BIODIVERSITY RESEARCH

Mapping the impact of alien species on marine ecosystems: the Mediterranean Sea case study

Stelios Katsanevakis[1,2]*, Fernando Tempera[1] and Heliana Teixeira[1]

[1]European Commission, Joint Research Centre (JRC), Institute for Environment and Sustainability (IES), Water Resources Unit, 21027 Ispra (VA), Italy, [2]Department of Marine Sciences, University of the Aegean, University Hill, 81100 Mytilene, Greece

ABSTRACT

Aim To develop a standardized, quantitative method for mapping cumulative impacts of invasive alien species on marine ecosystems.

Location The methodology is applied in the Mediterranean Sea but is widely applicable.

Methods A conservative additive model was developed to account for the Cumulative IMPacts of invasive ALien species (CIMPAL) on marine ecosystems. According to this model, cumulative impact scores are estimated on the basis of the distributions of invasive species and ecosystems, and both the reported magnitude of ecological impacts and the strength of such evidence. In the Mediterranean Sea case study, the magnitude of impact was estimated for every combination of 60 invasive species and 13 habitats, for every 10×10 km cell of the basin. Invasive species were ranked based on their contribution to the cumulative impact score across the Mediterranean.

Results The CIMPAL index showed strong spatial heterogeneity. Spatial patterns varied depending on the pathway of initial introduction of the invasive species in the Mediterranean Sea. Species introduced by shipping gave the highest impact scores and impacted a much larger area than those introduced by aquaculture and the Suez Canal. Overall, invasive macroalgae had the highest impact among all taxonomic groups. These results represent the current best estimate of the spatial variation in impacts of invasive alien species on ecosystems, in the Mediterranean Sea.

Main Conclusions A framework for mapping cumulative impacts of invasive alien species was developed. The application of this framework in the Mediterranean Sea provided a baseline that can be built upon with future improved information. Such analysis allows the identification of hotspots of highly impacted areas, and prioritization of sites, pathways and species for management actions.

Keywords

biological invasions, CIMPAL, cumulative impacts, indicators, invasive alien species, pathways.

*Correspondence: Stelios Katsanevakis, European Commission, Joint Research Centre (JRC), Institute for Environment and Sustainability (IES), Water Resources Unit, 21027 Ispra (VA), Italy.
E-mail: stelios@katsanevakis.com

INTRODUCTION

The idea of mapping the cumulative impact of human activities in the marine environment is a recent scientific endeavour that has been promoted by Halpern and colleagues, who applied it on a global level (Halpern *et al.*, 2008a) as well as regionally (Halpern *et al.*, 2009; Micheli *et al.*, 2013; Andersen *et al.*, 2015). As management and conservation of the oceans turns towards ecosystem-based spatial management (Borja, 2014), methods allowing for impact mapping are valuable to integrate spatial information for environmental management decisions and for setting explicit operational objectives.

New introductions of alien marine species have been accelerated in the recent decades by the rapid globalization and increasing trends of trade, travel and transport (Hulme,

2009; Katsanevakis *et al.*, 2013a). Alien marine species may become invasive with severe impacts on biodiversity and ecosystem services (Grosholz, 2002; Wallentinus & Nyberg, 2007; Molnar *et al.*, 2008; Katsanevakis *et al.*, 2014a). Ecological impacts of invasive alien species range from single-species interactions and reduction in individual fitness of native species to population declines, local extinctions, changes in community composition, and effects on entire ecosystem processes and wider ecosystem function (Blackburn *et al.*, 2014; Katsanevakis *et al.*, 2014a). Understanding, quantifying and mapping the impacts of invasive alien species across the seascape is a prerequisite for the efficient prioritisation of actions to prevent new invasions or for developing mitigation measures.

Hundreds of papers in the literature report ecological impacts of single or groups of alien marine species, more often on a single ecosystem in a specific location (see Katsanevakis *et al.*, 2014a for a European review). However, a comprehensive large-scale analysis of the cumulative impact of all alien marine species to all ecosystems is lacking, regionally or globally. Such analysis and subsequent mapping of impacts is urgently requested by policy makers and managers. For example, in the European Union (EU), the Marine Strategy Framework Directive – MSFD (EU, 2008) dictates that member states develop marine strategies and programs of measures to protect the marine environment and achieve 'good environmental status' in all marine waters by 2020. One of the requirements of good environmental status is that 'non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystems' (EU, 2008). Nevertheless, most of the EU member states failed so far to properly assess and quantify the impacts of invasive marine species in their initial assessments of the environmental status of their territorial waters (Palialexis *et al.*, 2014). This was largely due to the general lack of proper tools for mapping the impacts of alien species on ecosystems, at adequate scale (but see Zaiko *et al.*, 2011).

To assist management, we developed a standardized, quantitative method, on the basis of existing evidence, to map cumulative negative impacts of invasive alien species on marine ecosystems. This method was applied in the Mediterranean Sea as a case study. Impacts of invasive alien species were further analysed and mapped in relation to the main associated pathways of introduction in the Mediterranean Sea: Suez Canal, shipping, and aquaculture (Zenetos *et al.*, 2012). Mediterranean marine ecoregions are amongst the most impacted ecoregions globally by cumulative pressures (Halpern *et al.*, 2008a; Micheli *et al.*, 2013), and constitute one of the global hotspots of biological invasions, mainly because of the Suez Canal (Katsanevakis *et al.*, 2014b; Nunes *et al.*, 2014; Galil *et al.*, 2015).

The developed method offers a valuable new tool that may assist policy makers and managers in their efforts of developing strategies for mitigation of impacts of invasive species and improvement of the environmental status of marine waters. The method, although developed for the marine environment, can easily be transferred to the terrestrial environment as well.

METHODS

A framework for mapping Cumulative IMPacts of invasive ALien species (CIMPAL) on marine ecosystems was developed, based on a conservative additive model. A number of indicators were developed to compare the relative importance of species on cumulative impacts. Hereafter we describe the model and its components, as well as its application in the Mediterranean Sea.

Cumulative impact index

The study area has to be divided into equal-area cells. For each such cell cumulative impact scores I_c are estimated as $I_c = \sum_{i=1}^{n} \sum_{j=1}^{m} A_i H_j w_{i,j}$, where:

A_i is an index of the state of the population of invasive alien species i in the specific cell, transformed and normalized to range between 0 and 1. Abundance or relative abundance data are preferable for this state variable. The normalization function does not need to be linear but it would ideally reflect the actual relationship between abundance and impact (e.g. it could include a threshold above which $A_i = 1$ and another threshold below which $A_i = 0$). In the absence of such data, presence/absence data can also be used; in which case A_i will be binomial.

H_j is an index of the extent of habitat j in a specific cell, standardized to range between 0 and 1. Again if only presence/absence information of the habitat is available this will be a binomial index with 0 for absence and 1 for presence. The term 'habitat' is herein used as a recognizable space which can be distinguished by its physical characteristics and associated biological assemblage. Habitats are used here as the basic unit to identify impacts associated with individual invasive alien species, as they are easily defined spatially.

$w_{i,j}$ is the impact weight for species i and habitat j (the higher the impact of species i on habitat j, the higher the value of $w_{i,j}$).

n, m are the numbers of invasive alien species and marine habitats, respectively, that were included in the analysis.

Impact weights

Two approaches can be followed for the estimation of impact weights, representing two decision-making strategies: (1) an uncertainty-averse approach (Yemshanov *et al.*, 2013), when the decision makers show low-uncertainty preferences with respect to the impact index, i.e. when they prefer to invest the limited available funds and effort for mitigation measures of certain well-documented impacts; and (2) a precautionary approach, according to which we must assume impact even when the strength of evidence is low until we can no longer support that premise (Ojaveer *et al.*, 2015).

To apply the uncertainty-averse strategy (1), we propose that impact weights are estimated on the basis of both the reported magnitude of ecological impacts and the strength of such evidence (Fig. 1). Five semi-quantitative classes of magnitude – minimal, minor, moderate, major, and massive – are defined, following the Blackburn *et al.* (2014)

w_{ij}: impact weights for species i and habitat j

		Impact				
		Minimal	Minor	Moderate	Major	Massive
Strength of evidence	Robust	0	1	2	4	8
	Medium	0	0	1	2	4
	Limited	0	0	0	1	2

Strength of evidence

Robust	Impact is documented based on **Manipulative Experiments** (field or laboratory experiments that include treatments/control and random selection of experimental units) or **Natural Experiments** (one of the elements of manipulative experiments is missing and the experimental units are selected by nature, i.e. not randomly)
Medium	Impact is documented based on **Modelling** (i.e. as derived from ecosystem models), **Direct Observations** (an observation or direct measurement of the impact about which there is no doubt, but which is not based on experimental studies, e.g. large-scale mortality events because of harmful algal blooms), or **non-experimental-based Correlations** (inference based on an observed correlation between the species' presence/abundance and the impact, but not based on an experimental design for data collection)
Limited	Impact is based on **Expert Judgement**, usually on the basis of empirical knowledge, the species' traits, or the documented impact of similar species

Impact

Minimal	No effect on fitness of native species; negligible impact on native species due to competition, predation, parasitism, toxicity, bio-fouling, or grazing/herbivory; negligible impact on ecosystem processes and ecosystem functioning; negligible impact on keystone species or species of high conservation value; no chemical, physical or structural impact on the ecosystem (not an ecosystem engineer).
Minor	Reduction in individual fitness due to competition, predation, parasitism, toxicity, bio-fouling, or herbivory, but no substantial population declines; minor impact on ecosystem processes and ecosystem functioning with no related population declines; negligible impact on keystone species or species of high conservation value; or causes changes in chemical, physical or structural habitat characteristics without decline of native populations.
Moderate	Declines in population densities because of competition, predation, parasitism, toxicity, bio-fouling, or herbivory, but no changes in community composition; or displacement of no more than one species of similar niche; or impact on ecosystem processes and ecosystem functioning resulting in population declines but no substantial change in species composition; or reduction in individual fitness of at least one keystone species or species of high conservation value, but no substantial population declines; or ecological engineering, resulting in population declines but no substantial change in community composition.
Major	Changes in community composition and local or population extinction of at least one native species, because of competition, predation, parasitism, toxicity, bio-fouling, or herbivory; impact on ecosystem processes and ecosystem functioning resulting in species composition changes; or population decline of at least one keystone species or species of high conservation value; or ecological engineering, resulting in change in community composition. Induced changes are reversible in the short term (<1 decade) with proper management measures or if the alien species population declines naturally.
Massive	The same as in 'major' but changes are irreversible in the short term (<1 decade) or currently there is no known effective management action for the control of the invasive alien species and a natural decline of its population seems highly unlikely.

Figure 1 Impact weights defined on the basis of the magnitude of impact and the related strength of evidence (uncertainty-averse approach). Classification of the magnitude of impacts is based on Blackburn *et al.* (2014), adapted for the marine environment. The categories of type of evidence follow Katsanevakis *et al.* (2014a). For the precautionary approach, $w_{i,j}$ are estimated solely from the first line of the matrix, i.e. assuming 'robust' strength of evidence for all species.

framework, adapted for the assessment of impacts on marine ecosystems (Fig. 1). Evidence for most of the reported impacts of marine alien species in the literature is weak, mostly based on expert judgement or dubious correlations (Katsanevakis *et al.*, 2014a). Hence, we propose to downgrade the weights of such impacts with low supporting evidence, in comparison to impacts documented by manipulative or natural experiments. In the scheme we propose, impact weights range between 0 and 8 depending on the combination of reported magnitude and level of evidence (Fig. 1). For the precautionary approach (2), there is no downgrading of the impacts with low supporting evidence, but $w_{i,j}$ are estimated solely on the basis of the reported magnitude of ecological impacts. In this case, we suggest that uncertainty associated with the perceived magnitude of impacts is reported separately (see Appendix S1 in Supporting Information).

We herein assume that $w_{i,j}$ are spatially constant and are thus calculated according to the highest impact reported, which is a precautionary simplification in the absence of information on the variation of impacts in space. Nevertheless, this assumption can be relaxed if such information is available and thus $w_{i,j}$ can vary spatially.

Mediterranean case study

We applied the developed methodology in the entire Mediterranean Sea. A 10 × 10 km standard grid covering the Mediterranean basin was used (Lambert equal-area projection); the total number of cells was 26,890. We assembled spatial datasets for 13 habitats (H_j): *Posidonia oceanica* meadows, coralligenous communities, marine caves, sandy beaches, rocky intertidal, shallow sublittoral soft bottoms (<60 m depth), circalittoral soft bottoms (60–200 m), bathyal/abyssal soft bottoms (>200 m), shallow sublittoral hard bottoms (<60 m), circalittoral hard bottoms (60–200 m), bathyal/abyssal hard bottoms (>200 m), shallow pelagic (over seafloor with less than 60 m depth) and deep pelagic (over seafloor with more than 60 m depth). Layers for the first three habitats were sourced from Giakoumi *et al.* (2013), the last two were derived from EMODNET bathymetric data (http://www.emodnet-hydrography.eu), and the rest originated from Micheli *et al.* (2013) (see Fig. S1). For most of these habitats the available resolution was very coarse compared to the actual extents covered by the habitat (e.g. for *Posidonia* and coralligenous communities, only presence in 10 km × 10 km cells was available). Hence, the estimation of a reliable index of extent (e.g. percent coverage of benthic area) was not possible at a basin-wide level. We thus simplified H_i to a binomial index with 0 for absence and 1 for presence.

In a recent pan-European review of the impacts of alien marine species on ecosystem services and biodiversity (Katsanevakis *et al.*, 2014a), an inventory of 86 marine species of high documented impact was reported, representing 13 phyla and all major marine groups (phytoplankton, macro-phytes, polychates, crustaceans, molluscs, ascidians, bryozoans, cnidarians, ctenophores, echinoderms and fish; see Katsanevakis *et al.*, 2014a for the full list). From this inventory, we excluded all those species that are not present in the Mediterranean Sea, ending up with a subset of 60 species. This subset represents all marine invasive alien species that are present in the Mediterranean and for which there are documented impacts in the literature. Based on the latter review, we assessed the magnitude of impacts of each of these 60 species on each of the 13 herein assessed habitats in the Mediterranean Sea. As the aim of this work was to quantify and map the negative impacts of alien species on biodiversity, any positive impacts were not taken into account. In Katsanevakis *et al.* (2014a), the type of evidence of each impact was assessed by categorizing evidence into the following six categories: manipulative experiments, direct observations of impact, natural experiments, modelling, non-experimental-based correlations and expert judgement. This information allowed us to estimate for each of the 60 species and for each of the assessed habitats the impact weight $w_{i,j}$ as defined in Fig. 1 (see Table S1). We estimated $w_{i,j}$ and the related CIMPAL scores following both the uncertainty-averse strategy and the precautionary one, to compare the resulting cumulative impact maps. However, for the rest of the analyses we applied only the uncertainty-averse approach.

Distribution maps of the 60 included alien marine species on the 10 × 10 km grid (A_i; presence/absence data) were retrieved from the European Alien Species Information Network – EASIN (Katsanevakis *et al.*, 2015). These spatial data, integrated in EASIN, originate from the following sources: (1) the CIESM Atlas of Exotic Species (http://www.ciesm.org/online/atlas/index.htm); (2) the Global Biodiversity Information Facility (GBIF; http://www.gbif.org/); (3) the Global Invasive Species Information Network (GISIN; http://www.gisin.org); (4) the Regional Euro-Asian Biological Invasions Centre (REABIC; http://www.reabic.net/); (5) the Hellenic Network on Aquatic Invasive species (ELNAIS: elnais.hcmr.gr); and (6) EASIN-Lit (http://easin.jrc.ec.europa.eu/About/EASIN-Lit; Trombetti *et al.*, 2013).

Species were linked to one or in some cases to two most probable pathways of initial introduction in the Mediterranean Sea, following Zenetos *et al.* (2012) and Katsanevakis *et al.* (2013a). According to these authors, the most important pathways of initial introduction in the Mediterranean Sea are: (1) the 'Suez Canal', which refers to species of Indo-Pacific origin progressively introduced unassisted into the Mediterranean via the Suez Canal (also called Lessepsian immigrants); (2) 'shipping', which refers either to the transportation of holoplanktonic or meroplanktonic organisms, seeds or resting stages (e.g. cysts or eggs) in ballast water, or to the transportation of predominantly sedentary species that attach to ship hulls; and (3) 'aquaculture', which refers either to commercial species that were introduced to be cultured or to species accidentally introduced together with imported target species.

The correlation between alien species richness and the CIMPAL score was estimated and its significance tested, based on a modified t-test as described by Clifford *et al.* (1989). The modified t-test is based on corrections of the sample correlation coefficient between the two spatially correlated sequences, properly accounting for spatial autocorrelation in the data.

Four indicators were estimated to compare the relative importance of species on cumulative impacts across the Mediterranean basin: (D_1) the total area of occurrence (as total number of 10×10 km cells); (D_2) the number of cells with impact >0; (D_3) the sum of impact scores of the species across the entire study area; (D_4) the average impact across the range of occurrence (i.e., the average impact score of a species, estimated across the cells where the species is present).

RESULTS

### Estimation and mapping of CIMPAL – prioritization of sites, pathways and species

In the Mediterranean Sea, the proposed CIMPAL index, based on the uncertainty-averse strategy, showed strong spatial heterogeneity (Fig. 2) and varied between 0 and 149 (Fig. S2). Impact was largely restricted to coastal areas, and the cumulative impact index was zero in most offshore cells (Fig. 2). Without downgrading impacts with limited or medium strength of evidence, i.e. by following the precautionary approach, the overall magnitude of the CIMPAL score increased and varied between 0 and 188 (Fig. 3). In addition to the differences in magnitude, the two approaches generated significantly different distribution patterns and site ranking (non-parametric Wilcoxon Signed-Rank Test, $P < 0.001$; see Appendix S2 and Figs S6 and S7). By following the precautionary approach, some additional hotspots of alien species impacts came up, notably the eastern Mediterranean coastline (Fig. 3). Confidence on the CIMPAL index estimates was higher for the north-western part of the Mediterranean and the Adriatic Sea than for the southern Mediterranean coastline and the Aegean Sea (Fig. S3).

The spatial patterns of impact varied strikingly by pathway of initial introduction. Species introduced through the Suez Canal mostly impacted the eastern parts of the basin; those introduced by shipping had the highest impacts in many central and north-western sites; and two high-impact areas were evident in the Italian peninsula due to species
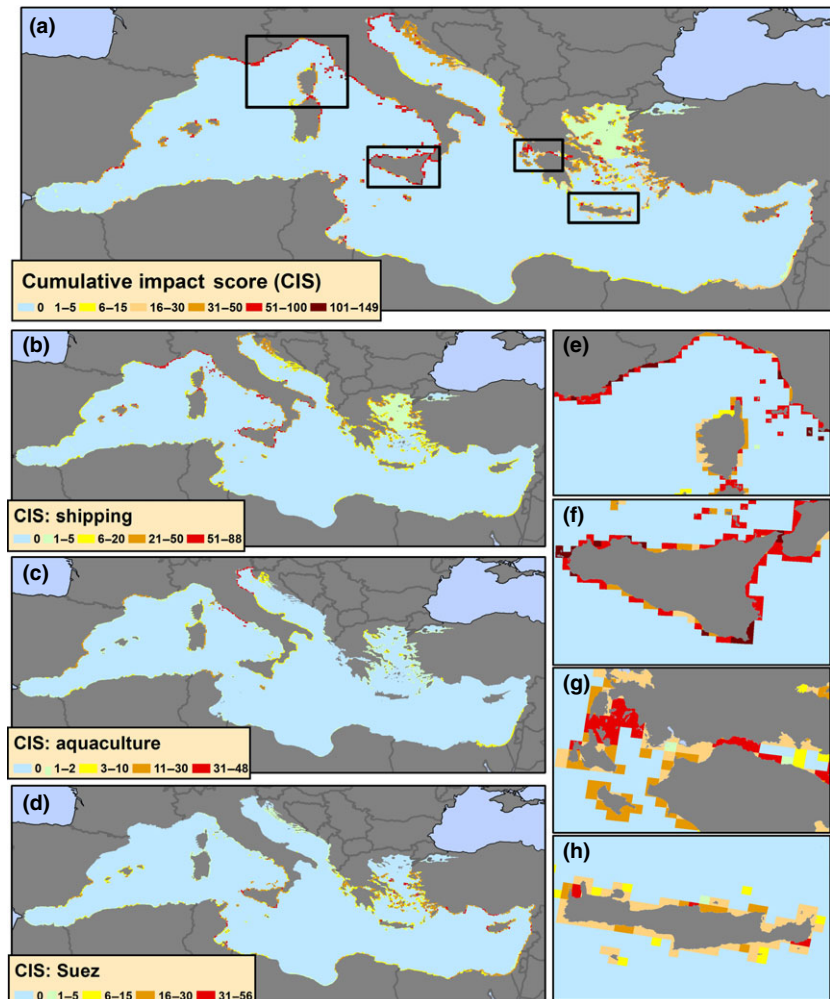
Figure 2 Mediterranean Sea map (a) of the cumulative impact (CIMPAL) score of 60 invasive alien species to 13 marine habitats, based on the uncertainty-averse strategy. Maps of cumulative impact scores to the same marine habitats by species likely introduced by shipping (b), aquaculture (c), and through the Suez Canal (d). Magnifications of the Ligurian Sea and Corsica (e), Sicily (f), the Greek Ionian Archipelagos and adjacent gulfs (g), and Crete (h).
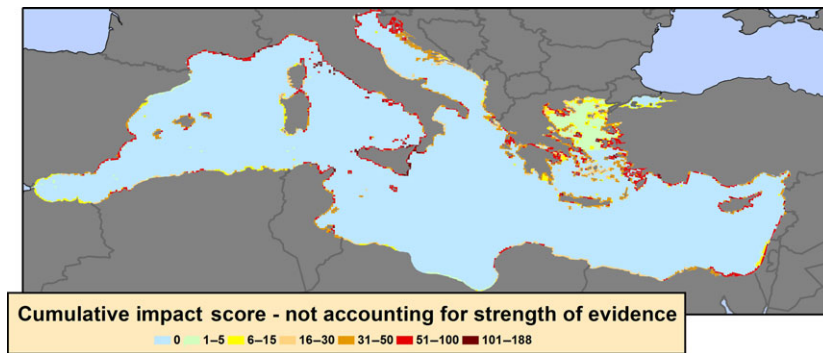
Figure 3 Mediterranean Sea map of the cumulative impact (CIMPAL) score, without accounting for the strength of evidence of reported impacts, i.e. following the precautionary strategy, which considers that all impacts have 'robust' strength of evidence (*sensu* Fig. 1).

introduced by aquaculture (Fig. 2). Species introduced by shipping exhibited the highest impact scores, with a maximum value of 88. Ignoring the zero values (mainly offshore waters), CIMPAL scores showed multimodal distributions with the main peak at low values (1 or 2) (Fig. S2). This high peak at low values represented ~60% of the cells in the case of species introduced by aquaculture, while it was markedly lower for impact scores of species related to shipping or the Suez Canal (25% and 22%, respectively). The marine area (i.e. total number of cells) impacted by shipping-related alien species was much larger (4047 cells) than the areas impacted by alien species introduced by aquaculture or through the Suez Canal (2504 and 2900 cells, respectively). The relative contribution of each pathway of introduction to the total cumulative impact of each country varied a lot (Table S2). Italy and Greece were the countries with the highest share of the total cumulative impact estimated across the basin.

The inventory and ranking of the most impacting species varied depending on the indicator used (Fig. 4). D_1 reflects merely the total invaded area in the sea basin, and is less relevant to assess impacts. Alien species richness (Fig. 5) and the estimated CIMPAL score (Fig. 2) exhibit quite different spatial patterns and their correlation was non-significant ($P = 0.57$; based only on cells adjacent to the coast, to avoid zero inflation produced by offshore cells). The five species with the highest occupancy were all macrophytes. When only cells with impacts to at least one of the 13 assessed habitats were included (indicator D_2), the rank changed substantially, e.g. *Asparagopsis armata* fell from the 1st to the 16th position. Based on D_3, which accounts not only for the number of impacted cells but also for the magnitude of impact, three macroalgae species (*Caulerpa cylindracea*, *Lophocladia lallemandii* and *Womersleyella setacea*) have markedly higher impact than the rest. Indicator D_4 expresses the magnitude of the impact in the invaded locations, and indicates the importance of species in terms of impacts at local scale rather than across the basin. Some species such as the bivalves *Crassostrea gigas* and *Venerupis philippinarum* appear to have very high impacts locally (1st and 3rd respectively, based on D_4) but less impact basin-wide (7th and 11th respectively, based on D_3). Species that were not included in the D_1, D_2, and D_3 top-20 lists but appear in the D_4 top-20

list indicate high impact but on a relatively small scale (*Crepidula fornicata*, *Ficopomatus enigmaticus*, *Hydroides elegans*, *Rhopilema nomadica*, *Gymnodinium catenatum*) (Fig. 4).

The maximum potential impact on each habitat was estimated as the sum of all impact weights of all alien species for this habitat, i.e. $\Sigma_i w_{i,j}$, which is the estimated cumulative impact score if all species impacting the specific habitat would be present in a cell. The habitat with the highest potential number of impacts was 'shallow sublittoral hard bottoms' (with $\Sigma_i w_{i,j} = 110$), followed by 'shallow sublittoral soft bottoms' ($\Sigma_i w_{i,j} = 54$), and 'rocky intertidal' ($\Sigma_i w_{i,j} = 46$). No impacts have been reported for 'marine caves', 'bathyal/abyssal soft bottoms', 'circalittoral hard bottoms', and 'bathyal/abyssal hard bottoms'. The present average cumulative impact was, in general, much lower than the maximum potential impact risk (Table 1). In some habitats, such as *Posidonia* meadows and coralligenous communities, the highest potential cumulative impact score was actually observed in some cells (in 42 cells for *Posidonia* meadows and in 25 cells for coralligenous communities). In all other habitats the maximum observed cumulative impact score was much lower than the maximum potential score (Table 1). Impacts on *Posidonia* meadows and coralligenous communities were reported with robust evidence (i.e., based on experimental studies), while there was overall much less evidence for reported impacts on shallow sublittoral soft or hard bottoms and rocky intertidal habitats (Table 1).

By disaggregating the CIMPAL score by habitat, further insight on the habitat-specific spatial variation of impact can be given. For example, the impacts on *Posidonia oceanica* and coralligenous communities – two ecosystems of high conservation importance – exhibit different spatial patterns that are controlled by the distinct distributions of the habitats and the impacting species, although there are some common hotspots (Fig. 6).

DISCUSSION

Our results represent the best estimate of the spatial variation in impacts of invasive alien species on ecosystems, currently available for the Mediterranean Sea. Our approach to estimate cumulative impacts of invasive alien species and the
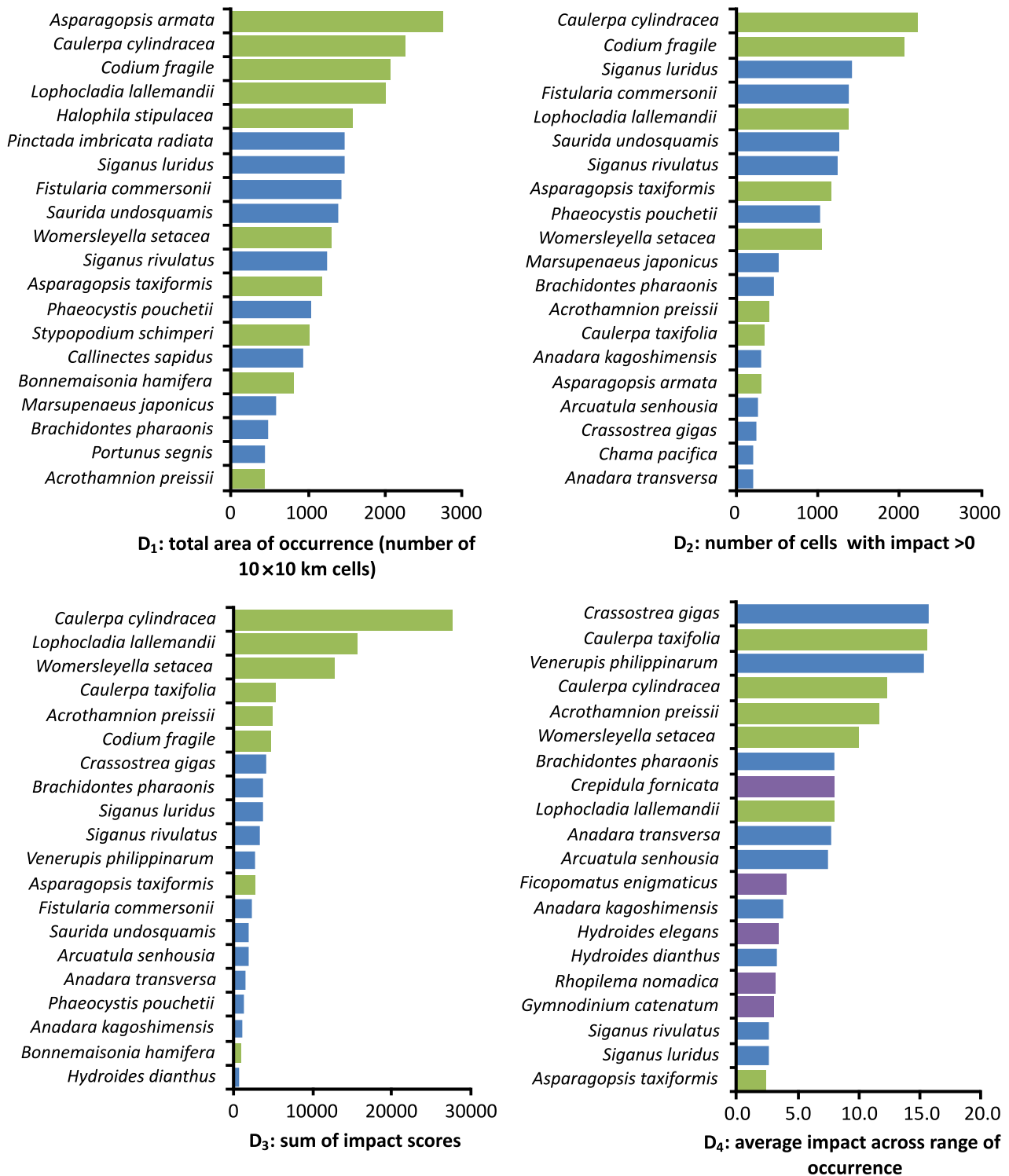
Figure 4 Relative importance of the 60 high-impact species as assessed by four indicators D_1–D_4 (only the top 20 species are shown in the charts). Macrophytes are coloured green; species that appear only in the D_4 top-20 graph and not in any other are coloured purple.

related species-specific indicators offer the means to: (1) identify hotspots of highly impacted areas; (2) assess the relative importance of pathways of initial introduction to the cumulative impact and its spatial variation; (3) rank invasive alien species according to the large-scale or local importance of their impacts; and (4) prioritize areas/pathways/species/

habitats for management actions and mitigation measures. Specifically for EU countries, the CIMPAL score is a valuable indicator that can be used alongside other indicators for the estimation of the environmental status of marine waters, as dictated by the Marine Strategy Framework Directive (EU, 2008).
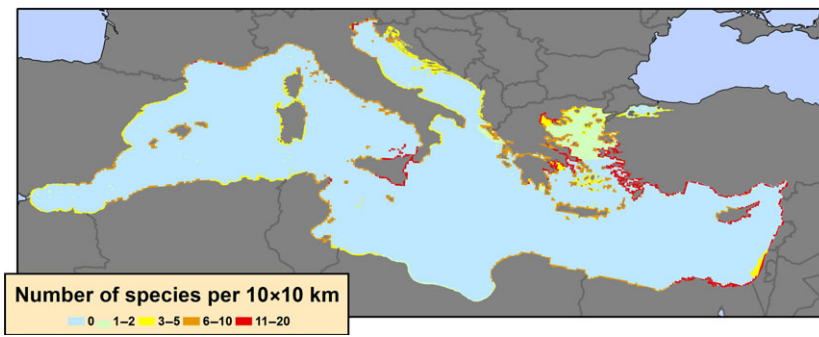
Figure 5 Richness (number of species per 10 × 10 km cell) of the 60 assessed alien marine species (see Table S1) in the Mediterranean Sea. The map is based on EASIN (European Alien Species Information Network) data.

Table 1 Cumulative impact results for specific habitats in the Mediterranean Sea. Values in brackets indicate the maximum potential cumulative impact (I_c) on biodiversity estimated for each habitat *j*, estimated as $\Sigma_i w_{i,j}$ (see Table S1). The last row refers to the percentage of reported species impacts on each habitat that has high strength of evidence (high confidence).

	Habitats of conservation interest		Habitats with highest potential impact risk		
	Posidonia meadows	Coralligenous comunities	Shallow sublittoral hard-bottom	Shallow sublittoral soft-bottom	Rocky intertidal
Maximum I_c observed (potential)	27 (27)	40 (40)	50 (110)	29 (54)	19 (46)
Average I_c across range of occurrence	6.8	8.4	10.9	5.1	4.7
Sum of I_c scores in the Mediterranean	22994	19480	9748	27035	18077
% impacted habitat in the Mediterranean	59.2	46.1	46.8	55.4	86.3
% of impacts with high confidence	83	100	36	31	21

The various assumptions and limitations to our analysis are discussed below with concerns being separated in data availability, model framework and evidence of IAS impacts. The implications of the current results for management are discussed under the 'Foreseeable action' heading.

Data concerns

Habitat data varied in quality, both among and within classes. This spatial heterogeneity in ecological data quality might have led to falsely low CIMPAL scores in data poor areas failing to highlight hotspots for management. When only presence/absence of habitats is known per grid cell, the weight of any habitat present will be the same, no matter if its coverage is low or high. In cells with multiple low coverage habitats, as is the case of highly vulnerable and patchy habitats, summing across habitats forces cells to have higher potential scores than cells with fewer habitats. The size of the grid cell used herein was 100 km², implying that single cells, especially near the coast, have often several habitats present. Some studies of cumulative human impacts have used a much finer scale, e.g. 200 × 200 m grid cell size in Ban *et al.* (2010) and 267 × 267 m grid cell size in Korpinen *et al.* (2013), which were essentially the pixels of their habitat maps. With such a fine scale the inflation of cumulative impact scores due to the presence of multiple habitats in a cell and using presence/absence instead of coverage would largely be overcome. However, both the habitat data and species distribution data were unavailable at such a fine scale in the entire Mediterranean basin. This technical inflation of cumulative impact scores is nevertheless acceptable under a precautionary perspective. Whilst this paper demonstrates the application of the CIMPAL to identify potential impact hotspots at regional level, managers working at more local scales will highly benefit from higher resolution habitat extent data per grid cell to establish their priority areas.

The broad habitat categories herein cover the main environmental gradients (substrate nature, light) and habitats of conservation importance. However, there are many more detailed habitat classifications in the Mediterranean Sea (EUNIS, 2002; Fraschetti *et al.*, 2008 and references therein) that could enhance habitat resolution to as many as ~100 different classes. Using different classifications would modify the results, as the relative contribution of habitats would change. However, maps of highly resolved habitats are not presently available for the whole Mediterranean basin. Using them would imply very broad data gaps that would further curtail the credibility of the assessment presented.
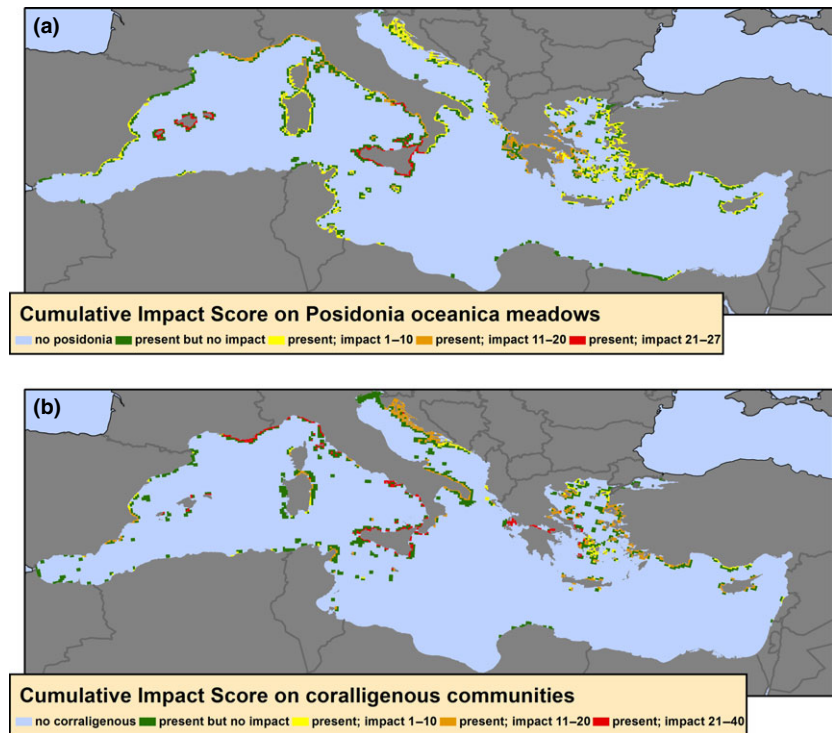
Figure 6 Cumulative impact of alien species on *Posidonia oceanica* meadows (a) and coralligenous communities (b), based on the uncertainty-averse strategy.

Additional bias may have been introduced by the uneven quality of alien species distributional data collected from different Mediterranean countries and sectors. Monitoring and reporting effort is variable (Katsanevakis *et al.*, 2013b) and cumulative scores in data poor regions are likely underestimated. The lack of high-quality widespread information throughout the basin made us use presence-only data as a status variable of the alien species in the present exercise. This implied that the impact of each species was taken as uniform across its reported range, although in reality the abundance of each species varies considerably across its distributional range. Where abundance data are to be used instead, deciding how to transform and normalize the index A_i of the state of the population may have important consequences for the resulting impact assessments. Such decisions would include whether skew in data is preserved, minimized or removed, and choosing a maximum value to set equal to 1.0 (Halpern & Fujita, 2013). The latter would necessitate knowledge of threshold values in the impacts of alien species on habitats, but these are currently largely unknown.

Very few invasive alien species have been reported in offshore areas (Fig. 5; see also Katsanevakis *et al.*, 2014b), which can be partly explained by the fact that all important vectors of alien species (Suez Canal, ships and aquaculture) operate in shallow waters, thus assisting the introduction of shallow-water species. Most of the recorded marine aliens in the Mediterranean are indeed shallow-water thermophilic demersal species (Katsanevakis *et al.*, 2014b). Nevertheless, there is a reduced sampling effort offshore, causing a monitoring and reporting bias in favour of coastal areas

(Danovaro *et al.*, 2010). Furthermore, studies on the impacts of alien species are generally restricted in coastal waters. Possible impacts of alien species on offshore habitats are largely unknown.

Among the nearly 1000 alien and cryptogenic species so far reported for the Mediterranean Sea (Zenetos *et al.*, 2012), impacts on biodiversity and ecosystem services have been documented for only 60 species (review by Katsanevakis *et al.*, 2014a). Although there are probably more species with impacts on marine biodiversity, which have not been studied yet, there is sufficient confidence that this set of 60 species includes most or all invasive species with very high and large-scale impacts.

Due to necessity and data limitation, it was assumed that the impact weights $w_{i,j}$ are spatially constant, and hence a specific habitat would respond the same way to a specific alien species at any location. However, this is not necessarily true for all species-habitats combinations, especially for a coarse classification of habitats as the one used herein. Among the various ecoregions of the Mediterranean and even within the same ecoregion, habitats such as 'shallow sublittoral hard bottoms' or 'coralligenous communities' correspond to a variety of biological assemblages, and thus would have inherently different responses to the same intensity of stressors by alien species. In the absence of such information, spatial and temporal variation in impacts was not taken into account, but only the highest impact reported was considered. This would cause the overestimation of the CIM-PAL index in some cells, which could be acceptable under the precautionary strategy for decision-making. In that sense, the estimated CIMPAL score is an index of potential impact

of aliens on habitats, which however might lead to false prioritization of some sites for management measures.

Model framework

In the absence of quantitative or statistically comparable data of cumulative impacts as is usually the case, an impact-scoring system (as the one used herein) can be used to make diverse data comparable and to allow comparisons between groups with different impact mechanisms (Kumschick *et al.*, 2015). Several other scoring systems of single-species environmental impacts have been proposed, e.g. the Generic Impact Scoring System – GISS (Nentwig *et al.*, 2010) and the Biopollution Level Index (Olenin *et al.*, 2007). Any such indicator could replace our proposed impact weight $w_{i,j}$. Although we would not expect substantial differences in the spatial patterns of cumulative impact, a comparison of the outputs based on different impact scores would be an interesting topic for future research.

The level of downscaling attributed to the different categories of evidence of impacts reflects our judgement on their validity and robustness, and is based on Katsanevakis *et al.* (2014a). However, in the absence of real quantification of the confidence in the available evidence of impacts, any other incremental scale could be adopted if deemed more suitable. We estimated the CIMPAL index by using different scales for the weights – a linear scale (1, 2, 3, 4) and a logarithimic scale (1, 10, 100, 1000) – and we produced maps that were very similar (especially when using the linear scale) to the ones presented in Fig. 2 (see Figs S4 and S5) in terms of the identified main hotspots and the spatial variation of impacts. Hence, although our scale for the impact weights is arbitrary, we do not expect any important implications if the mapping of cumulative impacts is based on a different scale.

Although we used a conservative additive model, synergistic effects might exist among some species. The cumulative impact of all species on a specific habitat may be much greater, or in some cases less, than the sum of individual impacts, because of interactive or multiplicative effects (Halpern *et al.*, 2008b). Such information is generally lacking, but if available it can be easily incorporated in the cumulative impact score formula by adding interactive terms. Hence, the CIMPAL score would be:

$$I_c = \sum_{i=1}^{n}\sum_{j=1}^{m} A_i H_j w_{i,j} + \sum_{i=1}^{n}\sum_{k=i+1}^{n}\sum_{j=1}^{m} A_i A_k H_j f_{i,k,j}(A_i, A_k),$$

where f is the function of the interaction between species i and species k in terms of their cumulative impact on habitat j, which would be generally a function of the status A_i and A_k of species i and k respectively. It can be positive, when impacts are multiplicative, or negative, when impacts are mitigative. Yet, in the absence of knowledge about where, when or why interactive effects occur, the default additive model remains the only feasible option (Halpern & Fujita, 2013). In the Katsanevakis *et al.* (2014a) review of the

impacts of alien marine species on ecosystem services and biodiversity in the Mediterranean, no study investigating synergistic effects between two or more invasive alien species was found. Understanding interactions among alien species is an open field of research, and for some combinations of species important interactions are anticipated (e.g. between *Siganus* spp. and *Caulerpa cylindracea* due to the grazing of the latter by the former).

An interesting topic also deserving further research would be the assessment of positive cumulative impacts of alien species. Alien species often benefit some components of native biodiversity or have negative effects on biodiversity within one trophic level but positive effects on the biodiversity of higher trophic levels (Katsanevakis *et al.*, 2014a; Thomsen *et al.*, 2014). Alien ecosystem engineers can create novel ecosystems that fulfil important roles that might otherwise be lost in degraded systems (Hobbs *et al.*, 2009). Hence, it would be interesting to compare the spatial variation in negative impacts of invasive alien species on ecosystems with the spatial variation of positive impacts, and assess the overall balance of all impacts.

In their global assessment of cumulative impacts of human activities, Halpern *et al.* (2008a) modelled the incidence of invasive species as a function of the amount of cargo traffic at a port, in the absence of actual data for the global distribution of invasive species. Micheli *et al.* (2013), in their Mediterranean assessment of cumulative impacts, improved the previous approach by replacing that layer with data on the actual distribution of a subset of invasive species in the Mediterranean. Although this is a substantial improvement, as herein shown, there is no significant correlation between the aggregated species distribution and their cumulative impacts. Hence, the use of the herein proposed CIMPAL index would further improve the invasive species layers in assessments of cumulative impacts of human activities, such as the study by Micheli *et al.* (2013) in the Mediterranean Sea.

Evidence of cause-effect for IAS

Most of the reported impacts of alien marine species on biodiversity are not supported by studies of high inferential strength. In our case study, 29%, 38% and 33% of reported species-habitats impacts were supported by studies of 'limited', 'medium' and 'robust' inferential strength respectively. Simple correlations (not based on experimental data) or mere expert judgement are insufficient to discriminate between the effect of an alien species and the cumulative effects of all the other human stressors or natural variability. The decline of natives within a community and the dominance of aliens may be a consequence of, rather than the driving force behind, ecosystem disturbance (Chabrerie *et al.*, 2008), as some alien species can better tolerate disturbance due to their generalist ecology and phenotypic plasticity (Smith, 2009; Goodenough, 2010). There is a tendency by many marine ecologists to exaggerate about the possible

impacts of alien species, being prejudiced by a 'native good, alien bad' perception (Goodenough, 2010). Expert judgements may be influenced by value-laden opinions, lack of experience, and conflicts of interest, and are sensitive to a host of psychological idiosyncrasies and subjective biases, often leading to overestimation of alien species impacts (Katsanevakis et al., 2014a). Hence, downgrading the impact weights (as in the uncertainty-averse strategy) seems reasonable and realistic, especially when results are meant to guide management decisions in prioritizing actions to direct the very limited available funds. Management and cost implications of false positives (i.e. assigning an impact when in fact there is none) are expected to be higher for impacts of higher stated magnitude. Hence, it makes sense to penalize stronger the impact weights (in absolute values) among the strength of evidence categories at higher impact levels than at lower ones (see Fig. 1).

Foreseeable action

Despite the above-mentioned limitations (mostly related to data availability), our analysis provides a framework and a baseline that can be built upon with future improved information. As additional spatial data on the distribution of habitats and species become available, they could be incorporated in new iterations of the analysis. Data gaps emphasize the need for further research on basic information such as habitat maps and the spatial distribution of species abundance. Any assessment of cumulative impacts faces the challenge of missing data. It is common to argue that assessments should not be conducted and policy makers should put off making important decisions when key gaps exist and available information suffers from high levels of uncertainty. But such a 'wait-until-uncertainty-is-greatly-reduced' approach is often unproductive as it will keep assessments from ever happening, since there will always be key gaps when conducting comprehensive large-scale assessments (Halpern and Fujita, 2013). Biological invasions in the Mediterranean are highly dynamic with one new species arriving approximately every two weeks, and new species continuously expanding their range (Zenetos et al., 2012). The 60 species used in this study were introduced between 1865 and 2003, with the exception of one species (Teredo navalis) that was introduced in 1792 (based on information on dates of introduction from EASIN; Katsanevakis et al., 2015), and thus many of them have not been long enough in the Mediterranean to occupy all available suitable niches. New impacts appear and a significant time-lag is expected between their appearance and the publishing of relevant documentation. Hence, even if an astronomic amount of money was invested today to minimize uncertainty by gaining high-quality information for the distribution of habitats and species and good knowledge on all alien species/habitats interactions, this would be outdated after some years. All decisions about complex natural resource management problems will include some degree of uncertainty, but postponing any action in a chase of certainty will lead to decision paralysis and can cause harm in many fragile ecosystems threatened by cumulative impacts (Kellon & Arvai, 2011).

Adaptive management is a way out of the trap of decision paralysis (Kellon & Arvai, 2011). Adaptive management should be perceived as managing according to a plan by which decisions are made and modified as a function of what is known and learned about the system, including information about the effect of previous management actions (Ludwig et al., 1993; Parma, 1998). Adaptive management of biological invasions should focus on monitoring, filling data gaps, and learning as the system changes, due to the dynamic nature of invasions but also in response to managing actions. Furthermore, to deal with uncertainty adaptive management calls for the proper design and monitoring of planned 'policy experiments', with control and replication of management treatments at appropriate spatial and temporal scales (Kellon & Arvai, 2011).

There is no consensus as to which strategy should be followed for the prioritization of actions to prevent new invasions or for developing mitigation measures. Many regulatory bodies and environment agencies aspire to evidence-based (essentially uncertainty-averse) policy and practise, while on the other hand it is argued that management that focuses solely on species known to cause harm fails to allow for the management of the unknowns (Ojaveer et al., 2015). The uncertainty-averse strategy will save funds, which is of utmost importance in a limited-funding environment for conservation, but according to the latter authors 'will inevitably cause more damage (and/or costs) as unanticipated invasions occur'. Nevertheless, in the current reality of minimum funding and inadequate mitigation measures in the Mediterranean Sea, taking action to address all known and unknown impacts seems unrealistic. The discussion is not about optimizing an existing ambitious and well-funded basin-wide strategy to mitigate the impacts of invasive species but rather to make a start and direct the limited available funds to developing and implementing mitigation measures for a handful of species/sites. Towards this direction, we believe that the uncertainty-averse approach is the safe way to guide decision-making for the prioritization of sites, pathways and species. The areas highlighted in Fig. 2 are those with high impacts/high certainty – exactly what is needed to inform management. If the safe way is followed the chances for success stories will be increased. Such successes are needed especially in the first steps of a basin-wide strategy to mitigate the impacts by biological invasions, to encourage further efforts.

REFERENCES

Andersen, J.H., Halpern, B.S., Korpinen, S., Murray, C. & Reker, J. (2015) Baltic Sea biodiversity status vs. cumulative human pressures. *Estuarine, Coastal and Shelf Science*, **161**, 88–92.

Ban, N.C., Alidina, H.M. & Ardron, J.A. (2010) Cumulative impact mapping: advances, relevance and limitations to marine management and conservation, using Canada's Pacific waters as a case study. *Marine Policy*, **34**, 876–886.

Blackburn, T.M., Essl, F., Evans, T. *et al.* (2014) A unified classification of alien species based on the magnitude of their environmental impacts. *PLoS Biology*, **12**, e1001850.

Borja, A. (2014) Grand challenges in marine ecosystems ecology. *Frontiers in Marine Science*, **1**, 1.

Chabrerie, O., Verheyen, K., Saguez, R. & Decocq, G. (2008) Disentangling relationships between habitat conditions, disturbance history, plant diversity, and American black cherry (*Prunus serotina* Ehrh.) invasion in a European temperate forest. *Diversity and Distributions*, **14**, 204–212.

Clifford, P., Richardson, S. & Hémon, D. (1989) Assessing the significance of the correlation between two spatial processes. *Biometrics*, **45**, 123–134.

Danovaro, R., Company, B.J., Corinaldesi, C., D'Onghia, G., Galil, B., Gambi, C., Gooday, A.J., Lampadariou, N., Luna, G.M., Morigi, C., Olu, K., Polymenakou, P., Ramirez-Llodra, E., Sabbatini, A., Sardà, F., Sibuet, M. & Tselepides, A. (2010) Deep-sea biodiversity in the Mediterranean Sea: the known the unknown and the unknowable. *PLoS ONE*, **5**, e11832.

EU (2008) Directive of the European Parliament and the Council Establishing a Framework for Community Action in the Field of Marine Environmental Policy (Marine Strategy Framework Directive). European Commission. Directive 2008/56/EC, OJ L 164.

EUNIS (2002). European Nature Information System (EUNIS) Habitat Classification. European Environment Agency. http://eunis.eea.eu.int.

Fraschetti, S., Terlizzi, A. & Boero, F. (2008) How many habitats are there in the sea (and where)? *Journal of Experimental Marine Biology & Ecology*, **366**, 109–115.

Galil, B., Boero, F., Fraschetti, S., Piraino, S., Campbell, M., Hewitt, C., Carlton, J., Cook, E., Jelmert, A., Macpherson, E., Marchini, A., Occhipinti-Ambrogi, A., Mckenzie, C., Minchin, D., Ojaveer, H., Olenin, S. & Ruiz, G. (2015) The Enlargement of the Suez Canal and Introduction of Non-Indigenous Species to the Mediterranean Sea. *Limnology and Oceanography Bulletin*, **24**, 41–43.

Giakoumi, S., Sini, M., Gerovasileiou, V., Mazor, T., Beher, J., Possingham, H.P., Abdulla, A., Çinar, M.E., Dendrinos, P., Gucu, A.C., Karamanlidis, A.A., Rodic, P., Panayotidis, P., Taskin, E., Jaklin, A., Voultsiadou, E., Webster, C., Zenetos, A. & Katsanevakis, S. (2013) Ecoregion-based conservation planning in the Mediterranean: dealing with large-scale heterogeneity. *PLoS ONE*, **8**, e76449.

Goodenough, A.E. (2010) Are the ecological impacts of alien species misrepresented? A review of the "native good, alien bad" philosophy. *Community Ecology*, **11**, 13–21.

Grosholz, E. (2002) Ecological and evolutionary consequences of coastal invasions. *Trends in Ecology and Evolution*, **17**, 22–27.

Halpern, B.S. & Fujita, R. (2013) Assumptions, challenges, and future directions in cumulative impact analysis. *Ecosphere*, **4**, 131.

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R. & Watson, R. (2008a) A global map of human impact on marine ecosystems. *Science*, **319**, 948–952.

Halpern, B.S., McLeod, K.L., Rosenberg, A.A. & Crowder, L.B. (2008b) Managing for cumulative impacts in ecosystem-based management through ocean zoning. *Ocean & Coastal Management*, **51**, 203–211.

Halpern, B.S., Kappel, C.V., Selkoe, K.A., Micheli, F., Ebert, C.M., Kontgis, C., Crain, C.M., Martone, R.G., Shearer, C. & Teck, S.J. (2009) Mapping cumulative human impacts to California Current marine ecosystems. *Conservation Letters*, **2**, 138–148.

Hobbs, R.J., Higgs, E. & Harris, J.A. (2009) Novel ecosystems: implications for conservation and restoration. *Trends in Ecology & Evolution*, **24**, 599–605.

Hulme, P.E. (2009) Trade, transport and trouble: managing invasive species pathways in an era of globalization. *Journal of Applied Ecology*, **46**, 10–18.

Katsanevakis, S., Zenetos, A., Belchior, C. & Cardoso, A.C. (2013a) Invading European Seas: assessing pathways of introduction of marine aliens. *Ocean and Coastal Management*, **76**, 64–74.

Katsanevakis, S., Gatto, F., Zenetos, A. & Cardoso, A.C. (2013b) How many marine aliens in Europe? *Management of Biological Invasions*, **4**, 37–42.

Katsanevakis, S., Wallentinus, I., Zenetos, A., Leppäkoski, E., Çinar, M.E., Oztürk, B., Grabowski, M., Golani, D. & Cardoso, A.C. (2014a) Impacts of marine invasive alien species on ecosystem services and biodiversity: a pan-European review. *Aquatic Invasions*, **9**, 391–423.

Katsanevakis, S., Coll, M., Piroddi, C., Steenbeek, J., Ben Rais Lasram, F., Zenetos, A. & Cardoso, A.C. (2014b) Invading the Mediterranean Sea: biodiversity patterns shaped by human activities. *Frontiers in Marine Science*, **1**, 32.

Katsanevakis, S., Deriu, I., D'Amico, F. *et al.* (2015) European Alien Species Information Network (EASIN): supporting European policies and scientific research. *Management of Biological Invasions*, **6**, 147–157.

Kellon, D. & Arvai, J. (2011) Five propositions for improving decision making about the environment in developing communities: Insights from the decision sciences. *Journal of Environmental Management*, **92**, 363–371.

Korpinen, S., Meidinger, M. & Laamanen, M. (2013) Cumulative impacts on seabed habitats: an indicator for assessments of good environmental status. *Marine Pollution Bulletin*, **74**, 311–319.

Kumschick, S., Bacher, S., Evans, T., Marková, Z., Pergl, J., Pyšek, P., Vaes-Petignat, S., van der Veer, G., Vilà, M. & Nentwig, W. (2015) Comparing impacts of alien plants and animals in Europe using a standard scoring system. *Journal of Applied Ecology*, **52**, 552–561.

Ludwig, D.R., Hilborn, R. & Walters, C.J. (1993) Uncertainty, resources exploitation and conservation: lessons from history. *Science*, **260**, 17–36.

Micheli, F., Halpern, B.S., Walbridge, S., Ciriaco, S., Ferretti, F., Fraschetti, S., Lewison, R., Nykjaer, L. & Rosenberg, A.A. (2013) Cumulative human impacts on Mediterranean and Black Sea marine ecosystems: assessing current pressures and opportunities. *PLoS ONE*, **8**, e79889.

Molnar, J.L., Gamboa, R.L., Revenga, C. & Spalding, M.D. (2008) Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, **6**, 458–492.

Nentwig, W., Kühnel, E. & Bacher, S. (2010) A generic impact-scoring system applied to alien mammals in Europe. *Conservation Biology*, **24**, 302–311.

Nunes, A.L., Katsanevakis, S., Zenetos, A. & Cardoso, A.C. (2014) Gateways to alien invasions in the European Seas. *Aquatic Invasions*, **9**, 133–144.

Ojaveer, H., Galil, B.S., Campbell, M.L., Carlton, J.T., Canning-Clode, J., Cook, E.J., Davidson, A.D., Hewitt, C.L., Jelmert, A., Marchini, A., McKenzie, C.H., Minchin, D., Occhipinti-Ambrogi, A., Olenin, S. & Ruiz, G. (2015) Classification of non-indigenous species based on their impacts: considerations for application in marine management. *PLoS Biology*, **13**, e1002130.

Olenin, S., Minchin, D. & Daunys, D. (2007) Assessment of biopollution in aquatic ecosystems. *Marine Pollution Bulletin*, **55**, 379–394.

Palialexis, A., Tornero, V., Barbone, E., Gonzalez, D., Hanke, G., Cardoso, A.C., Hoepffner, N., Katsanevakis, S., Somma, F. & Zampoukas, N. (2014) In-Depth Assessment of the EU Member States' Submissions for the Marine Strategy Framework Directive under articles 8, 9 and 10. JRC Scientific and Policy Reports (EUR collection), Publications Office of the European Union, EUR 25009 EN – Joint Research Centre.

Parma, A.M. (1998) What can adaptive management do for our fish, forests, food, and biodiversity? *Integrative Biology*, **1**, 16–26.

Smith, L.D. (2009) The role of phenotypic plasticity in marine biological invasions. *Biological invasions in marine ecosystems: ecological, management, and geographic perpectives* (ed. by G. Rilov and J.A. Crooks), pp. 177–202. Springer, Berlin, Heidelberg.

Thomsen, M.S., Byers, J.E., Schiel, D.R., Bruno, J.F., Olden, J.D., Wernberg, T. & Silliman, B.R. (2014) Impacts of marine invaders on biodiversity depend on trophic position and functional similarity. *Marine Ecology Progress Series*, **495**, 39–47.

Trombetti, M., Katsanevakis, S., Deriu, I. & Cardoso, A.C. (2013) EASIN-Lit: a geo-database of published alien species records. *Management of Biological Invasions*, **4**, 261–264.

Wallentinus, I. & Nyberg, C.D. (2007) Introduced marine organisms as habitat modifiers. *Marine Pollution Bulletin*, **55**, 323–332.

Yemshanov, D., Koch, F.H., Ducey, M. & Koehler, K. (2013) Mapping ecological risks with a portfolio-based technique: incorporating uncertainty and decision-making preferences. *Diversity and Distributions*, **19**, 567–579.

Zaiko, A., Lehtiniemi, M., Narščius, A. & Olenin, S. (2011) Assessment of bioinvasion impacts on a regional scale: a comparative approach. *Biological Invasions*, **13**, 1739–1765.

Zenetos, A., Gofas, S., Morri, C. *et al.* (2012) Alien species in the Mediterranean Sea by 2012. A contribution to the application of European Union's Marine Strategy Framework Directive (MSFD). Part 2. Introduction trends and pathways. *Mediterranean Marine Science*, **13**, 328–352.

SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

Table S1 Matrix of impact weights $w_{i,j}$ for all combinations of the 60 invasive alien species and the 13 marine habitats.

Table S2 Ranking of Mediterranean countries in terms of share of the total impact estimated across the basin.

Figure S1 The 13 habitat layers used for the estimation and mapping of the cumulative impact index of invasive alien species on the marine ecosystems of the Mediterranean Sea.

Figure S2 Histogram of cumulative impact (CIMPAL) scores depicting the number of 10 × 10 km cells that fall within each impact category (based on the uncertainty-averse strategy).

Figure S3 Confidence index of the estimates of the estimated cumulative impact (CIMPAL) score of alien species on the marine ecosystems of the Mediterranean Sea.

Figure S4 Mediterranean Sea map of the CIMPAL score of 60 invasive alien species to 13 marine habitats, as in Fig. 2 of the article, but by using a linear scale for the impact weights.

Figure S5 Mediterranean Sea map of the CIMPAL score of 60 invasive alien species to 13 marine habitats, as in Fig. 2 of the article, but by using a logarithmic scale for the impact weights.

Figure S6 Comparison of two versions of the CIMPAL index calculated with different decision-making strategies: uncertainty averse versus precautionary.

Figure S7 Mapping the spatial pattern of differences in CIMPAL magnitude between the two decision making strategies.

Appendix S1 Treatment of uncertainty.

Appendix S2 Comparison of the ranking of sites according to the CIMPAL index based on the two decision-making strategies: the uncertainty-averse and the precautionary approach.

BIOSKETCHES

Stelios Katsanevakis is an associate professor of marine ecology in the Department of Marine Sciences at the University of the Aegean. Previously, he worked in the Joint Research Centre for the development of the European Alien Species Information Network. His primary research interests include marine conservation, biological invasions and ecological monitoring.

Fernando Tempera is a Post-Doc researcher at the Joint Research Centre of the European Commission, focusing on modelling and mapping of marine ecosystem services. His interests extend to marine biotope mapping, marine biodiversity cataloguing, and ocean conservation. His research has often involved using geographic information systems and statistical modelling.

Heliana Teixeira is a Post-Doc researcher at the Joint Research Centre of the European Commission, focusing on the assessment and conservation of marine biodiversity. She is a benthic ecologist, with experience in the development of ecological indicators and tools for environmental assessments, from estuarine to coastal and marine environments.

Editor: Michael Bode