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A risk-based framework for assessing the cumulative impact of multiple fisheries

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ABSTRACT

Effective conservation of marine ecosystems requires the assessment and management of cumulative impacts of multiple activities occurring in the ecosystem. Productivity Susceptibility Analysis (PSA) is a widely used tool to assess the potential impacts of fishing activities on marine ecosystems, particularly in data-poor regions. Yet, PSA and other risk-based approaches often do not account for the cumulative effects of multiple fisheries operating in the same region. Here we amend PSA to incorporate multiple fisheries by proposing a new index for cumulative risk assessment, i.e. Aggregated Susceptibility (*AS*). We applied this extended PSA to 81 species caught in 5 small-scale fisheries along the coast of Baja California, Mexico, and compared the results to the original PSA. Using the original PSA approach, 18 species (22.2%) were scored as high risk, and twenty-five species (31%) are at low risk from all of the fisheries conducted in this region. When the cumulative risk posed by all fisheries is assessed using our proposed methodology, the proportion of species at high risk increases to 38.3%, whereas the proportion of species at low risk decreases to 21%. For 13 species, the high-risk assessment is made only when scores are aggregated. Among the 5 fisheries, the set gillnet fishery has the greatest impact, which accounted for half of the high risk species and should be the focus of further investigation on how to best manage this fishery. Our analysis demonstrates the importance of accounting for the potential cumulative impacts of multiple co-occurring fisheries for the conservation of coastal marine ecosystems, identifies relative risk imposed by multiple fisheries, and provides a tool for a preliminary evaluation of the possible outcomes of management alternatives.

1. Introduction

Ecosystem-based management (EBM) is widely considered as a strategy for achieving sustainable delivery of marine ecosystem services (Francis et al., 2011). Applications of the EBM framework to fisheries management – ecosystem-based fisheries management (EBFM) – is mandated in several nations around the world, including Canada (through Canada's Ocean Act of 1996) and the USA (through the Magnuson-Stevens Reauthorization Act of 2006). The USA has also implemented several fishery ecosystem plans among its regional fishery management councils (Tromble, 2008). Other organizations, such as the Food and Agriculture Organization (FAO) and European Union (EU) also promote policies on EBFM (FAO, 2003; Jennings and Rice, 2011).

These mandates and other calls for ecosystem-based management of the oceans have emphasized the fundamental need to assess and account for cumulative impacts of human activities (Ruckelshaus et al., 2008). Management focused on impacts of a single activity or stressor is often ineffective because co-occurring human activities lead to multiple simultaneous impacts on communities and individual species (Halpern et al., 2008). A key goal of EBFM is to assess and manage the cumulative impact of multiple fisheries, both on the species targeted by the fisheries and on the ecosystem as a whole, including non-target species. Operationalizing the concepts of EBFM includes developing system-level reference points, management triggers, and control rules to ensure that, for example, total biomass removal by multiple fisheries operating within the same region does not exceed a certain threshold fraction of total productivity (Pikitch et al., 2004). In addition, EBFM aims to reduce impacts on protected or endangered species and overall bycatch to protect ecosystem resilience and avoid irreversible change (Pikitch et al., 2004).

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Assessing the cumulative impacts of multiple fisheries operating within the same region or ecosystem presents major challenges. As Link et al. (2011) succinctly put it "we are now clearly beyond the why's and what's, and squarely in the middle of the how's". Many fisheries lack data on bycatch and, depending on the economic value and regulatory status of different fisheries, the amount of information available even for the target species of the fishery can greatly vary (e.g., Dowling et al., 2008; FAO, 2010; Shester and Micheli, 2011). In addition, because data-poor fisheries lack, by definition, extensive and reliable information on fishing effort and catches, assessments based on catch and effort data often cannot be performed. Thus, tools to assess and manage the cumulative impacts of multiple fisheries are needed, particularly in data-poor fisheries settings.

Faced with a lack of data on the impacts to target and non-target species fisheries, several risk-based frameworks and approaches have been developed to address potential risk posed by specific fisheries.

One well-accepted framework is the Productivity Susceptibility Analysis (PSA; Hobday et al., 2007). PSA is a semi-quantitative risk assessment tool that relies on the life history characteristics of a stock (i.e., productivity) and its susceptibility to a specific fishery. PSA was originally developed to classify differences in risk posed by bycatch in the Australian prawn fishery (Milton, 2001; Stobutzki et al., 2001). PSA was later modified by the Australian Ecological Risk Assessment (AERA) team (Hobday et al., 2004, 2007), who expanded it to include habitat and community components so that this tool could be used to assess ecosystem vulnerability. Subsequently, in the USA, an expert working group recommended that scientists evaluate the vulnerability for each stock based on an analysis of its productivity and susceptibility to the fishery, as a first step for setting annual catch limits (Rosenberg et al., 2007), and the NOAA Fisheries Vulnerability Evaluation Working Group (VEWG) chose a modified PSA approach as the best option to determine the vulnerability of data-poor stocks managed under fishery management plans (Patrick et al., 2009). PSA was also included in the Marine Stewardship Council Fisheries Assessment Methodology (2011).

The PSA approach is based on the assumption that the vulnerability of a species, habitat, or community will depend on two characteristics: (1) the extent of the impact due to the fishing activity, which is determined by the susceptibility to the fishing activities (Susceptibility) and (2) the productivity of the species, habitat or community (Productivity), which determines the rate at which recovery can occur after potential depletion or damage. The most vulnerable stocks are those that receive a low productivity score and a high susceptibility score, while stocks with a high productivity score and low susceptibility score are considered to be the least vulnerable (Patrick et al., 2009). It is important to note that PSA measures potential vulnerability (i.e., risk), not actual impact. A precise quantification of impact requires direct measures of abundance, demographic and spatial structure, vital rates and fishing mortality for the species, habitat or community, but this information is generally lacking in most small-scale and data-deficient fisheries. In fact, for most fisheries, even large-scale industrial fisheries, such information is generally available only for a limited number of target species. Thus, PSA is designed to allow assessment of vulnerability in the absence of abundance and mortality estimates (Hobday and Smith, 2008).

PSAs have been carried out for multi-species complexes and to evaluate vulnerability of both target and non-target species. However there is also a need to examine the cumulative impacts of co-occurring fisheries. When a stock is harvested by fisheries using different gear types, Patrick et al. (2009) suggested performing a separate PSA for each fishery, and then combining the results according to the proportion represented by each gear type in the total catch. Ormseth and Spencer (2011) took a different approach and scored attributes conservatively according to the fishery and gear type with the highest proportion of the total catch. These approaches have merits, but they are not exempt from limitations. For instance, when a new fishing activity is added to a system where a preexisting one is already operating, we would expect risk to increase in response to the additional fishing mortality. Yet, if the susceptibility score associated with the new fishery is lower than that of the preexisting fishery, the weighted average approach proposed by Patrick et al. (2009) would lower the vulnerability score instead of increasing it, while the approach suggested by Ormseth and Spencer (2011) would not result in any change in the final PSA score. In some cases, information might be available to allow scoring the susceptibility attributes with all the fisheries in mind (Patrick et al., 2009; Ormseth and Spencer, 2011). However, for most species and fisheries such information is not available, thereby making the direct assignment of an aggregated susceptibility score for all the fisheries altogether highly subjective, e.g. based on an arbitrary mental weighting of the relative intensities and impacts of different fisheries. Thus, assessing susceptibility with all the fisheries in mind is a legitimate but subjective approach.

New methods to address cumulative risks from multiple fisheries in a transparent and repeatable way are needed. Here, we (1) extend the original PSA approach developed by Hobday et al. (2004, 2007) to allow risk-based assessments of the cumulative effects of multiple fisheries operating within the same geographic region and (2) illustrate the application of this modified PSA using data from an actual system with spatially co-occurring small-scale fisheries. The resulting aggregated vulnerability scores quantify the cumulative risk posed to multiple species by co-occurring fisheries, and illustrate discrepancies between ranking relative threats to species based on single vs. multiple fisheries. By developing a new approach to capture the relative cumulative risk to species from multiple activities, we provide a general framework that can be extended to other types of risk assessments using similar axes of productivity and susceptibility to address risk from multiple stressors beyond fishing.

2. Materials and methods

2.1. Productivity-susceptibility analysis for multiple fisheries

PSA evaluates an array of productivity and susceptibility attributes for a stock (Table 1). In the original PSA methodology, productivity is computed by using basic information on life history of the species, such as age and size at maturity, fecundity, reproductive strategy, and trophic level (Hobday et al., 2007). Fishery susceptibility is assessed on the basis of information on the proportion of the spatial distribution of a given stock that overlaps that of the fishery, the encounterability, the selectivity of fishing gears, and post-capture mortality of discarded bycatch. More articulated classifications schemes including 22–75 attributes have been developed (e.g. Patrick et al., 2010) to adapt the PSA approach to available information. Here, we utilized the smaller original list of attributes to facilitate scoring of all species because basic information on many attributes is lacking for most of the species in our multi-species small-scale fisheries case study. Our goal is to propose an extended PSA methodology to assess cumulative risk, and thus to compare risk assessed for individual vs. multiple fisheries. While relative comparisons of risk within this case study are robust, it is important to note that risk assessments for individual species and fisheries cannot be directly compared with assessments from other applications of PSA that include a more extensive set of attributes or modified scoring systems (Field et al., 2010; Patrick et al., 2009, 2010; Ormseth and Spencer, 2011).

Table 1
PSA scoring attributes used in this analysis.

Productivity attributes	Low productivity/high risk (3)	Medium productivity/medium risk (2)	High productivity/low risk (1)
Average age at maturity	>4 years	2–4 years	<2 years
Average maximum age	>40 years	20–40 years	<20 years
Fecundity	<100 eggs per year	100–10,000 eggs per year	>10,000 eggs per year
Average maximum size	>80 cm	40–80 cm	<40 cm
Average size at maturity	>100 cm	40–100 cm	<40 cm
Reproductive strategy	Live bearer	Demersal egg layer	Broadcast spawner
Trophic level	>3.5	2.5–3.5	<2.5
Susceptibility attributes	**Low susceptibility/low risk (1)**	**Medium susceptibility/medium risk (2)**	**High susceptibility/high risk (3)**
Availability	Global	Pacific coast (North and South America)	Baja/Mexico only
Encounterability – habitat	Low overlap with fishing gear	Medium overlap with fishing gear	High overlap with fishing gear
Encounterability – bathymetry	Low overlap with fishing gear	Medium overlap with fishing gear	High overlap with fishing gear
Selectivity – nets	<17.8 cm average size at maturity	17.8–35.6 cm average size at maturity	>35.6 cm average size at maturity
Selectivity – fish traps	>18 cm average size at maturity	<3 cm average size at maturity	3–18 cm average size at maturity
Selectivity – lobster traps	>19.6 cm average size at maturity	<8.9 cm average size at maturity	8.9–19.6 cm average size at maturity
Selectivity – dive fishing	Non-target species	Not applicable	Target species
Post-capture mortality	Evidence of post-capture release and survival	Released alive	Retained species or majority dead when released

The PSA methodology assigns scores to the attributes on a three-point scale (Hobday et al., 2007), Productivity (P): 1 (high productivity = low risk) to 3 (low productivity = high risk), and Susceptibility (S): 1 (low susceptibility = low risk) to 3 (high susceptibility = high risk). Each of the attributes is scored independently of the others. From these scores, overall productivity and susceptibility scores are computed as the arithmetic average of the respective scores and plotted on an x–y scatter plot. The overall vulnerability score (V) for the stock is calculated by measuring the Euclidean distance of the data point from the origin of the plot, namely:

$$V = \sqrt{P^2 + S^2} \tag{1}$$

As described in Hobday et al. (2004, 2007), if all productivity and susceptibility scores are assumed to be equally likely, then one third of the vulnerability scores V will be greater than 3.18 (high risk), one third will be between 3.18 and 2.64 (medium risk), and one third will be lower than 2.64 (low risk).

We compare two alternative approaches for cumulative assessment of multiple fishing activities in data poor fisheries, when catch and effort data are not available, namely: Fisheries with the Greatest Impact (FGI) and the aggregated susceptibility index (AS).

2.1.1. Cumulative assessment: fisheries with the greatest impact (FGI)

A parsimonious approach when there are multiple overlapping fishing activities is to assume that the overall susceptibility score of each species is determined by the fishery assessed to pose the greatest risk (Ormseth and Spencer, 2011). This is equivalent to assuming that the total impact of fishing is driven by a dominant activity that overrides the effects of all others (e.g. Halpern et al., 2008). In this case, the susceptibility index S – hereafter referred to as FGI – is computed as follows:

$$FGI = \max(FSS_1, FSS_2, \ldots, FSS_{NoF}) \tag{2}$$

where FSS_i ($i = 1,\ldots,NoF$, where NoF is the number of fisheries) is a fishery-specific susceptibility score for each fishery affecting a target or non target species. The effect of fisheries with lower potential impact is thus overridden by those with the greatest impact, i.e. the fishery with the highest FSS. The overall vulnerability is calculated by setting the susceptibility $S = FGI$ and then combining the productivity and FGI score as in Eq. (1).

2.1.2. Cumulative assessment: aggregated susceptibility index (AS)

A second approach, when there are two or more fisheries affecting a species, is to assume that their cumulative potential impact may be larger (e.g., additive or multiplicative; Halpern et al., 2008) than that generated by the single fishery with greatest impact. To account for possible cumulative effects of multiple overlapping fishing activities, we developed an *Aggregated Susceptibility* (AS) score computed as 1 plus the Euclidean distance from the point $FSS_i = 1$ in the multidimensional NoF space. We bounded its maximum value to 3 to keep consistency with the range of susceptibility scores of the original PSA index (Hobday et al., 2007). In mathematical terms:

$$AS = \min\left(3,\quad 1 + \sqrt{(FSS_1 - 1)^2 + (FSS_2 - 1)^2 + \cdots + (FSS_{NoF} - 1)^2}\right) \tag{3}$$

AS increases with the number of fisheries that are accounted for, as long as their FSS is greater than 1. If two or more fisheries have FSS larger than 1, AS is larger than FGI, the maximum value among the FSS scores. For examples, if the FSS scores of two occurring fisheries using different fishing gears are 2 and 2.5, then $FGI = 2.5$ while $AS = 3$. Thus, multiple fisheries, each assessed to cause moderate risk, could result in high risk when combined, i.e. when simultaneously affecting the same species. A continuous, smooth version of formula (3) is reported in Appendix Fig. A1 along with the R code to implement it.

The overall vulnerability is calculated by setting the susceptibility $S = AS$ and then combining the productivity and the aggregated susceptibility scores as in Eq. (1). A MS-Excel file to compute the aggregated susceptibility, FGI and the corresponding vulnerability index is reported in Supplementary Online Materials.

2.2. Case study: the small-scale fisheries of the Pacific coast of Baja California, Mexico

2.2.1. Study system and fisheries

We studied the fishing cooperatives located along the coast of the Vizcaino Desert Biosphere reserve in the North Pacific region of Baja California Sur, Mexico (Fig. 1). The region encompasses the southern end of the California Current Large Marine Ecosystem and is characterized as temperate to subtropical (Martone, 2009). Cooperatives receive renewable 20 year concessions primarily for

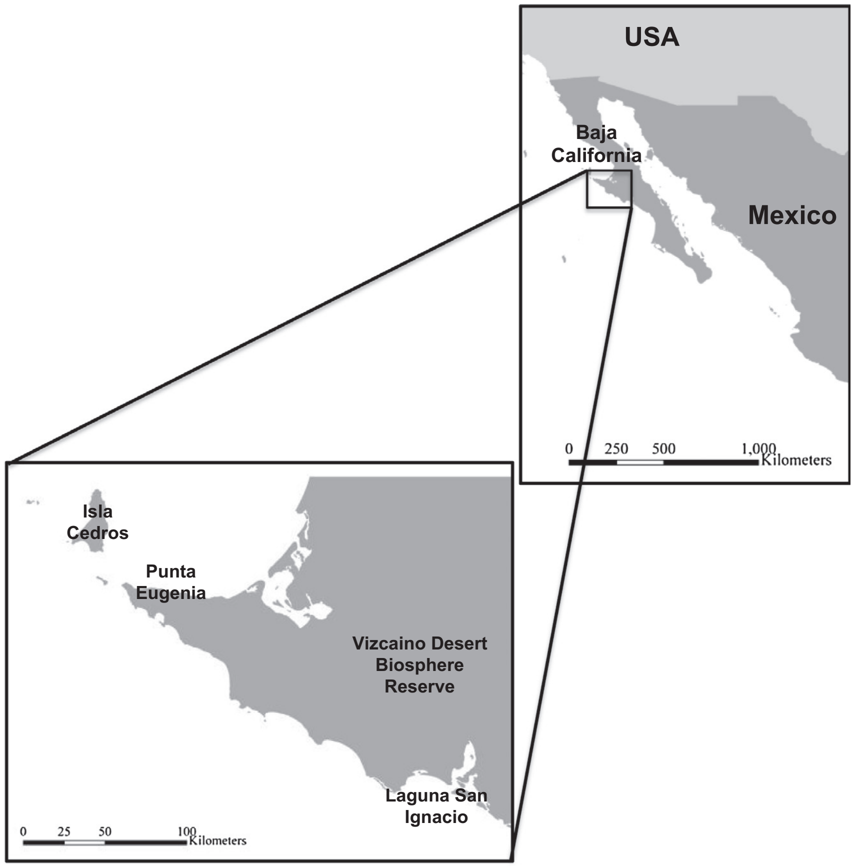

Fig. 1. Map of the study area. The fishing cooperatives of the Vizcaino region are located along the coastline from Isla Cedros to Laguna San Ignacio.

lobster (*Panulirus* spp., caught with traps), and abalone (*Haliotis fulgens* and *H. corrugata*, caught by hookah divers), as well as other benthic invertebrate and algal species, including the wavy turban snail *Megastraea undosa*, the sea cucumber *Parastichopus parvimensis*, the red sea urchin *Strongylocentrotus franciscanus*, and the red alga *Gelidium robustum*, all caught by hookah divers. The cooperatives also catch several dozen species of finfish, primarily barred sand bass (*Paralabrax nebulifer*), ocean whitefish (*Caulolatilus* spp.) and California halibut (*Paralichthys californicus*) with nets and traps. In contrast with benthic invertebrates and algae, cooperatives do not hold territorial rights for finfish. In total, fisheries in the area land and sell commercially over 33 fish and invertebrate species, and discard several dozen additional species exclusively as bycatch (Shester and Micheli, 2011). Depending on the target species and gear type, fishing occurs in several habitat types, from nearshore kelp forests, rocky reefs, and soft sediment to demersal habitat and pelagic waters several hundred meters deep. Thus, we have assigned these fisheries to five categories based on gear type, which are the dive fishery, fish trap fishery, lobster trap fishery, drift gillnet fishery, and set gillnet fishery.

We collected information on what species were directly targeted and which were discarded in each fishery through at-sea observation of fishing activities (Shester, 2008; Shester and Micheli, 2011). While bycatch can be defined in many different ways (e.g. MSRA, 2006; MLMA, 2004; Kelleher, 2005; MBA, 2006), we adopt the definition of the U.S. National Marine Fisheries Service to include all organisms that are caught in fishing gear, but not kept for sale or personal consumption (MSA, 1996). Observers joined fishing trips and recorded the species and size of all individual animals caught, and whether each was retained or discarded. Data were collected for a total of 4,940 lobster traps, each soaked for ~24 h and retrieved during 56 fishing trips, 502 fish traps, each soaked for ~30 min and retrieved during 16 trips, 83 daily set gillnet retrievals conducted during 30 fishing trips, and 4 overnight drift gillnet deployments (Shester and Micheli, 2011). In addition, we joined dive fishery trips, both on board of the fishing boats and in the water, and made qualitative observations on the depth ranges, habitats visited and species caught in these fisheries (Shester, 2008).

Life history information for productivity (*P*) scores for each species (Table 1) was obtained from online databases (FishBase, www.fishbase.org; SeaLifeBase, www.sealifebase.org), books (Eschmeyer et al., 1983; Santelices, 1988; Love, 1996), and journal articles (e.g., Grigg, 1977; McEuen, 1987; Cameron and Frankboner, 1989; Casas-Valdez et al., 2005). When we were not able to find data through these means, we used data from other species in the same genus or family as the species in question from FishBase, (www.fishbase.org) or SeaLifeBase (www.sealifebase.org). In other PSA applications, when no information is available the highest score was conservatively assigned (Rosenberg et al., 2007), though in our case study we were able to find information for closely related species. We modified Hobday's et al. (2007) Australian PSA thresholds for each attribute based on Field et al. (2010) application of PSA to fisheries of southern California (Table 1, Appendix Table A1). All productivity scores are reported in Appendix Table A2.

We used the attributes for susceptibility (*S*) from Hobday et al. (2007), modifying the thresholds to better represent the fisheries of Baja California (Table 1, and Appendix Table A1). In the case of the attribute "Availability" we modified Hobday's et al. (2007) threshold to be Global = score 1, Pacific Coast (greater than just Mexico) = score 2, and Baja and Mexico Pacific Coast only = score 3. In the case of "Selectivity" we based scoring on different gear types: selectivity for drift and set gillnets was based on the mesh size of 17.8 cm, for lobster trap on the 8.9 cm × 4.1 cm mesh, escape vents 25.4 cm × 4.5 cm, and entrance opening 19.6 cm × 14.1 cm, with the opening height above the seafloor of 0 cm, and for fish trap dimensions on 3 cm × 3 cm mesh, 18 cm diameter opening, and opening height above the seafloor of 6 cm (Table 1; see Hobday et al., 2007 for a description of criteria for selectivity). We obtained these values based on *in situ* observations, however, many of these values are specified in regulations. We scored sessile or sedentary species like abalones, algae, snails, urchins, sea cucumbers and gorgonians as low susceptibility for fish and lobster traps because they are not likely to go into the trap due to their low mobility. Species that could crawl into traps like crabs, lobster, and octopus were scored for susceptibility based on size. All susceptibility scores are reported in Appendix Table A3.

2.2.2. PSA and extended PSA calculations

We first computed the productivity scores for each species, and the susceptibility of each species to each fishery. Resulting scores were then combined to compute the overall vulnerability for each species by using Eqs. (1)–(3). According to Hobday et al. (2004, 2007), a species with vulnerability V smaller than 2.64 was classified as being at low risk, between 2.64 and 3.18 as at medium risk and larger than 3.18 as a species at high risk.

3. Results

3.1. Individual fisheries assessment

When fisheries are assessed independently of each other, over half of the 81 species that are caught in local fisheries were found to be at low risk from individual fisheries (Table 2, Fig. 2). Fish traps pose low risk to a majority of species (80.2% of species), followed by dive fisheries (76.5%) (Table 2, Fig. 2). The high proportion of species at low risk for fish traps is likely due to the fact that finfish that are susceptible to this gear have moderate to high productivities, while benthic invertebrates and other seafloor species that have lower productivities are unable to enter the trap because the entrances are several centimeters above the seafloor. The high proportion of species at low risk from dive fisheries is likely explained by the high selectivity of these fisheries, given that individuals are hand picked by divers, combined with the high productivity of species in some cases (e.g., the wavy turban snail; Table 2). Drift gillnets also pose low risk for a large fraction of the species assessed (70.4%) as they interact only with the top part of the water column and thus only with the species using this habitat. Proportions of species at low risk among the remaining fisheries are 61.7% for lobster traps, and 53.1% for set gillnets (Fig. 2).

Among the different fisheries conducted by the Vizcaino fishing cooperatives, set gillnets are assessed to pose a high risk for the greatest number of species (Table 2, Fig. 2). Eighteen of the 81 species assessed (22.2%) are at high risk from set gillnet fishing (Fig. 2). In contrast, 9 species are at high risk from drift gillnets (11.1%), 1 from lobster traps (1.2%), and none from fish traps or dive fisheries (Fig. 2). All of the species at high risk from drift gillnets and the only one at high risk from lobster traps (small individuals of the leopard shark, *Triakis semifasciata,* are caught in traps as bycatch) are also at high risk from set gillnets (Table 2). In general, these high risk species are species with low productivities (e.g., sharks, rays, and marine mammals).

3.2. Cumulative assessment: fisheries with the greatest impact (FGI)

FGI analysis identified 25 species (31%) that are at low risk from all of the fisheries conducted in this region (Table 2, Figs. 2 and 3a). Of these, 18 are not targeted by any of the fisheries (Table 2). However, the remaining seven are fishing targets: the California scorpionfish (*Scorpaena guttata*), the yellowtail amberjack (*Seriola lalandi*), the pacific mackerel (*Scomber japonicus*), the pacific sardine (*Sardinops sagax*), the spiny lobster (*Panulirus interruptus*), the wavy turban snail (*M. undosa*), and the giant keyhole limpet (*Megathura crenulata*) (Table 2). These species have relatively low overall vulnerability as the risk posed by fishing is mitigated by their high productivity scores (Appendix Table A2).

Eighteen species (22.2%) are at high risk from at least one fishery (Table 2, Figs. 2 and 3a). These included two of the main targets of drift and set gillnets, the shortfin corvina (*Cynoscion parvipinnis*) and the giant seabass (*Stereolepis gigas*), and 16 species caught as bycatch by these fisheries, including several species of sharks and rays, the bottlenose dolphin (*Tursiops truncatus*), and the California sea lion (*Zalophus californianus*) (Table 2).

The high risk posed by fishing in this system is caused primarily by drift and set gillnet fisheries. These fisheries pose a high risk to both target species and species that have no commercial value or that are protected by law. The remaining 38 (45.7%) species are at medium risk from at least one fishery (Table 2, Figs. 2 and 3a).

Simultaneous consideration of risk posed by all fisheries within a region indicates that risk associated with fishing may in fact be greater than what one would conclude by individual assessment of individual fisheries, even when focusing on the fisheries with the greatest impact (in this case, set gillnets). Comparison of vulnerability scores computed by accounting for set gillnets only and by using the fishery with greatest impact scores in Table 2 indicates that the proportion of species at high risk is the same (22.2%, 18 species) in the set gillnets and FGI scenario, suggesting that assessment of set gillnets captures most high risk cases. However, the number of species at low risk decreases by 40% (from 41 to 22) and that at medium risk increases by 73% (from 22 to 38) when computing susceptibility with the FGI index instead of accounting for only set gillnets, indicating that risk is underestimated if fisheries are not assessed in combination (Fig. 2).

If set gillnets, the gear type that poses high risk to the greatest number of species, are removed from the assessment, the *FGI* scores decrease for 40 species by an average of 30.4% (±9.3%). Of these 40 species, 23 change risk categories, whereas 14 (37%) of the medium risk species moved to low risk and 9 (50%) of the high risk species moved to medium risk (Table 2, Figs. 2 and 3b).

3.3. Cumulative assessment: the aggregated susceptibility index (AS)

Aggregating the susceptibility scores of the different fisheries operating in the same area results in a remarkable increase (+72%), compared to the FGI case, of the number of species at high risk – from 18 (22.2%) to 31 (38.3%) – and a reduction (−32%) of the species at low risk – from 25 (30.9%) to 17 (21%) – and medium risk – from 38 (46.9%) to 33 (40.7%) (Table 2, Figs. 2 and 4a). Eight species move from the low to the medium risk class, and 13 from the medium to the high class. The mean susceptibility score increases by 11.2 ± 7% (maximum relative increase 29%) when moving from the FGI to the AS approach, which translates into a 6.9 ± 5% increase in the overall vulnerability score (maximum relative increase 22%).

Regardless of the actual score values, these results indicate that there is a potential for significant cumulative impacts of multiple fisheries on several species. Aggregated scores indicate that the species at greatest risk from combined fishing impact in this system include some finfishes, several elasmobranchs, and two marine mammals, the bottlenose dolphin (*T. truncatus*), and the California sea lion (*Z. californianus*) (Table 2). These species should therefore be priorities for monitoring and stock assessment.

Some of these species were already assessed as high risk through the individual fisheries assessment or *FGI* approach (Table 2). However, for 13 species, the high-risk assessment is

Table 2
PSA and extended PSA results showing scores and corresponding risk levels (low, medium, and high risk) for the 5 individual fisheries, the fishery with greatest impact scenario, and aggregating scores among the fisheries, both with and without set gillnets.

Common name	Target fishery	Risk: set gillnets	Risk: drift gillnets	Risk: lobster traps	Risk: fish traps	Risk: dive fishing	Fishery with greatest impact PSA score	Risk: fishery with greatest impact	Aggregated susceptibility PSA score	Aggregated susceptibility risk	Aggregated susceptibility without set gillnets PSA score	Aggregated susceptibility risk without set gillnets
Shortfin corvina	Drift	**High**[a]	**High**[a]	Low	Low	Low	3.46	**High**	3.46	**High**	3.46	**High**
California moray eel	None	Med[a]	Low	Low	Low	Low	3.16	**Med**	3.28	**High**	2.72	**Med**
Gulf grouper	None	**High**[a]	**Med**	**Med**	Low	Low	3.26	**High**	3.38	**High**	2.83	**Med**
California flounder	Set	**Med**[a]	Low	Low	Low	Low	3.16	**Med**	3.26	**High**	2.67	**Med**
Pacific bonito	Drift	**Med**[a]	**Med**[a]	Low	Low	Low	2.98	**Med**	3.43	**High**	2.99	**Med**
Pacific barracuda	Drift	**Med**[a]	**Med**[a]	Low	Low	Low	2.98	**Med**	3.43	**High**	2.99	**Med**
Giant sea bass	Set	**High**[a]	**High**[a]	**Med**	**Med**	Low	3.36	**High**	3.82	**High**	3.44	**High**
California lizardfish	None	Low[a]	Low	Low	Low	Low	2.36	Low	2.50	Low	2.14	Low
Pacific electric ray	None	**High**[a]	**High**[a]	**Med**	**Med**	**Med**	3.47	**High**	3.87	**High**	3.48	**High**
Pelagic stingray	None	**Med**[a]	**Med**[a]	**Med**	Low	Low	2.94	**Med**	3.10	**Med**	2.94	**Med**
Spiny dogfish	None	**High**[a]	**High**[a]	**Med**	**Med**	**Med**	3.30	**High**	3.44	**High**	3.30	**High**
Smooth stargazer	None	Low[a]	Low	Low[a]	Low	Low	2.36	Low	2.70	**Med**	2.41	Low
Pacific angelshark	None	**High**[a]	**High**[a]	**Med**	**Med**	**Med**	3.57	**High**	3.96	**High**	3.58	**High**
Scalloped hammerhead	None	**High**[a]	**High**[a]	**Med**	**Med**	**Med**	3.30	**High**	3.44	**High**	3.30	**High**
Yellowtail amberjack	Drift	Low[a]	Low[a]	Low	Low	Low	2.59	Low	2.77	**Med**	2.59	Low
Plainfin midshipman	None	Low[a]	Low	Low	Low	Low	2.54	Low	2.68	**Med**	2.34	Low
Popeye catalufa	None	Low[a]	Low	Low	Low	Low	2.27	Low	2.43	Low	2.04	Low
Roughbar frogfish	None	Low	Low	**Med**[a]	Low	Low	2.66	**Med**	2.94	**Med**	2.71	**Med**
Pelagic cormorant	None	**Med**[a]	**Med**[a]	Low	Low	Low	2.82	**Med**	3.01	**Med**	2.85	**Med**
Harbor seal	None	**Med**[a]	**Med**[a]	**Med**	**Med**	**Med**	3.18	**Med**	3.32	**High**	3.18	**Med**
Bottlenose dolphin	None	**High**[a]	**High**[a]	**Med**	**Med**	**Med**	3.42	**High**	3.56	**High**	3.42	**High**
California sea lion	None	**High**[a]	**High**[a]	**Med**	**Med**	**Med**	3.57	**High**	3.96	**High**	3.58	**High**
California butterfly ray	None	**High**[a]	**Med**	**Med**	**Med**	**Med**	3.47	**High**	3.58	**High**	3.07	**Med**
Ocean whitefish	Fish	**Med**[a]	Low	Low	Low	Low	2.98	**Med**	3.13	**Med**	2.55	Low
Giant kelpfish	None	Low[a]	Low	Low	Low	Low	2.54	Low	2.68	**Med**	2.34	Low
Kelp bass	Fish	Low	Low	**Med**[a]	Low	Low	2.89	**Med**	3.18	**Med**	2.97	**Med**
Swell shark	None	**High**[a]	**Med**	**Med**	**Med**	**Med**	3.47	**High**	3.58	**High**	3.07	**Med**
Longnose skate	None	**High**[a]	**Med**	**Med**	**Med**	**Med**	3.57	**High**	3.68	**High**	3.19	**High**
California scorpionfish	Set	Low[a]	Low	Low	Low	Low	2.64	Low	2.77	**Med**	2.44	Low
Pile perch	None	**Med**[a]	Low	Low	Low	Low	2.85	**Med**	2.97	**Med**	2.67	**Med**
Leopard shark	None	**High**[a]	**High**	**High**	**Med**	**Med**	3.68	**High**	3.79	**High**	3.31	**High**
Cabezon	None	Low[a]	Low	Low	Low	Low	2.54	Low	2.68	**Med**	2.34	Low
California sheephead	Set	**Med**	Low	**Med**[a]	**Med**[a]	Low	3.16	**Med**	3.69	**High**	3.61	**High**
Black croaker	None	Low[a]	Low	Low	Low	Low	2.54	Low	2.68	**Med**	2.34	Low
Wavyline grunt	None	**Med**[a]	Low	Low	Low	Low	2.81	**Med**	3.04	**Med**	2.46	Low
Shovelnose guitarfish	None	**High**[a]	**Med**	**Med**	**Med**	**Med**	3.57	**High**	3.68	**High**	3.19	**High**
Banded guitarfish	None	**High**[a]	**Med**	**Med**	Low	Low	3.26	**High**	3.38	**High**	2.83	**Med**
California sargo	None	Low	Low	**Med**[a]	Low	Low	2.81	**Med**	3.25	**High**	3.06	**Med**
Barred sand bass	Fish	Low	Low	**Med**[a]	Low	Low	2.89	**Med**	3.18	**Med**	2.97	**Med**
Pacific porgy	None	**Med**[a]	Low	Low	Low	Low	2.81	**Med**	2.94	**Med**	2.29	Low
Bocaccio rockfish	None	**Med**[a]	Low	Low	Low	Low	3.16	**Med**	3.28	**High**	2.72	**Med**
Burrito grunt	None	Low[a]	Low	Low	Low	Low	2.36	Low	2.50	Low	2.14	Low
Kelp perch	None	Low	Low	**Med**[a]	Low	Low	2.89	**Med**	3.05	**Med**	3.00	**Med**
Thornback guitarfish	None	**High**[a]	**Med**	**Med**	Low	Low	3.26	**High**	3.38	**High**	2.83	**Med**
Rubberlip sea perch	None	**Med**[a]	Low	Low	Low	Low	2.74	**Med**	2.87	**Med**	2.56	Low
Finescale triggerfish	None	Low[a]	Low	Low	Low	Low	2.45	Low	2.59	Low	2.24	Low
Rainbow sea perch	None	Low	Low	**Med**[a]	Low	Low	2.98	**Med**	3.23	**High**	3.03	**Med**

(*continued on next page*)

Table 2 (*continued*)

Common name	Target fishery	Risk: set gillnets	Risk: drift gillnets	Risk: lobster traps	Risk: fish traps	Risk: dive fishing	Fishery with greatest impact PSA score	Risk: fishery with greatest impact	Aggregated susceptibility PSA score	Aggregated susceptibility risk	Aggregated susceptibility without set gillnets PSA score	Aggregated susceptibility risk without set gillnets
Rock wrasse	None	Low[a]	Low	Low	Low	Low	2.27	Low	2.43	Low	2.04	Low
Black perch	None	Low	Low	**Med**[a]	Low	Low	2.73	**Med**	3.02	**Med**	2.97	**Med**
White seabass	Drift	**Med**[a]	**Med**[a]	Low	Low	Low	2.98	**Med**	3.43	**High**	2.99	**Med**
Horn shark	None	**High**[a]	**Med**	**Med**	Low	Low	3.26	**High**	3.38	**High**	2.83	**Med**
California spotted stingray	None	Low	Low	**Med**[a]	**Med**	Low	3.07	**Med**	3.33	**High**	3.28	**High**
Garibaldi	None	Low[a]	Low	Low	Low	Low	2.45	Low	2.63	Low	2.29	Low
Bat ray	None	**High**[a]	**Med**	**Med**	**Med**	Low	3.36	**High**	3.48	**High**	2.95	**Med**
Senorita	None	Low	Low	Low[a]	Low	Low	2.59	Low	2.89	**Med**	2.84	**Med**
Pacific mackerel	Drift	Low[a]	Low	Low	Low	Low	2.09	Low	2.20	Low	1.92	Low
Bullseye puffer	None	Low	Low	**Med**[a]	Low	Low	2.73	**Med**	3.18	**High**	2.98	**Med**
Blue crab	None	Low	Low	Low[a]	Low[a]	Low	1.66	Low	1.72	Low	1.72	Low
Yellow rock crab	None	Low[a]	Low	Low	Low	Low	2.22	Low	2.34	Low	2.14	Low
Eugorgia ampla	None	**Med**[a]	Low	Low	Low	Low	3.07	**Med**	3.18	**Med**	2.57	Low
Red gorgonian	None	**Med**[a]	Low	Low	Low	Low	3.07	**Med**	3.18	**Med**	2.57	Low
Leptogorgia diffusa	None	**Med**[a]	Low	Low	Low	Low	3.07	**Med**	3.18	**Med**	2.57	Low
Golden gorgonian	None	**Med**[a]	Low	Low	Low	Low	3.07	**Med**	3.18	**Med**	2.57	Low
Red octopus	Dive	Low	Low	**Med**[a]	Low	**Med**[a]	2.73	**Med**	3.32	**High**	3.26	**High**
Pacifigorgia sp.	None	Low[a]	Low	Low	Low	Low	2.59	Low	2.63	Low	2.38	Low
California spiny lobster	Lobster	Low	Low	Low[a]	Low	Low	2.44	Low	2.55	Low	2.46	Low
Blacksmith	None	Low	Low	Low[a]	Low	Low	2.46	Low	2.72	**Med**	2.66	**Med**
Halfmoon perch	None	Low	Low	**Med**[a]	Low	Low	2.73	**Med**	3.00	**Med**	2.78	**Med**
Warty sea cucumber	Dive	Low	Low	Low	Low	**Med**[a]	2.66	**Med**	2.68	**Med**	2.67	**Med**
Pacific sardine	Drift	Low[a]	Low[a]	Low[a]	Low	Low	1.87	Low	1.98	Low	1.94	Low
Opaleye	None	**Med**[a]	Low	Low	Low	Low	2.66	**Med**	2.80	**Med**	2.11	Low
Slate pencil urchin	None	Low[a]	Low	Low	Low	Low	1.64	Low	1.71	Low	1.66	Low
Pink abalone	Dive	Low	Low	Low	Low	**Med**[a]	2.66	**Med**	2.66	**Med**	2.66	**Med**
Green abalone	Dive	Low	Low	Low	Low	**Med**[a]	2.73	**Med**	2.73	**Med**	2.73	**Med**
Striped sea chub	None	Low[a]	Low	Low	Low	Low	2.27	Low	2.43	Low	2.04	Low
Wavy turban snail	Dive	Low	Low	Low	Low	Low[a]	2.59	Low	2.61	Low	2.60	Low
Giant keyhole limpet	Dive	Low	Low	Low	Low	Low[a]	2.59	Low	2.60	Low	2.60	Low
Red sea urchin	Dive	Low	Low	Low	Low	**Med**[a]	2.66	**Med**	2.66	**Med**	2.66	**Med**
Purple sea urchin	Dive	Low	Low	Low	Low	**Med**[a]	2.73	**Med**	2.73	**Med**	2.73	**Med**
Southern sea palm	None	Low[a]	Low	Low	Low	Low	2.45	Low	2.61	Low	2.26	Low
Red algae	Dive	Low	Low	Low	Low	**Med**[a]	2.73	**Med**	2.91	**Med**	2.85	**Med**

[a] Indicates fishery with greatest impact. Medium risk is reported in bold, and high risk is underlined. For each species, the main fishery targeting the species is reported (drift = drift gillnets, set = set gillnets, lobster = lobster traps, fish = fish traps, dive = dive fishing, and none = not targeted by any fishery).

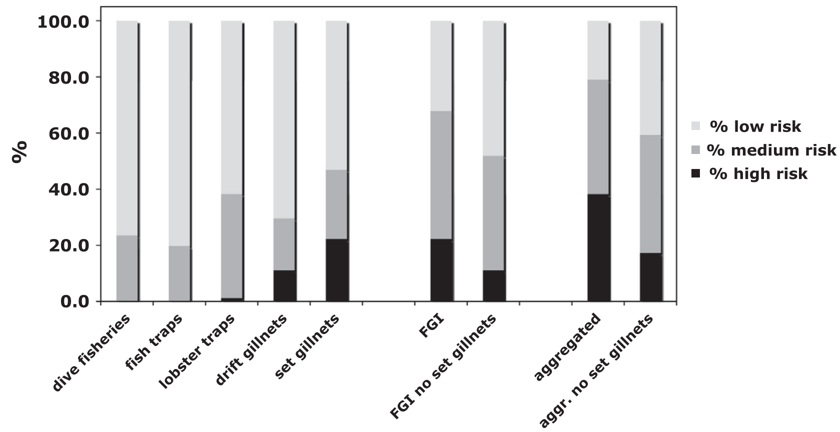

Fig. 2. Proportions of species at low, medium and high risk from individual fisheries, considering the high risk score estimated for any one fishery (fishery with greatest impact case, *FGI*), *FGI* without set gillnets, and aggregating susceptibility scores with and without set gillnets.

made only when scores are aggregated. This is because the aggregated susceptibility scores account for all the individual scores of fisheries, whereas the *FGI* scenario accounts only for the highest score. For example, under the *FGI* case, three or more fisheries resulting in medium risk for a given species result in an assessment of high risk when scores are aggregated, but medium risk if only the highest score is considered (i.e., the California sheephead, *Semichossyphus pulcher*, the harbor seal, *Phoca vitulina*, and the red octopus, *Octopus rubescens*; Table 2). Thus, the aggregated scores capture the possible cumulative effects of multiple simultaneous impacts and identify the species for which high vulnerability may result from the overlap of fisheries. Moreover, three target species that are at low risk from each of the five fisheries shift to medium risk when scores are aggregated: the spiny lobster (*P. interruptus*), the wavy turban snail (*M. undosa*), and the yellowtail (*S. lalandi*). This result indicates that assessments of these valuable species that focus only on their target fishery may underestimate risk. For example, if lobster bycatch in gillnets is not monitored, risk could be under-estimated for this species.

When set gillnets are removed from the assessment the aggregated scores decrease for 72 of the 81 species, by an average 20% (±11%) (Table 2, Figs. 2 and 4b). Of these 72 species, 33 change categories: 16 move from medium to low risk and 17 from high to medium risk. By removing set gillnets the proportion of species classified as high risk decreases by 55% while that as low risk increases by 94%. Thus, focusing management attention on the set gillnet fishery may be key to reducing the number of species at high risk (at least 1/2 of species at high risk) (Table 2, Figs. 2 and 4b).

4. Discussion

Our extension of PSA provides a tool for evaluating risk posed by overlapping fisheries within an ecosystem-based management framework that accounts for the full suite of extractive activities and their possible interactions. Like other risk-based approaches (Pitcher and Preikshot, 2001; Samhouri and Levin, 2012), our method can only assess potential vulnerability not actual cumulative impact of multiple fisheries. However, it provides a means of conducting a first screening of the many target and non-target species potentially affected by multiple fisheries within a given ecosystem. This approach can be easily and inexpensively applied across different systems, especially in data-deficient settings, by relying on values derived from the literature, specific characteristics of fishing gear, and local knowledge. When resources are available, additional data on both fishing activities and target and bycatch species can be gathered to improve the application of PSA. PSA results can also be used to prioritize activities and ecosystem components for more intensive and costly field and modeling studies that may be conducted to assess cumulative impacts. For example, this extended PSA can help prioritize assessment and monitoring efforts under limited resources by focusing on the species at highest aggregated risk and the fisheries with the greatest contribution to cumulative impacts. We recommend that the aggregated susceptibility index (*AS*) be used for assessing cumulative risk, especially when catch and effort data are not available.

The result that combining PSA scores can change assessments of potential vulnerability for both target and non-target species is perhaps not surprising: multiple fisheries simultaneously affecting a species are more likely to cause risk relative to each individual fishery. However, the extended PSA approach we developed provides a means of systematically and transparently comparing cumulative impacts, thereby producing some useful insights. For example, four fisheries affecting species A may pose lower risk than only two fisheries affecting species B. This is because species A may have life history traits making it more resilient to fishing impacts than B, and/or because fishing impacts are relatively minor in the first case, but greater in the second one. For instance, the bottlenose dolphin (*T. truncatus*) is at high risk even though this species is affected by only two fisheries (set and drift gillnets), while several finfish species, including the giant kelpfish (*Heterostichus rostratus*), cabezon (*Scorpaenichthys marmoratus*), rock wrasse (*Halichoeres semicinctus*), garibaldi (*Hypsypops rubicundus*), and striped sea chub (*Kyphosus analogus*) are affected by four fisheries but are still classified as low risk, as all species have a high productivity score (low vulnerability). This demonstrates the importance of assessing and accounting for the potential cumulative impacts of multiple co-occurring fisheries, and provides a flexible tool for conducting such assessments.

Results from individual assessments should be interpreted with caution, in part because we used only a subset of the attributes used in other analyses and this may affect overall risk scores (Field et al., 2010; Patrick et al., 2010). While relative comparisons of risk among species and between individual and aggregate assessments are robust, absolute assessments of risk should be re-evaluated for species where information is available for a broader suite of attributes and fishing activities. Here, we used the shorter original list of attributes (Hobday et al., 2007) in order to maintain scoring consistent across the 81 fish and invertebrate species directly or indirectly targeted by fishing in our study system, for which data availability is greatly variable. We recommend

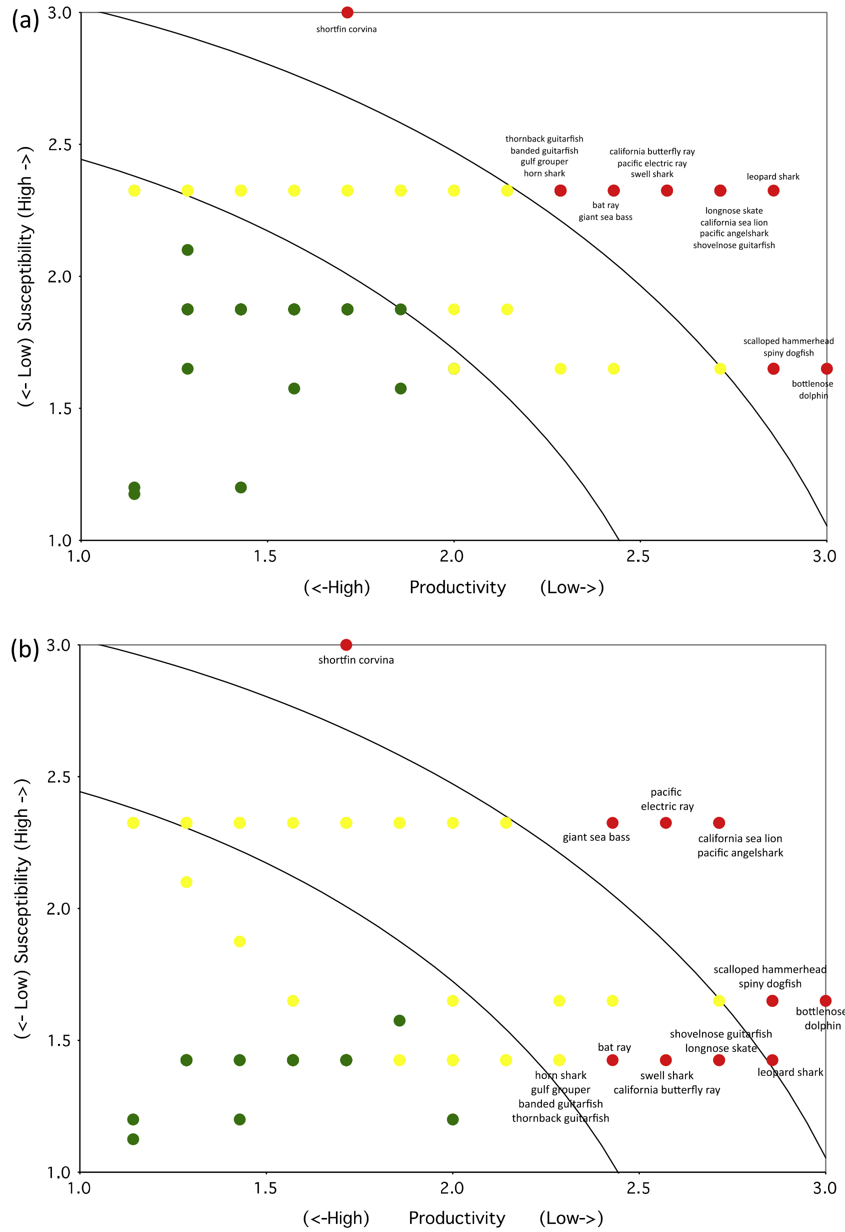

Fig. 3. PSA results for the fishery with greatest impact (*FGI*) case (a) and for *FGI* after set gillnets are excluded (b). The colors of the dots are based on the original colors from the FGI case, in panel a (red = high risk, yellow = medium risk, green = low risk), to highlight the movement of species to lower risk categories when set gillnets are excluded. The common name of species at high risk is reported in each panel.

that the subset of species identified as potentially at high risk from the cumulative effect of multiple fisheries should be re-assessed using an extended attribute list, whenever information is available.

Another important consideration is whether our risk assessments reflect actual stock status. Due to a lack of data, we were unable to examine whether the cumulative risk scores reflect the actual status of the stocks. Quantitative stock assessments have been conducted only for the primary target species in these Baja California fisheries (e.g., spiny lobster; Vega et al., 2010). Future applications of PSA using a larger number of attributes, where these data exist, could be directly compared to available stock assessments to determine whether aggregate PSA scores actually reflect the stocks status (Patrick et al., 2010; Cope et al., 2011).

It is also important to note that fishing is not the only threat to these coastal species. Stock assessments of green and pink abalone indicate that, overall, stocks have been recovering, following major decline in the 1980s and 90s through a combination of excessive fishing effort and the impacts of ENSO (El Nino Southern Oscillation) events (Guzmán del Próo, 1992; Morales-Bojórquez et al., 2008; Searcy-Bernal et al., 2010). However, recent mortality events possibly due to hypoxia (Micheli et al., 2012) pose an additional threat to these populations and have reversed the recovering trends at some locations. This further emphasizes the need for risk- and impact-based assessments that take multiple stressors, beyond fishing, into consideration (Halpern et al., 2008; Samhouri and Levin, 2012).

Applications of an extended PSA approach, exemplified here through our Baja California case study, allow for an identification of the species facing the greatest risk of depletion from the multiple fisheries that target them, directly or indirectly, as well as of priorities and opportunities for managing fisheries or shifting into fishing activities that pose lower risk. In this region, similar to

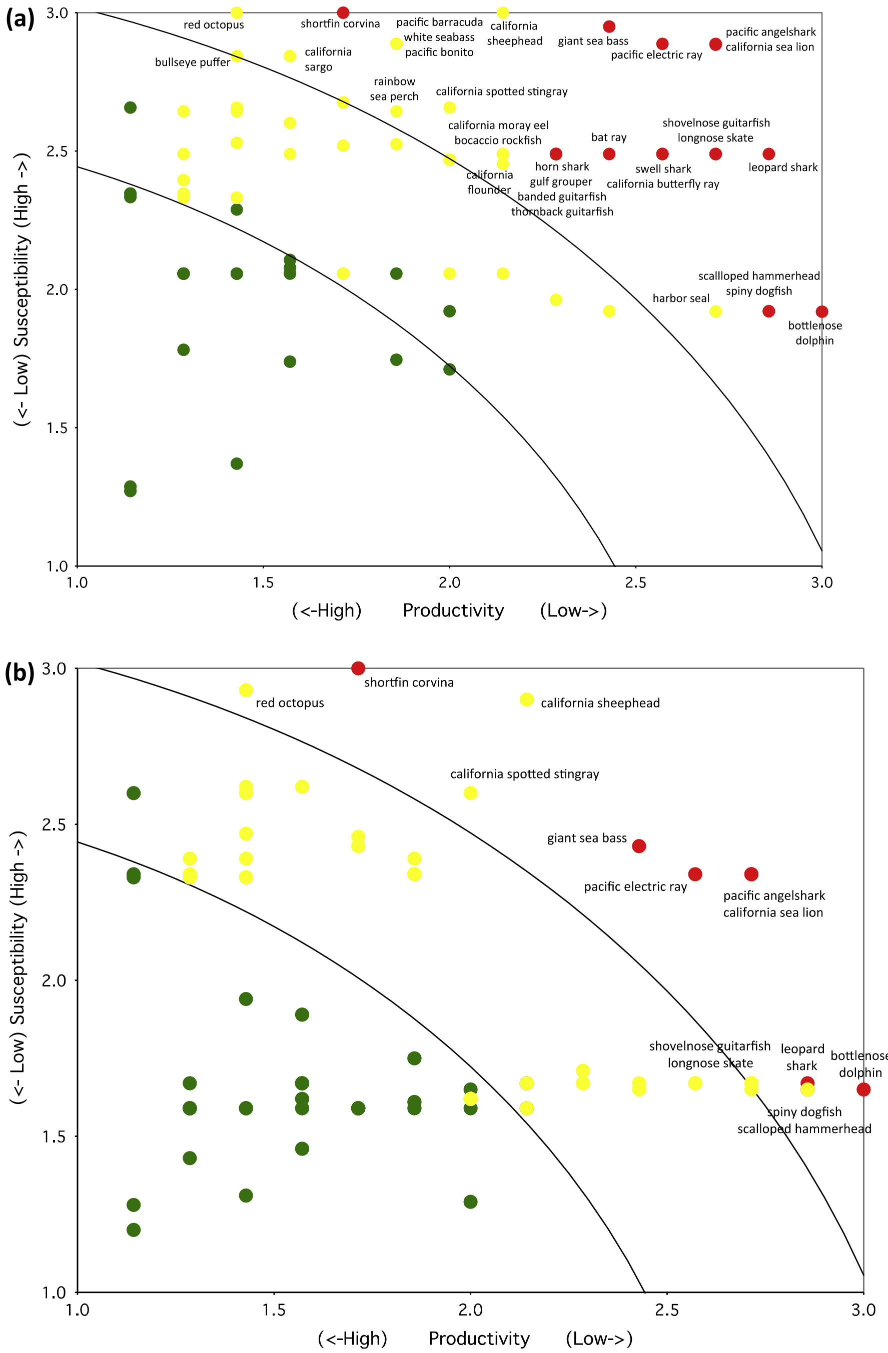

Fig. 4. PSA results for aggregated susceptibility scores (*AS*) among the 5 fisheries (a) and for *AS* after set gillnets are excluded (b). The colors of the dots are based on the original colors from the *FGI* case, with (Fig. 3a) and without (Fig. 3b) set gillnets (red = high risk, yellow = medium risk, green = low risk), to highlight the movement of species to higher risk categories when multiple fisheries are combined. The common name of species at high risk is reported in each panel.

many others, it is critical that the resources and fisheries with the lowest vulnerability, and thus the greatest potential to sustain local economies in the coming decades are identified.

Among the species currently targeted by local fisheries, Pacific mackerel and Pacific sardine are at low risk for both individual fisheries and when all fisheries are considered in the aggregated susceptibility score. However, our analysis did not include the industrial purse seine fisheries, which is likely the dominant source of mortality for these stocks. Moreover, such low trophic level species may be just as susceptible to fishery collapses as other species despite their high productivity (Pinsky et al., 2011). Several additional species (see results) were never assessed as high risk, even when susceptibility to individual fisheries is aggregated. For these species, life history characteristics (high productivity) and selective catch by only one or two fisheries combine to yield an assessment of low vulnerability. Thus, such species do not appear to experience high risk from the cumulative effects of multiple fisheries. Although other factors influencing the future sustainability of these fisheries need to be accounted for as well (e.g., possible climatic impacts on benthic invertebrates, lack of management plans for finfish species, lack of exclusive access rights; Shester, 2008; Micheli et al., 2012; McCay et al., 2013), results of our risk-based analysis highlight these species as those with the lowest potential vulnerability to the fisheries currently occurring in our study region.

PSA also highlighted the high vulnerability of several of the target and non-target species. Species characterized by low productivity, like the shortfin corvina (*Cynoscion parvipinnis*), the giant sea bass (*S. gigas*), both targeted by gillnet fisheries, and 17 non-target species, including several sharks and rays, the bottlenose dolphin (*T. truncatus*), and the California sea lion (*Z. californianus*) were scored as high risk in individual assessments. However, aggregated scores highlighted additional species at high risk, not identified through individual fisheries assessments, including the California halibut (*P. californicus*), pacific barracuda (*Sphyraena argentea*), harbor seal (*P. vitulina*), kelp bass (*Paralabrax clathratus*), California scorpionfish (*S. guttata*), California sheephead (*S. pulcher*), and white seabass (*Atractoscion nobilis*). These results highlighted those species that are caught as main targets or bycatch in multiple fisheries and have moderate to low productivity, thereby creating the potential for negative cumulative impacts. The assignment of relative risk across gear types by our PSA approach and the identification of species at high risk is consistent with recent studies comparing bycatch and habitat impacts across fisheries through *in situ* experiments and fishery observation (Shester and Micheli, 2011).

The extended PSA approach presented here also allows for an evaluation of how possible management changes may influence multiple species within the ecosystem. Thus, different options can be directly compared to gain some information on what strategy may yield the greatest ecological benefits. In our case study, because gillnet fisheries have the greatest potential impacts on a large fraction of species, it is important to examine how the overall risk posed to species by fishing may change if these fisheries were progressively eliminated or shifted to other fishing gears (traps, and hook and line) instead (e.g. Shester, 2008; Shester and Micheli, 2011; McCay et al., 2013). For example, the State of California, USA, enacted a state waters set gillnet ban in 1994 out to 3 miles from the coastline. Our analysis shows that the removal of set gillnets off Baja California would reduce risk from high to medium for several species of sharks and rays, and from medium to low for a suite of finfish species and for all gorgonian corals. This is consistent with empirical fishery data from the Southern California Bight showing significant recovery of white seabass, soupfin shark, leopard shark, and giant seabass as a result of the nearshore gillnet closure (Pondella and Allen, 2008). In our case, a set gillnet ban would reduce the percentage of species at high risk by half and would double the percentage of species at low risk (Table 2, Fig. 2). Several species would no longer be at high risk, based on aggregated vulnerability, including important target species like white seabass (*A. nobilis*), kelp bass (*P. clathratus*), and barred sand bass (*Paralabrax nebulifer*), as well as the harbor seal (*P. vitulina*). Removal of both drift and set gillnets (i.e., a full gillnet ban) would result in no species assessed as high risk in individual assessments, except for leopard sharks, that are caught as bycatch in trap fisheries (Table 2). At the least, these results indicate that conservation and data collection efforts focused on set gillnets would likely be most cost effective in reducing overall risk to the nearshore Baja California marine ecosystem.

The PSA methodology does not account for possible synergistic effects of multiple, indirect impacts, and thus it may underestimate risk. For example, one fishery may impact the prey base (Smith et al., 2011; Madigan et al., 2012) or habitat of other species, not directly targeted by the fishery (Shester and Micheli, 2011). While a complete understanding of cascading effects can be reached only through large-scale experiments and complex and data rich food web models, risk-based approaches that take into consideration linkages among ecosystem components may better reflect cumulative effects (Patrick et al., 2010). Our approach would be improved by integrating ecological interactions into an assessment of cumulative risk. Furthermore, additional possible stressors from climate change, hypoxia, land-based activities, such as pollution or coastal development, and other ocean-based activities are not considered in this approach. However, one main advantage of our extended PSA is that it can provide a framework for assessing risk from multiple stressors.

Here, we proposed a simple tool for simultaneously assessing multiple, geographically co-occurring fisheries and establishing priorities for monitoring, management and conservation efforts. Risk-based frameworks do not require large amounts of detailed information, thus this method can be broadly applied to both industrial and small-scale fisheries, and in data-rich to data-poor settings. This extended PSA could also be used to assess individual and cumulative risk associated with developing new fisheries and provides an analytical approach that can be adapted to assess risk from multiple human activities. Furthermore, though risk-based approaches do not capture actual impacts, they can be used for exploring possible alternative approaches for decreasing risk to focal species or species groups of concern, or the possible benefits of changing gear types, modifying mesh sizes, and relocating fishing activities to different habitats or depths. Ultimately, our approach underscores the clear need and provides an approach to incorporate the cumulative effects of multiple human activities on marine and coastal systems for ecosystem-based fisheries and oceans management.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.biocon.2014.05.031.

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