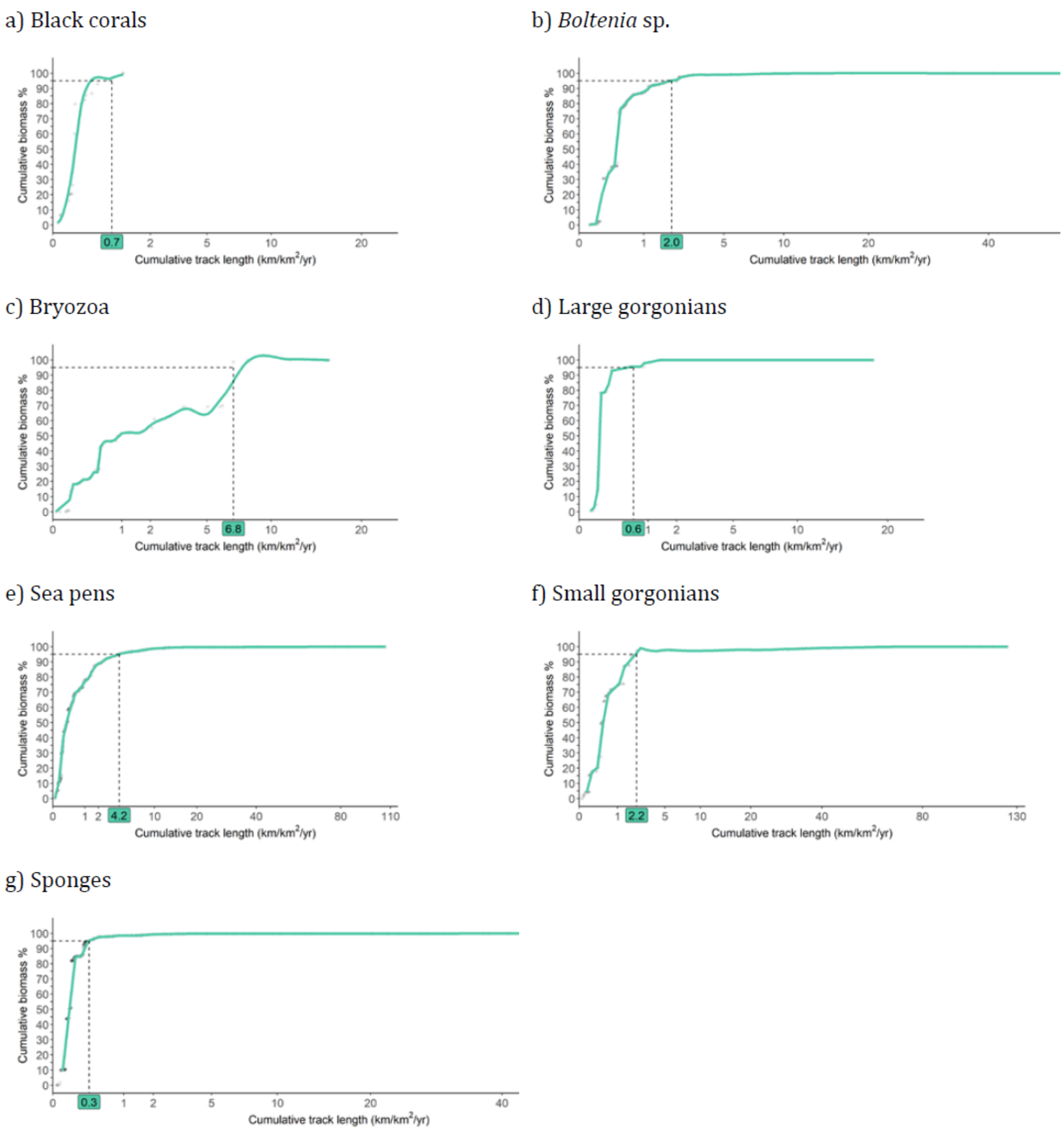

Figure 3 | Cumulative plots of VME biomass (for different types of VME indicator taxa) against fishing intensity. Threshold values for presumed impact by fishing effort to VMEs are identified at the 95% cumulative biomass level. ***Reproduced from NAFO (2020).***

NAFO now also uses additional metrics to assess SAIs (NAFO 2020): "Index of VME sensitivity", which is the inverse of the VME impact threshold; "Fishing stability index", the proportion of the total fishing effort for each VME polygon associated with cells repeatedly fished above the VME impact threshold over a 10-year period; "Overlapping VME index", which is the proportion of VME polygon area and biomass overlapping with 2 or more VME indicator taxa inside fishery closures; "Index of VME fragmentation/proximity", which is the spatial extent (size) and location (distance) of VME polygons in relation to their neighbours of the same VME indicator taxon; and "Number of overlapping [ecosystem] functions in unprotected VME" polygon areas. The latter metric is derived from spatial polygons of ecosystem function (nutrient cycling, bioturbation, habitat provision) derived from Kernel density analysis of fish and invertebrate biomass data for parts of the NAFO area, which are overlapped with VME polygons. Data limits mean that this metric can only currently be applied to assess "Impacted" and "At Risk of Impact" areas within the Fishing Footprint. The thresholds to distinguish between these two areas was determined by assessing the accumulation of functional biomass response curves against fishing intensity (Figure 4) (NAFO 2020).

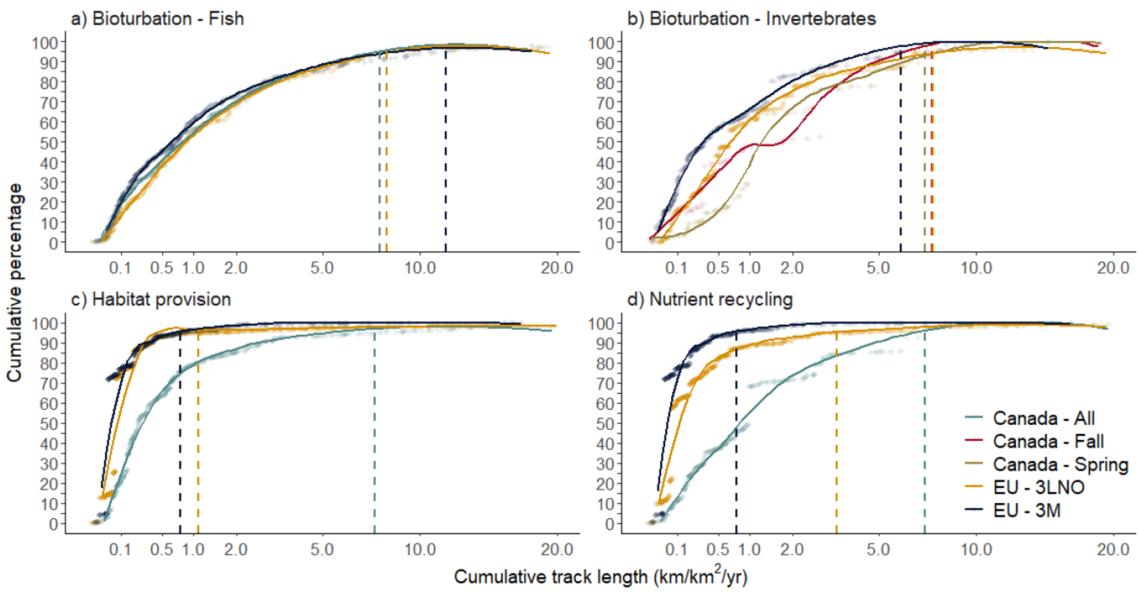

Figure 4 | Biomass accumulation curves for fish and invertebrate taxa (including VME indicator taxa) assessed for each biological data survey. (a) all fish bioturbators, (b) all invertebrate bioturbators, (c) all invertebrate taxa providing habitat structure and (d) all invertebrate taxa providing a nutrient recycling function. Line colour differentiates data source by the different surveys. Dotted lines indicate the fishing intensity (in km/km2/year) where 95% of total functional biomass has been accumulated. ***Reproduced from NAFO (2020).***

In total, NAFO currently uses eight metrics to assess SAIs, covering the six factors in the Deep-Sea Fisheries Guidelines for assessing SAIs. Metrics which relate to the first two factors are "considered to be of greater importance (and hence influence) in determining the overall assessment of SAI", and the overall weighting of the assessment is based primarily on the "Protected" area SAI metric score. The scores assigned to all other SAI metrics providing some overall context and confidence in the assessment of the primary SAI risk score. Therefore, by "focussing the result of the assessment on the protected VME biomass status, the assessment is essentially one which determines the risk of SAI occurring as opposed to the assessment of whether or not SAI has occurred." (NAFO 2020). The thresholds used to define Low, Intermediate, and High risk of SAI for the "Protected" areas were >60%, 30-60%, and <30% of VME polygon area/biomass, respectively (NAFO 2020). These thresholds were those retained from an earlier review of VMEs (NAFO 2019). These risk thresholds are **'spatial extent thresholds'** but they are directly equated with the quality status of VMEs (i.e., they become in effect **'quality thresholds'**). Low risk equates to Adequate-Good VME status, Intermediate risk to Limited-Incomplete VME status, and High risk to Inadequate-Poor VME status (NAFO 2020). By implication, 70% of likely VME habitat would need to be protected to avoid High risk of SAI (i.e., maintain Limited to Good VME status). Thresholds for the remaining SAI metrics were defined by dividing the range of the metric values by 3 (with rounding to the nearest whole number). The "Overall SAI Risk" assessment against all metrics is by expert assessment (NAFO 2020).

4.4 Data limitations and spatial scale

As already noted above, RFMOs don't typically have access to the sorts of data that allow for the more 'ideal' approaches, outlined by Hiddink et al. (2023), that would be suitable for defining and detecting SAIs to VMEs. Neither is there currently data available that would allow for most RFMOs to address quantitatively all six factors that should be considered when assessing the scale and significance of an impact to VMEs. Obtaining such data are likely to continue to be beyond their resources, so instead RFMOs are obliged to take a precautionary approach and/or use what methods suit their available data, and work to refine and improve these "best available" approaches (FAO 2009). Among this effort to design better approaches, SPRFMO has sought over time to resolve the question of the most appropriate spatial scale or scales to identify the likely presence of VMEs and detect and avoid SAIs. Therefore, the spatial scale-relevant aspects of identifying VMEs and assessing SAIs (see above) are reviewed further below.

5. Spatial Scale

The definition and recognition of ecosystems and impacts is fundamentally scale dependent, including the spatial scale. The relationship between spatial scale and the identification of VMEs is bound up in the definition of VMEs as vulnerable "populations, communities and habitats" (FAO 2009). These ecological units can exist across a range of spatial scales from metres to 1000s of kilometres. Impacts can also operate across a range of spatial scales, and the significance of an impact depends upon the relative scale at which it is operating upon a population, community or habitat (Hall 1994).

5.1 Practical/methodological spatial scales for identifying VMEs

Despite attempts by the scientific community to clarify and refine the definition of VMEs and the most appropriate spatial scale for identifying VMEs (e.g., Watling & Auster 2017, Baco et al. 2023), VMEs or potential VMEs are typically identified by RFMOs at a practical/methodological scale rather than an ecological spatial scale. That is, ranging from the smallest sampling unit or grid cell used for collecting or collating VME-related data to the extent of the known/modelled area for VMEs/VME indicator taxa within the convention area of the RFMO. For example, the draft SPRFMO encounter review process is based on data that potentially indicates the presence of a VME at the spatial scale of up to a single bottom trawl (overall mean area across nations, bottom types, and methods = 0.78 km^2) (SC09-DW08). The ICES 'benchmark' methods for identifying VMEs are based on data collated at the spatial scale of a "C-Square", which is a nominal 5×5 km grid cell or 25 km^2 at the equator (ICES 2022). These cells may be combined during the data compilation process to identify spatial contiguous areas of known or likely VMEs, which can vary considerably in size. SPRFMO primarily identifies the potential extent of VMEs based on the full modelled distribution of suitable habitat for VME indicator taxa (between 200 and 3000 m water depth) across a sub-set of the Convention Area (the "Evaluated Area"). These habitat suitability predictions cover areas of ~4.5 million km^2. While these habitat suitability data layers are produced at a resolution of 1 km grid cells, no attempt is currently made by SPRFMO to identify the spatial scales of discrete suitable habitat for VME indicator taxa within the extent of these modelled layers across the Evaluated Area (accepting that the FMAs are considered to pragmatically represent a "ecological relevant spatial scale" within the extent of these layers; (COMM11–Doc07, SC11-DW01_rev1, CMM03-2023). However, it is possible to identify ecologically relevant spatial scales of a VME *habitat* and/or *community* (beyond the minimum sample/grid cell size), and for a group of VME indicator taxa *populations*.

5.2 Ecological spatial scales for identifying VMEs

Spatially distinct VME communities and VME habitat can be directly observed and/or revealed by association/classification data analysis from video transects of the seafloor, and therefore the scale of their lateral extent along the transects can be determined. For example, Du Preez et al. (2020) observed dense assemblages of VME indicator taxa at scales of 100s of metres on Cobb Seamount in the NPFC Convention Area. Modelled data can be used to better resolve the 2-dimensional extent of VME communities and habitats. For example, the NAFO kernel density analysis and interpolated mapping approach have identified spatially discrete biomass concentrations of VME indicator taxa, deemed to be synonymous with VMEs (Kenchington et al. 2014), which range in size from 9687 km^2 (sponge VME) to 0.2 km^2 (large gorgonian coral VME) (Wang et al. 2024). Using abundance models for a structure-forming stony coral and an abundance threshold at which a "high diversity" of other invertebrates is apparently "dependent on the structuring organism" (i.e., reference to criteria (v) for characterising VMEs; FAO 2009) (Rowden et al. 2020), the VME habitat 'stony coral reef' on the Louisville Seamount Chain in the SPRFMO Convention Area has been estimated to range in size from 0.06 – 2.81 ha (based on a modelled grid cell of 25 m x 25 m; SC9-DW07). However, the spatial scales for this particular VME habitat have been estimated to be larger elsewhere in the South Pacific (0.02 – 1.16 km^2, Williams et al. 2020). However, it is worth noting that the coral reef estimates of Williams et al. (2020) are based on simple extrapolation from the VME habitat status distributions observed between adjacent radial seafloor camera transects, or around single transects, and the adjacent 50 m isobaths. Recently Warawa et al. (2021, 2022) used a similar approach to Rowden et al. (2020) for identifying coral-based VMEs (communities/ habitat) and the area they occupy on seamounts in part of the NPFC Convention Area. These authors did not report the size range of the individual predicted VME habitats, but the total area predicted varied depending on the analysis method used (3,137 km^2 compared to 1,542 km^2; Warawa et al. (2021, 2022), and not surprisingly on the grid size resolution of the model (when subsequently a smaller grid size was used - 500 m x 500 m compared to 1 km x 1 km; Warawa pers comm).

Regarding populations of VME indicator taxa that could represent VMEs, the spatial scale of their ecological coherence can be estimated via their genetic structure, predicted larval dispersal capability among populations, or through bioregional analysis. For example, Zeng et al. (2017), using microsatellite variation, detected genetic structure in populations of the stony coral *Solenosmilia* *variabilis* between biogeographic provinces. This comparison between provinces included sites on the Louisville Seamount Chain and in the south Tasman Sea in the SPRFMO Convention Area, separated by ~3000 km (shortest distance) (Zeng et al. 2017). Within the same biogeographic province, these authors found no genetic structure differences (also using mitochondrial and nuclear DNA sequence variation) between populations of the same species at sites on the Louisville Seamount Chain and the Northwest Challenger Plateau, separated by ~1000 km (shortest distance). Kenchington et al. (2019) used biophysical models and habitat suitability models for VME indicator taxa to assess the connectivity among areas closed to fishing to protect VMEs in a portion of the NAFO Convention Area. Their study predicted that while structural connectivity was possible among most closed areas 10s – 100s km apart (i.e., larvae could potentially drift among these sites), functional connectivity (i.e., larval drift among the most likely potential source populations of VME indicator taxa) was in some cases predicted to be more spatially limited (Kenchington et al. (2019). This finding illustrates the importance of having an understanding of the likely source of propagules when trying to determine connectivity, or lack of, as a proxy for identifying ecologically relevant spatial scales. The recent inclusion of habitat patch metrics (fragmentation/proximity) by NAFO in their SAI assessment (NAFO 2020, see above), and the work of Wang et al. (2024) on quantifying the effects of fragmentation on the connectivity networks for VMEs, also points to the usefulness of connectivity studies to identify relevant ecological spatial scales, which may vary among VME indicator taxa. Recently, SIOFA has identified bioregions for combined populations of VME indicator taxa (at family, genus, and species level) using classification/modelling analyses (Ramiro-Sanchez et al. 2023a). Using a 'group first-then predict' approach (Ramiro-Sánchez & Leroy 2022) 3 bioregions have been identified across the Indian Ocean, in which are nested 8 sub-bioregions. The latter were used for SIOFA's trial use of a spatial management decision-support tool, and within the SIOFA Convention Area these sub-bioregions range in spatial scale from ~130,700 to 5,900,400 km^2 (Ramiro-Sanchez et al. 2023b).

The Deep-Sea Fisheries Guidelines note that "topographical, hydrophysical or geological features" (including seamounts) can "potentially support the species groups or communities" that have "characteristics consistent with possible VMEs" (FAO 2009). These features (which may range considerably in areal size, e.g., seamounts in the New Zealand region range from < 0.5 km^2 to 34,700 km^2; Rowden et al. 2005) have been protected by some RFMOs as "VME Elements" even though "detecting the presence of an element itself is not sufficient to identify a VME" (FAO 2009). For example, seamounts have been included in the latest NAFO closed areas (NAFO 2024), and these collectively protect 363,301 km^2, ranging in size from 145 km^2 (2J East 2) to 183,663 km^2 (New England Seamounts) , and their total area represents 13.4% of the NAFO Regulatory Area.

5.3 Spatial scales for assessing SAIs

SPRMFO

Understanding the spatial scale of a VME is important for understanding the potential scale or extent of an impact. The fundamental relationship between spatial scale and a SAI is recognised in the Deep-Sea Fisheries Guidelines (see section 1). Of the six factors listed in the guidelines that should be considered for determining a SAI, the second refers explicitly to spatial scale: i.e., "the spatial extent of the impact" should be assessed "relative to the availability of the habitat type affected". As already noted above, the 2023 Australian-New Zealand BFIA for SPRFMO presents measures of RBS, and compares these to the 80% MSC-based SAI threshold, at the FMA (including the area of the BTMA within) spatial scale. This spatial area is considered by SPRFMO as "the appropriate scale for assessing the "performance of the spatial management areas" (paragraph 73 of the SC08-Report) and "likely to be a more biologically appropriate compared with larger scales" (COMM11–Doc07, SC11-DW01_rev1, CMM03-2023). Assessing at the FMA scale allows only for an assessment of SAIs to potential VMEs within a subset (i.e., the FMA area) of the data available for suitable habitat for VME indicator taxa in the SPRFMO area (i.e., the modelled area within the Evaluated Area). The previous 2020 BFIA reported on RBS values at three spatial scales; the Evaluated Area, FMA, and BTMA (SC08-DW07_rev1). However, although the amount of bottom trawling contact and suitable habitat for VME indicator taxa was calculated that year for bioregions within the Evaluated Area, RBS metrics were not determined for this spatial scale. As noted above, bioregions can be considered a means by which to distinguish the ecologically relevant spatial extent of a population of VME indicator taxa (or its suitable habitat). Therefore, the calculation of RBS at the bioregional scale would, arguably, allow for a more closely aligned assessment of the second of the Deep-Sea Fisheries Guidelines' factors for determining SAIs. Undoubtedly, determining RBS for bioregions as well as the Evaluated Area, FMAs, and BTMAs, would allow for greater understanding of the relative spatial scale of potential SAIs using the MSC-based threshold.

The draft (aka proposed but not formally adopted) encounter review process for assessing the potential for SAIs in the SPRFMO area (see section 4) does include consideration of "a broad range of scales in the context of the best available science on, inter alia, the distribution of VME across spatial scales." (SC09-DW08). For its assessment, the SPRFMO Scientific Committee is provided with details of the fishing activity, "including the bioregion" in which it occurs (SC09-DW08). For the single encounter review to date, undertaken by New Zealand (SC09-DW09_rev1), data were first assessed to determine if a VME had been encountered (as opposed to detecting the presence of VME indicator taxa above a particular weight threshold, which in itself does not constitute a VME). This assessment was carried out at a range of spatial scales, including the site of the potential impact (i.e., with reference to the first of the six factors listed in the Deep-Sea Fisheries Guidelines for assessing SAIs: "When determining the scale and significance of an impact", the "intensity or severity of the impact at the specific site being affected" should be considered). The encounter review; deemed that the encounter constituted evidence of the presence of a VME at the encounter trawl tow scale, but not at the scale of the encounter area (1 NM closed area around the encounter trawl track), nor a 5 NM buffer area around the encounter, nor of the area of 'trawl tow cluster'. The latter spatial scale was used as a proxy for the size of the discrete areas (patches) that the particular type of VME being reviewed (Gorgonian coral reef/garden) could potentially occupy (based on literature estimates). The five trawl tow cluster areas identified for the data assessment ranged in size from 4–15 km^2 (see Figure 9, SC09-DW09_rev1). For the SAI assessment, the review considered each of the six SAI factors in the Deep-Sea Fisheries Guidelines (see section 2). Here the 'impact' was only considered significant at the scale of the individual encounter trawl tow, with the likelihood of a SAI at the rest of the other assessed scales (at the encounter area, 5NM buffer area, and tow clusters) deemed unlikely. The 'availability of the habitat type affected' was defined as the modelled suitable habitat for the VME indicator taxon of interest in the FMA (outside of the open areas, i.e., BTMAs) in which the encounter took place. Using this spatial scale approach, the spatial extent of the encounter impact relative to the availability of the VME indicator taxa habitat affected was found to be very small. Overall, when considering all six factors and a range of spatial scales, the risk of future SAIs occurring was deemed by New Zealand to be "moderate at the spatial scale of the encounter area, low at the spatial scale of the Bottom Trawl

Management Area and low at the spatial scale of the FMA." (SC09-DW09_rev1). The review of this assessment by the Scientific Committee modified the "risk of SAI resulting from reopening the encounter area to be high at the spatial scale of the encounter area", but agreed that the risk was "low at the spatial scale of the FMA" (SC09-Report). The Scientific Committee also noted that, at the time, "while the appropriate scale to assess and manage impacts on VMEs has not been defined in SPRFMO, the scale of the Fishery Management Areas is likely to be a more biologically appropriate scale at which to assess and manage SAIs on VMEs than larger scales" (SC09-Report), and presumably, smaller scales.

Other RFMOs

The latest SAI assessment by NAFO considers all six SAI factors, including therefore the ones that relate to assessing the relative spatial scale of any impact (NAFO 2020). As already noted above, of the 8 metrics that NAFO uses to assess SAIs to VMEs, most emphasis is placed on the metric that determines the risk to VMEs in "Protected" areas. That is, placing most emphasis on assessing the risk of SAI to VME habitat that is within the total area of all fishery closures, which is equivalent (in a SPRFMO context) to assessing SAI for the total area of all FMAs, excluding the areas occupied by the BTMAs. The current NAFO "Protected" areas (NAFO 2024) within the Fishing Footprint, cover a combined area of 27,849 km^2, compared to approximately 2.1 million km^2 for the total area of FMAs (excluding BTMAs). The average area of a SPRFMO FMA (excluding BTMAs) is 209,000 km^2 (range 10,404 – 1,233,963 km^2). The other metrics used by NAFO to assess SAIs overall are applied at a range of spatial scales up to the area covered by the Fishing Footprint (120, 048 km^2) (and some adjacent areas where fishery closures extend beyond the footprint boundary), which together total 138,051 km^2. This spatial scale represents the area in which any VME habitat has or can potentially be affected by the type of fishing assessed within the NAFO Convention Area, and where risk of SAI to VMEs is therefore likely to be higher. No similar spatial polygon for the fishing footprint has been formally delineated for the SPRFMO Convention Area. However, for comparison the combined area of all BTMAs, where fishing can currently take place within the Evaluated Area of the SPRFMO Convention, is 60,231 km^2.

As already previously noted, SIOFA has recently included the use of bioregions for VME indicator taxa in its recent assessment of the applicability of approaches to identify SAIs to VMEs in its convention area (Ramiro-Sanchez & Leroy 2023). This approach provides an estimate of the potential for SAI, measured as "fishing Intensity impact index", at two spatial scales – bioregion and sub-bioregion. As such, this approach satisfies the spatial scale-specific factor for determining SAIs included in the Deep-Sea Fisheries Guidelines. The relative risk assessment for SAIs proposed for use by NPFC does not currently specifically state that it aims to address this spatial scale factor. However, examples of its application indicate visual mapping of SAI risk for 1 km^2 grid cells within individual seamounts (Figure 5), as well as being part of the assessment of the overall risk across the assessed area (Cobb-Eickelberg seamounts) within the NPFC Convention Area (Gasborro et al. 2022). As such, the approach in effect includes consideration of "the spatial extent of the impact relative to the availability of the habitat type affected" (FAO 2009).

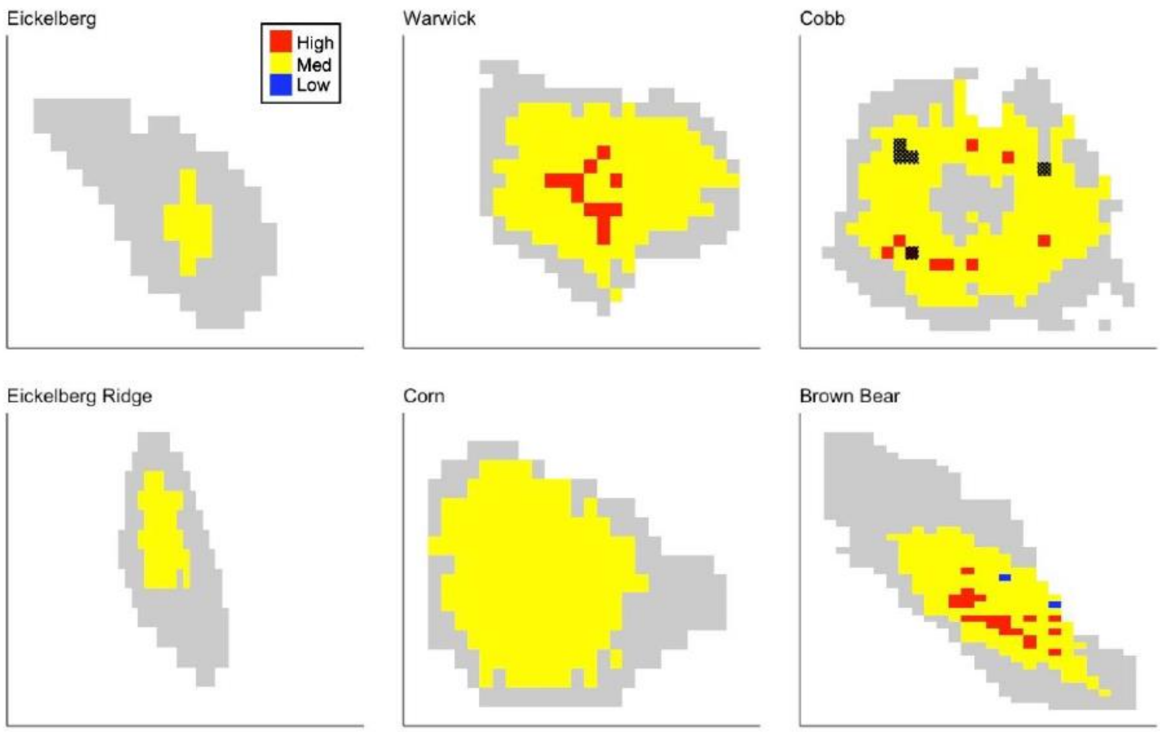

Figure 5 | Relative risk of SAIs at Cobb-Eickelberg seamounts within the NPFC Convention Area. Red, yellow, and blue areas represent high, medium, and low relative risk categories, respectively. Grey cells denote seamount areas with no fishing footprint from 2006-2021 while hatched cells on Cobb seamount show grid cells with VMEs identified by Warawa et al. (2022). ***Reproduced from Gasborro et al. (2022).***

ICES

The ICES 'benchmark' process proposed for assessing and managing SAIs to VMEs, first assesses SAIs at the same spatial scale at which it determines the known or likely presence of a VME. That is, the 'C-square' (see above). These small spatial scale assessments of SAIs are cumulatively assessed within a risk assessment framework, for several risk and management outcome scenarios, for each ecoregion in the ICES area (or "advice requester" area, e.g., NEAFC Convention area). This approach includes an assessment of the total available "VME polygons" within each ecoregion (ICES 2022). In effect, this approach addresses the Deep-Sea Fisheries Guidelines spatial scale-specific factor for determining SAIs, in a similar post-hoc way to the spatial management "effectiveness" assessment process of SPRFMO, but does so by including the more obviously ecologically relevant spatial scale of ecoregion rather than FMA (accepting that FMAs may be a useful proxy for an ecologically relevant scale, however, this has not yet been demonstrated through any analysis). ICES considers ecoregions as the "most relevant spatial units to synthesise the evidence for [an] ecosystem approach". The ICES ecoregions are different from the ICES fishing management areas, and the use of "ecoregions enhances ICES ability to research ecosystem and social dynamics and translate those findings into consolidated ecosystem-based advice" (ICES 2023). The ecoregion boundaries have changed over time, and have taken into consideration political, social, economic, and management divisions (ICES 2023). While the largest spatial scale that NAFO currently reports its SAI risk assessment relates to its Fishing Footprint (see above) (NAFO 2020), it has previously identified three spatial scales as relevant for the development for ecosystem summaries and ecosystem-level management plans; bioregion, Ecosystem Production Units (EPUs), and ecoregion. The EPU is considered "more appropriate for integrated fisheries management plans because it defines a major geographical subunit within a Bioregion characterised by distinct productivity and a reasonably well defined major marine community/food web system" (NAFO 2022).

6. Discussion

The above review illustrates that RFMOs have used a variety of approaches to identify VMEs and assess SAIs to VMEs. However, according to the review of potentially available approaches for distinguishing between 'good' and 'degraded' ecosystem state by Hiddink et al. (2023a), none of these approaches can be judged to be 'ideal' for identifying thresholds for assessing SAIs.

6.1 Use of 'ideal' approaches for assessing SAIs

Hiddink et al. (2023a) note that the "most objective method for setting a good-state threshold is based on maintaining the state within the range of natural variation in undisturbed systems.". Data to support such a type of assessment is not currently available to SPRFMO, nor will it realistically be in the future. Even RFMOs with access to considerably more data than SPRFMO are not using the "*Natural variation*" approach (*sensu* Hiddink et al. 2023a). To utilise such an approach, reliable and <u>standardised</u> time-series data for state indicators (e.g., VME indicator taxa biomass) would be required <u>from undisturbed as well as disturbed areas</u> (unless it was possible to partition out the effect of disturbance). Furthermore, the utilisation of the approach requires regular re-collection of such data (i.e., monitoring). The requirement for these sorts of data is difficult to satisfy, even from within EEZs (e.g., New Zealand; Bowden et al. 2015) let alone the high seas.

The "*Maintain function*" approach is the second of the two approaches identified by Hiddink et al (2023a) as ecologically and quantitatively based. This approach would arguably be the best approach to use for assessing SAIs to VMEs, given that the five characteristics of VMEs listed by the Deep-Sea Fisheries Guidelines relate directly or indirectly to ecosystem functions (e.g., "ii. functional significance of habitat – discrete areas or habitats that are necessary for the survival, function, spawning/reproduction or recovery of fish stocks, particular life-history stages (e.g., nursery grounds or rearing area), or of rare, threatened or endangered species"), and that preventing SAIs is intended to safeguard these functions (FAO 2009). The "*Maintain function*" approach is based on establishing a state-function relationship and using an objective way to identify a threshold for what is 'good' function. Typically, this threshold is estimated from the position of an inflection point in the state-function relationship in space (or time) (Hiddink et al. 2023a). For example, Rowden et al. (2020) established a coral density threshold, that was equivalent to about 30% coral cover (state), at which coral provide structural habitat to support high levels of diversity (function). This work was designed to illustrate how to operationalise the definition of VME habitat in the SPRFMO Convention Area, but results of this research are equally relevant for operationalising the definition of what constitutes an SAI (i.e., "Significant adverse impacts are those that compromise ecosystem integrity (i.e. ecosystem structure or function).", FAO 2009). That is, any reduction in coral cover caused by fishing can be used to assess "the extent to which ecosystem functions may be altered by the impact" (SAI factor (v), FAO 2009), and the identified threshold in the state-function relationship can be used to determine whether a SAI has occurred (i.e., a transition between a 'good' and 'degraded' state). While the

"*Maintain function*" approach is conceptually simple, as well as relevant for assessing SAIs to VMEs, a great deal of targeted and potentially region-specific research will be necessary to establish the state-function relationships for those ecosystem functions associated with the characterisation criteria for a VME. This research will need to be carried out at a variety of spatial scales, unless the relationship is scale free – which might sometimes be the case (see Rowden et al. 2020). The temporal dynamics of the recovery of the state-function relationship from fishing impacts will also need to be understood, in order to consider another SAI factor listed in the Deep-Sea Fisheries Guidelines (i.e., SAI factor (iv), "the ability of an ecosystem to recover from harm, and the rate of such recovery", FAO 2009). Given the requirement for all of the aforementioned research, the "*Maintain Function*" approach, while attractive is currently impractical for application by SPRFMO. However, it is worth noting here again that NAFO has developed a SAI metric which assesses the overlap between benthic ecosystem function in general (beyond that which might be provided by VME indicator taxa) and the occurrence of VMEs (see above). Further work to extend, improve and refine this approach (among others) is ongoing by NAFO (2020, 2021), particularly since they note that "the question of quantifying what loss of VME biomass constitutes a significant impact, will depend on further developments in quantifying the VME functions" (NAFO 2020). Therefore, it would be sensible for SPRFMO to remain aware of these research developments and the potential for their application as a "*Maintain Function*" approach in the future.

6. 2 Improving the current "*Recovery possible*" approach

SPRFMO currently uses the RBS metric, combined with the MSC-based threshold, to assess the state of possible VMEs. This "*Recovery possible*" approach (*sensu* Hiddink et al. 2023a) is a pragmatic attempt to determine the overall state of possible VMEs in managed areas (i.e., FMAs) that are both open to bottom trawling (i.e., BTMAs) and closed to trawling (i.e., the rest of the FMA), and thereby the effectiveness of the spatial management measures to avoid SAIs within the managed area. However, there are several gaps and uncertainties associated with this form of current assessment. These include:

(1) no calculation of RBS values for six VME indicator taxa;

(2) the use of transformed habitat suitability index values as a means to represent the likelihood of the presence of a VME, rather than using abundance-based model outputs that are likely to be more useful for identifying VMEs;

(3) limiting the calculation of RBS for FMAs only, when assessing RBS at multiple spatial scales would be more informative, particularly including the use of an ecologically relevant bioregional scale that would arguably allow for closer alignment with one of the spatial factors for assessing SAIs that is included in the Deep-Sea Fisheries Guidelines;

(4) the use of the RBS-integral depletion (d) and recovery (R) values for VME indicator taxa that are derived mainly from studies in shallow water, i.e., values which may not be appropriate for deepwater representatives of the taxa;

(5) using a form of RBS which does not take into account changes in fishing history over time (i.e., uses a long-term average of SAR), when a form of RBS exists that can account for the temporally dynamic nature of fishing (e.g., annual variability);

(6) the use of a subjective 80% threshold for SAIs borrowed from the MSC, a threshold which would ideally be objectively determined by research studies; and

(7) the lack of an evaluation of the recovery time part of the MSC habitat impact criterion, which is aligned with the Deep-Sea Fisheries Guidelines for assessing SAIs to VMEs.

With regard to point **(1)**, SPRFMO is currently (2024) engaged in addressing the gap in the calculation of RBS values for all VME indicator taxa (SC12 -DWXX, and see also below). For point **(2)**, efforts are also underway to improve the reliability of the abundance-based models for VME indicator taxa that were produced recently (SC11–DW07_rev1), so that they can potentially be adopted for use by SPRFMO for a range of purposes, including the re-calculation of RBS values in the future. For point **(3)**, SPRFMO could, in addition to calculating RBS at the FMA scale, return to also calculating RBS directly for BTMAs (to allow assessment of SAIs where fishing occurs) and bioregions (to allow assessment of SAIs at a known ecologically-relevant spatial scale). For the latter, it would be useful to re-evaluate the appropriateness of the previous bioregion schemes used (SC08-DW07_rev1), or use approaches for identifying SPRFMO-specific bioregions for each individual VME indicator taxon (e.g., Victorero et al. 2023) or multiple taxa (e.g., Ramiro-Sanchez et al. 2023), and/or genetic and/or biophysical modelling approaches (e.g., Abecasis et al. 2023, Díaz-Cabrera et al. 2023) for identifying ecologically relevant spatial scales for the SAI assessment (or representative protection assessment). Such work is now in progress (SC12 DWXX). Undertaking assessments of SAIs at smaller ecologically relevant spatial scales, i.e., those that are typically associated with discrete community or habitat VMEs (such as coral reefs and sponge grounds) would be reliant upon seafloor video and bathymetric surveys across FMAs (including within BTMAs). Such surveys are carried out by other RFMOs, including for site-scale assessments of SAIs (e.g., NPFC, see references above). However, there are currently no plans to undertake such surveys in the SPRFMO Convention Area, although their desirability has been noted for several other reasons in a SPRFMO context (e.g., for VME indicator taxa abundance modelling (SC11–DW07_rev1) and catchability assessments (SC10-DW04)).

Regarding point **(4)**, obtaining the RBS-integral depletion (d) and recovery (R) values for VME indicator taxa derived only from deepwater studies should be an obvious future goal to reduce uncertainty around the current assessment of SAIs, although it would to be costly to achieve by direct experimental research. However, deep-sea values for d and R could be obtained from model simulations (Rowden et al. 2024), or expert-derived/modified values could be used instead. Expert-derived values of d and R have now been obtained as part of recent work to update RBS values for SPRFMO (SC12 DWXX). For point **(5)**, the temporally dynamic approach for determining RBS, so-called dRBS, requires a timeseries of fishing distribution and effort (as SAR) for each timestep (Pitcher et al. 2015). The inclusion of such data in the calculation of dRBS means that it doesn't rely, as simple RBS does, on assumptions of the continuation of fishing in the same locations and intensity into the future (Pitcher et al. 2017). Therefore, dRBS can provide an estimate of VME indicator taxa abundance/biomass up to the end of the timeseries of SAR, rather than an estimate assuming continued fishing at "equilibrium" (Pitcher al al. 2015, 2017). In addition, including consideration of finer temporal scale impacts may be important when fishing occurs heavily for only a few years and not consistently across the timeseries. Heavy fishing over just a few years, particularly at the beginning of a fishing history, has been shown to have a disproportionate impact upon the depletion and recovery of benthic fauna (Clark & Tittensor 2010, Goode et al. in prep.). Such an impact may be masked when using long-term average data for SAR. Annual bottom trawling data are available for the

SPRFMO Convention Area, and therefore it is possible to calculate dRBS, and thereby gain the aforementioned advantages over RBS for assessing SAIs to VMEs. Work to determine dRBS values for all SPRFMO VME indicator taxa has recently been completed (SC12 DWXX). For point **(6)**, evaluating whether it is possible to replace the MSC-based 80% SAI threshold with one derived from empirical study would be a very useful. There could potentially be some suitable historical data (from SPRFMO and/or New Zealand Australian EEZs) to try deriving, refining, or confirming such a threshold, and further exploration of these data and suitable analysis methods would be worthwhile pursuing. In the meantime, it is worth noting that Hiddink et al. (2023a) observed that preliminary "time-series analyses of marine seabed community biomass suggest this threshold [for 'good' state] is located between 54 and 79% of the undisturbed state." (based on six European studies across a range of habitats and spatial scales; see Table S3 in Hiddink et al. 2023a). Furthermore, a report by NAFO (2020) noted (citing Korpinen et al. 2013) that studies done in the context of the EU Habitat Directive had "considered that impacts of 25% or more of the total habitat area as the criteria for deeming those habitats to be in unfavorable conditions", and "Therefore, by definition, if 76% of the habitat were protected, it would be in favorable condition". NAFO itself, as noted previously, considers a value of 70% "Protected" VME indicator taxa biomass to indicate avoidance of a "High risk" of SAI, and thereby the maintenance of Limited-Good VME status (NAFO 2020). Note that the aforementioned % values are **'spatial extent thresholds'**, however, they each represent an inferred '**quality threshold'** for benthic status. Given these threshold values, it is possible to argue that the MSC-based 80% quality status threshold used by SPRFMO for application to values of RBS is reasonable, and on the conservative side of the range of similar currently known values (accepting that these values are not directly comparable).

Finally, to point **(7)**, evaluations should be conducted as to whether VME habitat recovery is possible within an appropriate time frame, so as to demonstrate likelihood of a temporary impact. That is, apply the second part of the MSC criterion for assessing impacts to VMEs, the part that relates to the Deep-Sea Fisheries Guidelines definition of a SAI being "more than temporary" (see above; FAO 2009). Such an evaluation can be carried out by modelling the RBS response of each VME indicator taxon following the theoretical cessation of fishing within each FMA (and/or other designations of managed area) (see Figure 6 for hypothetical examples from Hiddink et al. (2023b)). Hiddink et al. (2023b) outline a procedure for such an evaluation, which they have made available for applicants of MSC certification. Recent work for calculating dRBS for SPRFMO included future modelling of VME indicator taxa status up to 20 years following cessation of fishing to predict recovery (SC12 – DWXX). However, some additional work is likely needed to decide on suitable recovery timeframes for this modelling with dRBS. While the Deep-Sea Fisheries Guidelines indicate temporary impacts are "in the order of 5-20 years", they also note that "such time frames should be decided on a case-by-case basis" and take "into account the specific features of the populations and ecosystems" to recover (see above; FAO 2009). For example, NAFO conducted agent-based modelling to assess recover times for seapens, and found that following the theoretical cessation of fishing, recovery of an area could take 50 years (given initial depletion of 60% seapen abundance) (NAFO 2019). Furthermore, it is worth noting that climate change effects could influence recovery times of some VME indicator taxa, and this should also be considered when determining an appropriate temporary impact time-frame for use with any impact level threshold (Levin et al. 2020).

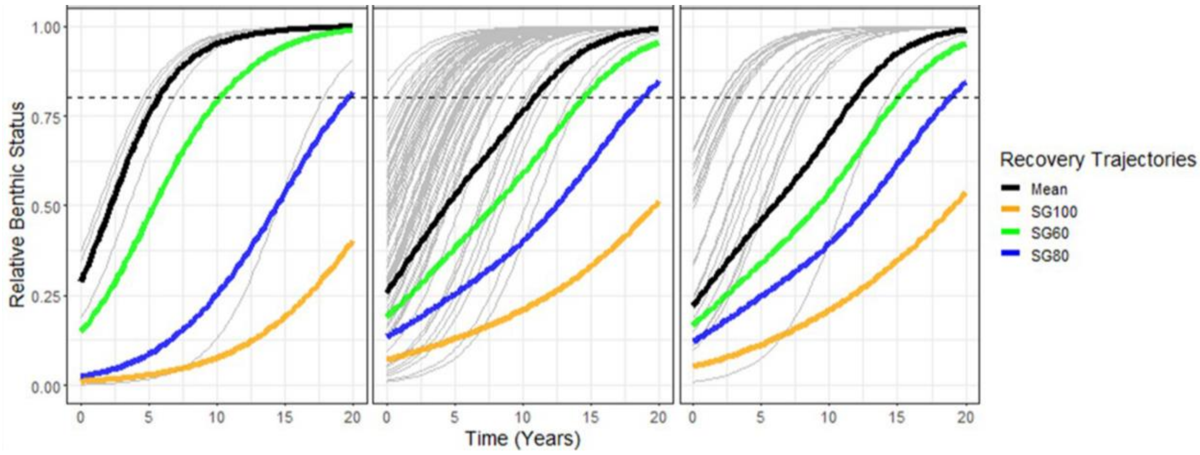

Figure 6 | Hypothetical recovery trajectories from fishing for three habitats. The thin grey lines indicate the recovery trajectories of individual cells using the median R estimate, while the solid-coloured lines represent the RBS and the recovery trajectories estimated using the lower percentiles of the R distribution. The dashed line indicates the recovery threshold of RBS = 0.8. Where a solid-coloured line crosses this threshold within 20 years, this Scoring Guidepost (SG) is achieved. In this example each habitat scores SG80. ***Reproduced from Hiddink et al. (2023b).***

6.3 Adopting a multi-criteria and multi-spatial scale risk assessment approach

The draft encounter review method used by SPRFMO to assess if SAIs have occurred when trawl bycatch meets or exceeds the possible VME detection thresholds (SC09-DW09_rev1), is an example of a multi-criteria assessment, or a multiple lines of evidence approach (Morato et al. 2018). Such a multi-criteria approach for assessments is widely used in environmental decision-making processes (e.g., Carvalho et al., 2020, Turschwell et al., 2023, Vanegas-Cantarero et al, 2022) and is included in the NAFO and ICES 'benchmark' processes for identifying VMEs and assessing SAIs (NAFO 2020, ICES 2022). However, the draft SPRFMO encounter review method, although it does include quantitative analyses, is overall an "*Expert judgement*" risk assessment approach (*sensu* Hiddink et al. 2023a). There is an opportunity to introduce more structure and rigor to this form of SAI assessment, particularly given that SPRFMO's encounter review process is currently only a proposed methodology. For example, the method could be re-defined and re-configured as a formal ecological risk assessment (ERA), adopting standards, and using multiple approaches that have been tried and tested for comparable situations. For example, the "Ecological Risk Assessment for the Effects of Fishing" (ERAEF) framework developed in Australia (Hobday et al. 2011). Components of the ERAEF framework, e.g., a Productivity-Susceptibility Analysis (PSA), are already included in ERAs for fish undertaken by New Zealand and Australia as part of their periodic Cumulative Bottom Fishing Impact Assessments (BFIA) for SPRFMO (SC11-DW01_rev1), yet this framework is not used to assess the risk of SAIs to VMEs. It is worth noting that the ERAEF framework has already been used to assess risk to seamounts (i.e., VME features or elements) on the Chatham Rise within the New Zealand EEZ (Clark et al. 2011), and the PSA component of the ERAEF has been trailed as an ERA methodology for assessing the risk of impacts from fishing to deepwater corals (i.e., VME indicator taxa) in the New Zealand EEZ (Clark et al. 2014). Work on further developments of an ERA for corals using multiple risk assessment approaches is currently underway for New Zealand's Department of Conservation (project INT2022-04, CSP Annual Plan 2022_24). These latter developments include the trialling of dRBS (see above) and a probabilistic approach for assessing risk using Bayesian Network modelling methods (Kaikkonen et al. 2020). The use of modelling approaches can more easily allow for the carrying out of whole-ecosystem assessments.

Indeed, a recent review of the impact assessments carried out by RFMOs recommended that modelling approaches be developed that assess whole ecosystems rather than focus impact assessments on just VME indicator taxa (DOSI 2022, Kaikkonen et al. 2024). Should efforts be made in these directions to improve the encounter review process for SAIs (under the Multi-Annual Workplan task *Develop an encounter review standard*), then it would be worth considering extending the encounter review process to become part of a broader multi-criteria and multi-spatial scale risk assessment approach for SAIs to VMEs in the SPRFMO Convention Area as a whole (thereby expanding the Multi-Annual Workplan task *Developing multi-spatial scale risk-based approach to assess encounters with VME indicator taxa*). Such a broader ERA could be part of the BFIAs that member states provide to SPRFMO. This risk assessment can include those data compiled as part of the annual review of bycatch data, and thereby address another task currently on the Multi-Annual Workplan (*Development of a process to review all recent and historical benthic bycatch data*; SC10-DW03, SC11-DW10). However, it is worth reiterating here the longstanding uncertainty-related concern about the use of VME indicator taxa bycatch data for identifying VMEs and subsequent assessments of SAIs to these VMEs (SC09-Report). This concern rests primarily in the lack of specific understanding about the trawl catchability of VME indicator taxa (SC10-DW04), and further work to address this issue is clearly required by SPRFMO and other RFMOs. This need is acknowledged in SPRMO's latest Multi-Annual Workplan, although it is currently unclear whether gaining such an understanding is financially feasible to achieve (*Assess the feasibility and develop a research programme within the SPRFMO Convention Area to allow the determination of taxon-specific estimates of catchability for VME indicator taxa*; SC10-DW04).

For a new multi-criteria and multi-spatial scale risk assessment, there is potential value in considering the adoption of approaches for identifying VMEs and assessing SAIs that are included in the ICES 'benchmark' process, and those of other RFMOs. For example, use of a "VME index" and potentially greater use of "VME Elements" as potential indicators of VMEs (see above; NAFO 2020, ICES 2022). However, it is worth noting that ICES is committed to periodically reviewing and updating its 'benchmark', and highlighted that future work should aim to address concerns about the use of its "VME Index" under-representing certain types of VME, as well as the need to develop a robust "confidence" measure for this index. In addition, updating of the "VME Elements" information is required, along with further refinement of the methods to identify these features (ICES 2022). The relative risk categorization procedure proposed for use by NPFC (see above, Gasbarro et al. 2022) could potentially be used by SPRFMO. This type of risk assessment is relatively simple, but based mainly on quantitative analyses, and may be an attractive option given that it will likely be used by another RFMO and could be co-developed further.

On the other hand, there will likely be methodologies used by other RFMOs that may not be appropriate or applicable for the SPRFMO situation. For example, the use of fishing intensity-based thresholds for identifying SAIs (i.e., "*Tipping point*" approach *sensu* Hiddink et al. 2023a) that are used by NAFO and are also included in the ICES 'benchmark' (see above). Primarily, the concern with using this fishing intensity-based approach in a SPRFMO context relates to the amount of bycatch data that are available in each area of assessment. Exploratory evaluation of this method using available SPRMO data demonstrated that establishing a robust cumulative biomass curve, on which this method relies, was difficult or impossible for some FMAs. Furthermore, ICES itself is recommending that they continue to explore the sensitivities around the application of the fishing intensity-based SAI threshold, given that the underlying assumptions will influence the level of precautionary protection applied. In addition, work is recommended to explore the relationship between fishing effort and the potential for VMEs to recover from harm (ICES 2022). Given all of the aforementioned, it doesn't seem currently sensible to adopt this simple fishing intensity-based threshold approach for assessing SAIs in the SPRFMO Convention Area. However, it would be worth SPRFMO keeping a close 'watch' on developments by NAFO and ICES on this "*Tipping point*" approach given its potential utility (including links to a "*Maintain function*" approach), and because it can be a useful component of a multi-criteria and multi-spatial scale risk assessment approach to assessing SAIs across all six factors listed in the Deep-Sea Fisheries Guidelines. Finally, it is worth noting that multi-criteria risk assessments also have the benefit of allowing evaluations of the similarity/dissimilarity of the results among different methods and approaches, and their merits or otherwise for assessing SAIs to VMEs.

7. Conclusions

Most RFMOs have adopted spatial management measures to prevent SAIs to VMEs, however, their assessment of SAIs has previously been identified as not very advanced (FAO 2016, Bell et al. 2019, SC07-DW18). However, attempts to improve assessment of SAIs are taking place across RFMOs, and these are occurring within structured and scientifically-guided processes (e.g., the ICES 'benchmark' process, Working Groups and Scientific Committees of individual RFMOs). Despite these efforts, few of the individual RFMO approaches used to assess SAIs incorporate all of the SAI factors included in the Deep-Sea Fisheries Guidelines (FAO 2009), primarily because of data limitations as well as a lack of basic understanding about the ecology of VME indicator species. Furthermore, there remains a deal of uncertainty around the underpinning data and ecological validity of the thresholding methods that are currently used by some RFMOs to distinguish SAIs to VMEs. None of the thresholding approaches used by SPRFMO, NAFO or ICES comply with the "*Natural variation*" or "*Maintain function*" approaches identified by Hiddink et al. (2023a) as ideal for monitoring 'good' versus 'degraded' ecosystem state. The "*Maintain function*" is arguably the more appropriate approach for assessing SAIs to VMEs, given the definition of VMEs and the ecosystem functions of their characterizing populations, communities and habitats, and the aim to maintain these functions by preventing SAIs from fishing (FAO 2009). Until such a time that RFMOs can obtain appropriate data to utilise the aforementioned approaches, multi-criteria and multi-spatial scale ecological risk assessments are a justifiable and pragmatic approach for assessing SAIs to VMEs. To adopt the use of such a type of risk assessment, SPRFMO would need to engage in work to improve and modify its current approaches for assessing SAIs, and could include these in a formal ERA framework for VMEs, alongside those for fish, in the regular BFIAs.

8. Recommendations

It is recommended that the Scientific Committee:

- **Notes**:
  - That a review has been undertaken to identify gaps and uncertainties associated with operationalising the definition of significant adverse impacts (SAI).
- **Agrees**:
  - That the "Maintain function" approach identified by Hiddink et al. (2023a) would arguably be the best single approach for assessing SAIs to VMEs, but SPRFMO currently lacks the appropriate data to implement this approach;
  - That evaluating whether it is possible to replace the MSC-based 80% SAI threshold with one derived from an empirical study tailored to the dRBS approach would be very useful.

- **Recommends**:
  - That a multi-criteria and multi-spatial scale risk assessment approach is developed for inclusion in the SPRFMO Encounter Review Standard to improve SPRFMO's current approaches for assessing SAIs to VMEs;
  - That to improve the utility of the dRBS approach for assessing SAIs on VMEs, additional data is collected from areas of interest to management (where feasible) and existing data is processed (where available) to better inform VME indicator taxa abundance model development and validation;
  - Developing other metrics for assessing SAIs on VMEs that can be used in a multi-criteria and multi-spatial scale risk assessment approach to help inform the SPRFMO encounter review process and benthic impact risk assessments.

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11. Appendix: Table 1 **reproduced from Hiddink et al. (2023a)**

Table 1. An overview of approaches that can be used to choose thresholds that define good ecosystem state.

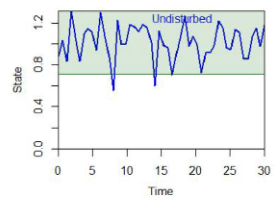
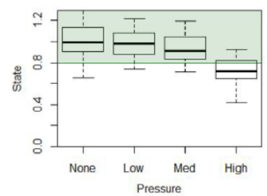
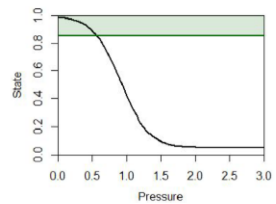
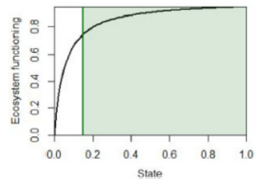
Basis for threshold	Distinguishes ecologically good vs. degraded?	Quantitative from data?	Advantages	Disadvantages	How?	Examples	Graphical illustration. The solid green line indicates the threshold for good state and the green polygon indicates the region above threshold (where present)
Natural variation: State is within the range of pressure-free variation	Yes	Yes	Easy to understand, and ecologically meaningful. Objective way of defining quantitative threshold	Require time-series of undisturbed populations/communities	Defined using time-series of undisturbed populations/communities. Use detrended time-series if long-term trends exist	Baseline from undisturbed systems. E.g. no-take zones that have had long-term protection (Babcock *et al.*, 2010)	
Detectable change: State that is just statistically detectably different from the baseline	No	Yes	Objective way of defining quantitative threshold	Statistically detectable change does not equate to change to degraded state and relates to power of analysis. Lack of pristine reference areas	Estimated from the state–pressure relationship in space or time	Response of reef building fauna to pot fishing, reductions of abundance of >50% (Rees *et al.*, 2021)	
Tipping point: Breakpoint in statistical relationship between state and pressure	Often, but not always, as degraded state could occur before the tipping point	Yes	Objective way of defining a quantitative threshold. Defines where the rate of harm per unit disturbance goes up	Tipping points are rarely statistically detectable and often do not exist (Fu *et al.*, 2020, Hillebrand *et al.*, 2020)	Estimated from position of inflection point on the state–pressure relationship. Often defined as a pressure rather than a state	Defined by pressure rather than state, swept-area-ratio by trawls at threshold between 0.7 and 4.6 (Lambert *et al.*, 2017) and 0.08 to 0.60 (Jac *et al.*, 2020)	
Maintain function: Maintaining ecosystem function at levels without pressure	Yes	Yes	Objective way of defining quantitative threshold, that defines good state based on the ability to maintain ecosystem functioning	Relies on establishing a state–function relationship and still relies on setting a threshold for what function is good	Estimated from position of inflection point on the function–state relationship in space or time	30% forest cover needed to maintain forest bird community integrity (Banks-Leite *et al.*, 2014)	

Table 1. Continued

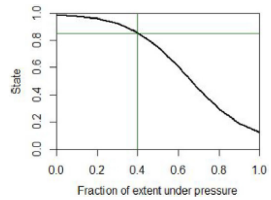
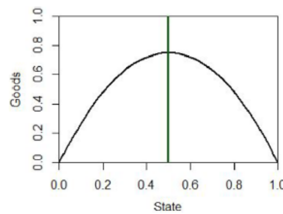
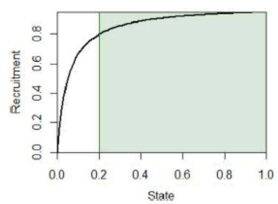
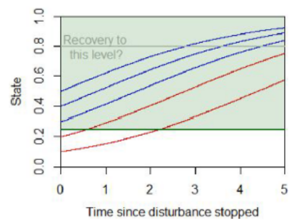
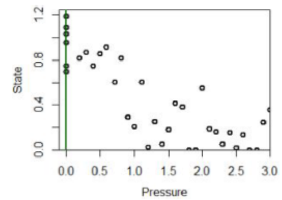
Basis for threshold	Distinguishes ecologically good vs. degraded?	Quantitative from data?	Advantages	Disadvantages	How?	Examples	Graphical illustration. The solid green line indicates the threshold for good state and the green polygon indicates the region above threshold (where present)
Trade-off: Find the point at which the increase in conservation benefits decreases relative to the decrease in the delivery of goods, as an optimal solution	No, it defines a societally desirable state	Yes	Explicitly evaluates what is important to society	The definition of good state will vary depending on the intensity and distribution of human pressures. Good state defined as a point estimate rather than a threshold	Estimated as a position on the goods delivery–state relationship that balances the importance of the state and human use	Lester *et al*. (2013) established the optimum combination of fishing, wave energy, and property values	
Maximize ecosystem goods: Maximize the amount of goods that are produced for human use	No	Yes	The maximum is a clearly justified target	Maximizing human benefits is not equivalent to achieving good environmental state. The target is not a threshold above which the state is good, but a point estimate	Usually based on population dynamics of exploited species in fisheries	Maximum sustainable yield maximizes fisheries landings. $B/K \sim 0.4$ for single benthic invertebrate stocks (Ricard *et al*., 2012)	
Avoid collapse: Prevent collapse of ability to withstand (or recover from) pressure, safeguarding future uses	No	Yes	State below this level is clearly not good	The ability to maintain recruitment does not imply that ecosystem state is maintained, and state is likely to be low	Identification of the point at which recruitment collapses on the population size –recruitment relationships	B_{lim} as used in fish stock management, as the point where the stock–recruitment relationship flattens out. Assumed to be around $B/K = 0.2$ for many commercially fished species (Ricard *et al*., 2012)	
Recovery possible: Recovery of state possible within a specified time once the pressure is removed, safeguarding future use	No	Yes	State below this level is clearly not good	State is already degraded at this level, and only preserves the ability to be good in the future	Modelling of recovery based on empirical recovery rate estimates	The MSC certification standard specifies that recovery to $B/K = 0.8$ should be possible in 5–20 years for broad habitats, and that $B/K = 0.8$ should be maintained for VMEs as their recovery is likely to be very slow (Marine Stewardship Council, 2018)	

Table 1. Continued

Basis for threshold	Distinguishes ecologically good vs. degraded?	Quantitative from data?	Advantages	Disadvantages	How?	Examples	Graphical illustration. The solid green line indicates the threshold for good state and the green polygon indicates the region above threshold (where present)
Expert judgment: Expert elicitation to convert narrative description into quantitative description	Yes	No	Quantifies what looks natural to humans. Low demand for quantitative data. Less likely to result in resistance from stakeholders	Most subjective, inconsistent, and open to bias. Often not specified which type of good (from the other eight options) is the target	Combining the judgment of multiple experts in a structured process (Dorrough *et al.*, 2020)	Expert-driven process resulted in a proposal for an MPA network with 24% coverage to maximize the number of biologically unique seamounts and minimize socio-economic impacts (Wedding *et al.*, 2013)	Can aim to approximate any of the relationships above
Zero pressure: Any level of pressure results in a degraded state	No	Minimal data involved	Data requirements minimal as it only requires mapping of pressure without quantifying intensity	Many systems can withstand some level of pressure without resulting in significant adverse impacts, and impact varies between habitats and communities. Same state can be defined as either good or degraded	Map spatial distribution of pressure, assign all areas with no pressure as good state	% unfished area in seabed trawl impact assessments (ICES, 2021b)	

Approaches are reviewed against two evaluation criteria: (1) do thresholds distinguish good from degraded and (2) are the methodologies based on objective, repeatable methods that estimate the threshold quantitatively.