**A framework to quantify the vulnerability of insular biota to global changes**

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Running head : Vulnerability of islands

Keywords (6-10) : islands, biodiversity, anthropogenic threats, risk, adaptive capacity, exposure, sensitivity

**Words : 4670**

**Figures : 2**

**Table : 1**

**Box : 2**

**Abstract:**

**The majority of vulnerability assessments of biodiversity to global changes have so far being designed for mainland systems, overlooking islands. However, islands harbour unique biodiversity and are epicentres of ongoing extinctions.** **We thus introduce a new framework for quantifying the vulnerability of terrestrial insular biota to multiple threats. This framework uses markers of exposure, sensitivity, and adaptive capacity that reflect the unique characteristics of island biodiversity, from population to island levels. Our framework involves five steps: (1) identifying the scope of the vulnerability assessment, (2) selecting the most appropriate markers, (3) computing the vulnerability metric, (4) evaluating uncertainties, and (5) providing recommendations for conservation. We discuss the need and urgency to deploy this framework to guide evidence-based decisions for the conservation of insular biodiversity and for an improved attention to insular biota at the science-policy interface.**

**Introduction**

Islands harbour the most vulnerable ecosystems affected by global changes (Fernández-Palacios, Kreft, et al., 2021). Approximately 75% of extinctions have occurred on islands, and over half of all terrestrial species that face imminent extinction are island-dwellers, with invasive alien species and land-use change as leading drivers of species’ declines. Climate change emerges as a growing threat for island biota and it may interact in unexpected ways with invasive alien species and land-use change (Capdevila et al., 2021; Mantyka-Pringle et al., 2011). As a result, islands are commonly considered the epicentres of past, imminent, and potential future species extinctions (Supplementary Material S1).

At least three reasons can explain the disproportionate vulnerability of island ecosystems. Most insular species are intrinsically more sensitive to any given threat due to their specific traits deriving from the so-called ‘island syndrome’(Benítez-López et al., 2021; Biddick et al., 2019; Lomolino, 1985; Rozzi et al., 2023). Second, insular species are less likely to adapt to new threats due to their inherent demographic features (e.g., small population sizes, naturally fragmented distribution ranges). Lastly, the physiography of islands, specifically in the case of isolated, small-sized ones, renders their biota more exposed to threats and also less able to escape compared to their mainland counterparts (Fernández-Palacios et al., 2021; see also the “uniqueness of insular biota” section for a more complete description of those inherent vulnerabilities).

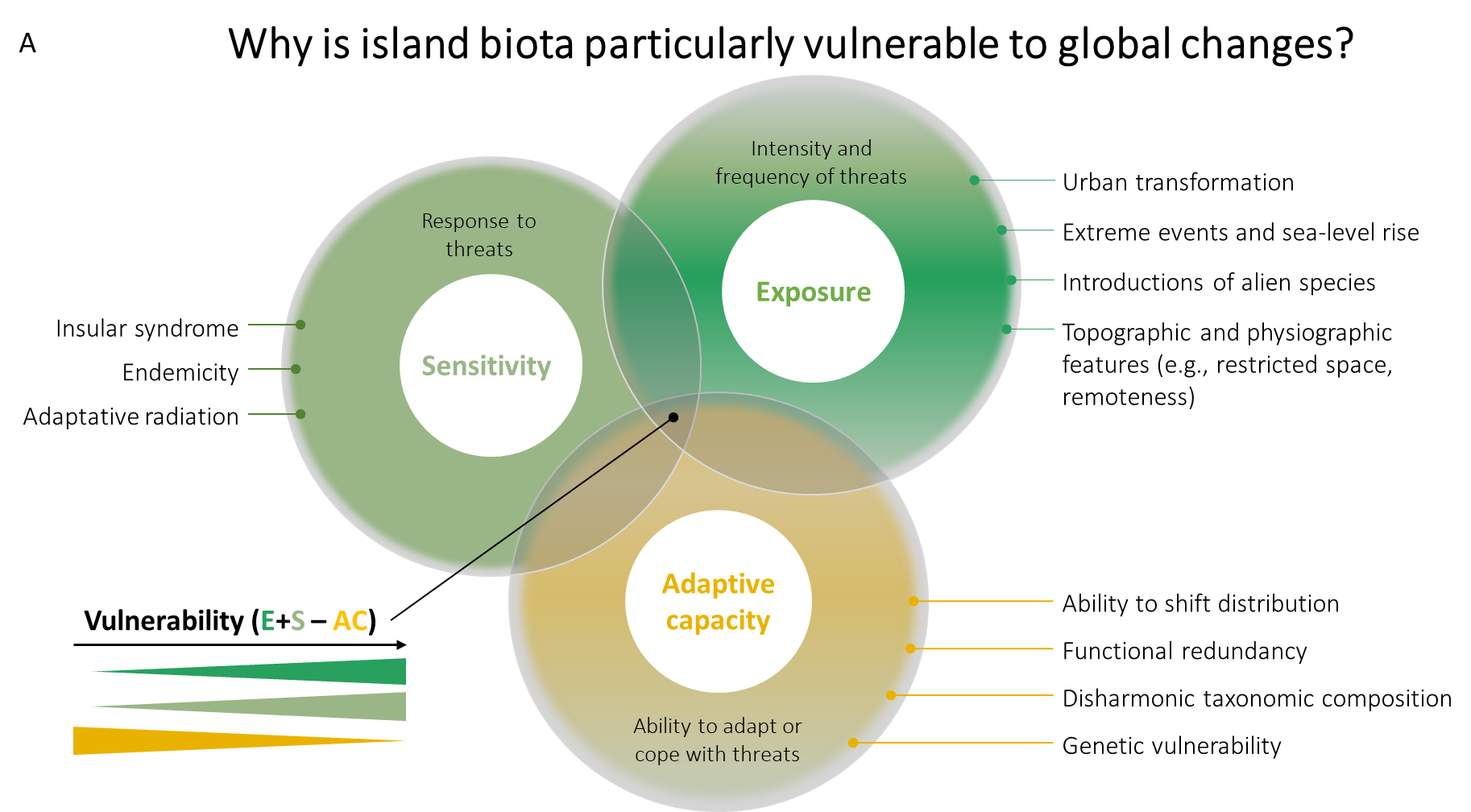
Despite the urgency needed to protect the unique and highly vulnerable island biodiversity from ongoing global changes, insular biota is only briefly mentioned in international biodiversity policy frameworks. In December 2022, the 196 parties of the Convention on Biological Diversity (CBD) adopted the [Kunming-Montreal Global Biodiversity Framework](https://www.cbd.int/doc/decisions/cop-15/cop-15-dec-04-en.pdf) (KM-GBF), involving 23 action-oriented global targets. Among them, only one target (Target 6) explicitly mentions islands as priority areas for conservation (though not explicitly mentioning insular biota) and only two other targets (Target 3 and 4, GBF) require strong actions on islands. This lack of attention reflects the scientific bias towards mainland or charismatic species (Albert et al., 2018; Rodrigues et al., 2010), with remote islands receiving less consideration in biodiversity assessments, hindering the development of effective conservation plans on insular biota (Troudet et al., 2017).

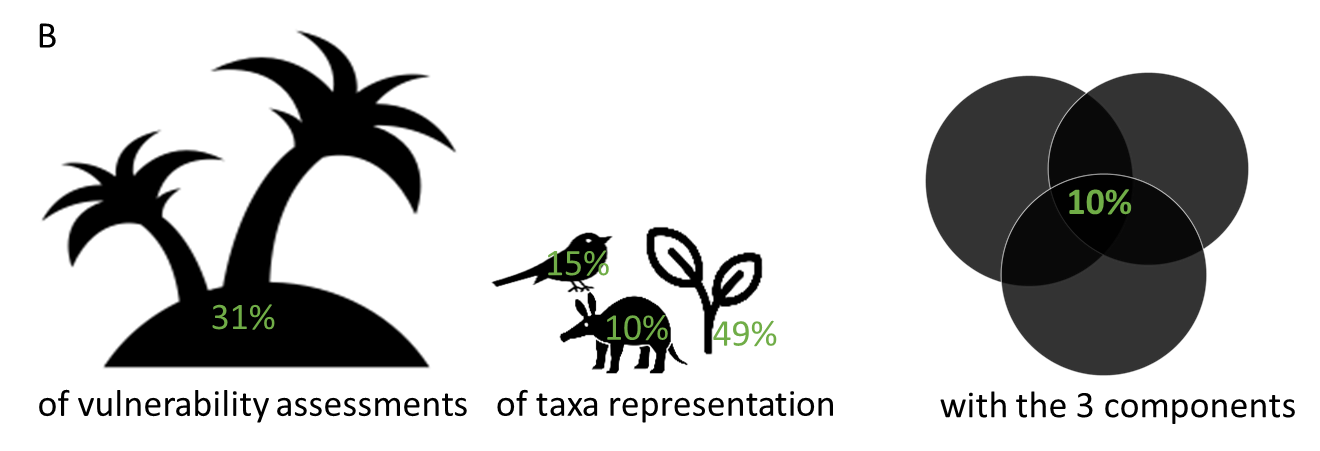
Here, we introduce a new framework for quantifying the vulnerability of terrestrial insular biota to multiple threats, which is specifically designed to reflect the challenges associated with the uniqueness of insular biota, enhances their vulnerability to global changes (e.g., island syndrome, isolated nature of islands, high endemism; see also Figure 1A). We define vulnerability across multiple biodiversity dimensions, considering the exposure, sensitivity, and adaptive capacity of insular biota to multiple threats. The originality of our framework is that it is specifically designed for insular biodiversity, with the inclusion of multiple threats, taxa, and dimensions of diversity, such as functional and phylogenetic diversity, as well as the inclusion of vulnerability markers at species, community and assemblages levels of islands. We first describe the characteristics that contribute to the vulnerability of island biota to global changes, which need to be considered in insular biodiversity vulnerability assessments. Then, we describe how our framework can address some of the ongoing questions on island biota vulnerability.

**Challenges**

*The uniqueness of insular biota enhances their vulnerability to threatening processes*

The inherent uniqueness of insular biota (e.g., endemic species, distinct traits, unique lineages) increases their overall vulnerability to global change drivers (Fernández-Palacios, Kreft, et al., 2021). Islands are isolated, have complex topographies, are prone to extreme events (e.g. volcanic eruption, landslides, hurricanes), and harbour fewer potential refuge (compared to mainland), making them more exposed to threats (Russell & Kueffer, 2019) (Figure 1A). Islands also host a large number of endemic species: up to 90% of endemic non-vagile taxa (i.e., non-flying vertebrates, seed plants, molluscs or arthropods) occur in islands (e.g., Madagascar, Antonelli et al., 2022). Endemics have small population sizes and restricted geographical ranges (sometimes limited to an archipelago, an island or even a single peak, volcano or cliff) (Fernández-Palacios, Otto, et al., 2021), increasing their extinction probability (Manes et al., 2021). An iconic example is the radiation of Hawaiian Honeycreepers, a group of small birds particularly vulnerable to invasive alien species and habitat modification **(Box 1**).





***Figure 1:******A*** *Conceptual figure of the vulnerability of insular biota to global changes with a non-exhaustive list of the characteristics contributing to each vulnerability component (i.e., exposure, sensitivity, and adaptive capacity). For example, insular syndrome traits, endemicity, and adaptive radiation are characteristics that increase the sensitivity of insular biota to global changes compared to mainland biota, whereas (i) a lack of functional redundancy, (ii) disharmonic taxonomic composition, and (iii) genetic vulnerability are characteristics that decrease adaptive capacity.* ***B*** *Percentage of island biota representation, taxonomic representation, and vulnerability components representation in current climate change vulnerability assessments. The numbers are calculated from de los Ríos et al. (2018).*

Insular species have also evolved distinct, and unique ecological, physiological, behavioural, morphological, and life-history traits, a phenomenon commonly referred to as the ‘island syndrome’ (Baeckens & Van Damme, 2020). These distinct and unique traits increase their sensitivity to current and future anthropogenic threats, including overharvesting, habitat loss, increased drought and invasive alien species (Rozzi et al., 2023; Zizka et al., 2022) (Figure 1A). This phenomenon is for example associated with dioecy in plants, which makes them more likely to disappear if their mutualists’ species go extinct, presumably increasing their vulnerability to human-induced perturbations (Schrader et al., 2021). Reduced herbivory and predation pressure on islands leads to low levels of spinescence compared to their mainland counterparts, making island species more susceptible to the introduction of exotic herbivores (Burns, 2016; Clavero et al., 2009), unless they have coevolved with megafaunal herbivores (Barton et al., 2024). The loss of flight capacities renders insular birds unable to escape from anthropogenic threats (Sayol et al., 2021), making them more susceptible to extreme weather events, such as hurricanes, or gradual environmental change (e.g., progressive directional changes such as climate change) (Burns, 2019; Roff, 1990).

Additionally, the intricate and often rugged topography of oceanic islands frequently leads to long-term isolations of populations, which may result in genetic drift and then inbreeding when population sizes are small (Frankham, 2002). Furthermore, island colonization within archipelagos and isolation by distance could also lead to high population structure between islands (Parmakelis et al., 2015; White & Searle, 2007). These populations are thus more prone to genetic diversity loss, which may result in fewer opportunities to adapt to changes. Although this does not necessarily increase their vulnerability *per se*, the loss of genetic diversity due to natural or anthropogenic disturbances, especially when exposed to disturbances over a short time period (Inamine et al., 2022), can cause a demographic or genetic collapse. Thus, losing an insular population can cause demographic and genetic loss, whereas it causes only demographic loss on the mainland. Because of that, the effectiveconservation units on islands should be, in most cases, at the population level, rather than species level (e.g., (Melo & O’Ryan, 2007)).

Besides population- and species-level characteristics, the composition of insular species’ assemblages harbours features that make them more vulnerable. One example is their disharmonic taxonomic composition, which refers to the systematic over- or under-representation of certain taxonomic groups compared to mainland assemblages. Notable examples include the absence of several families of non-volant mammals on many isolated islands (Brace et al., 2015), and the over-representation of pteridophytes in island (Kreft et al., 2010). The absence of certain functional groups on islands, although partially compensated by in-situ diversification and differentiation, allows new exotic species filling out vacant niches (Vitousek et al., 1997). These exotic species can exert a new role (e.g., predation, competition) unknown to the native community, which is evolutionarily naive and functionally unequipped to withstand these novel pressures. Additionally, species traits in island communities are often complementary rather than redundant, leading to communities with low functional redundancy (Harter et al., 2015; Whittaker et al., 2014). This makes them more sensitive to threats (McConkey & Drake, 2015) due to the lack of 'ecological insurance' to replace the missing functions (Loreau et al., 2021).

*State of the art of vulnerability assessments on insular biota*

Despite the inherent vulnerability of insular biota to global changes, most biodiversity vulnerability assessments have designed and focused on mainland systems.Climate change vulnerability assessments emerged in the 1990s to anticipate impacts and prepare appropriate responses (Foden et al., 2019; Solomon et al., 2007). Most definitions concur that a species’ vulnerability to a threat is determined by three components: **exposure**, **sensitivity**, and **adaptive capacity** (Butt et al., 2022; Foden et al., 2013, 2019). Using this definition, a recent review showed that among the 741 studies assessing climate change vulnerability, the majority focused on mainland systems, with less than one third (n = 231) including islands (de los Ríos et al., 2018) (see Table S2 for examples) and only 136 studies associated with a specific insular country. Although this would be representative of the small land surface area occupied by islands (6.7%), it falls short in terms of biodiversity representativeness, as island’s biodiversity represents 20% of biodiversity worldwide, with 50% of threatened and 75% of known extinctions (Fernández-Palacios, Otto, et al., 2021). These island vulnerability assessments are often geographically and taxonomically restricted towards high income countries (e.g., Australia, the UK, and the USA account for 60% of studies on insular biota) and plants (49% of studies) (Figure 1B), respectively. In addition, the large majority of vulnerability assessments do not consider the influence of multiple threats (but see Santos et al., 2021; Sousa et al., 2021; Ureta et al., 2022), which are likely to act together, interact with and be exacerbated by climate change (e.g., habitat loss, overexploitation, biological invasions), potentially leading to synergistic impacts (Leclerc et al., 2018). Note that these geographic, taxonomic, or conceptual biases occur in both mainland and insular assessments (see also (de los Ríos et al., 2018).

Assuming that exposure is a suitable proxy for the impact of a particular threat, most vulnerability assessments focused on exposure alone, with less than 10% considering the three components of vulnerability (Butt et al., 2016; de los Ríos et al., 2018). This assumes that all species will have equal responses to a threat, which is highly unlikely. In fact, the likelihood of species being impacted by certain threats, either on the mainland or on islands, is modulated by their traits (Fromm & Meiri, 2021; Leclerc, Villéger, et al., 2020; Marino et al., 2022; Soares et al., 2022). We argue that trait-based vulnerability assessments, when applied to a range of taxa and threats, can provide a useful approach for (i) developing a more comprehensive index of vulnerability to threats, and (ii) informing effective management actions for conservation (Gallagher et al., 2021).

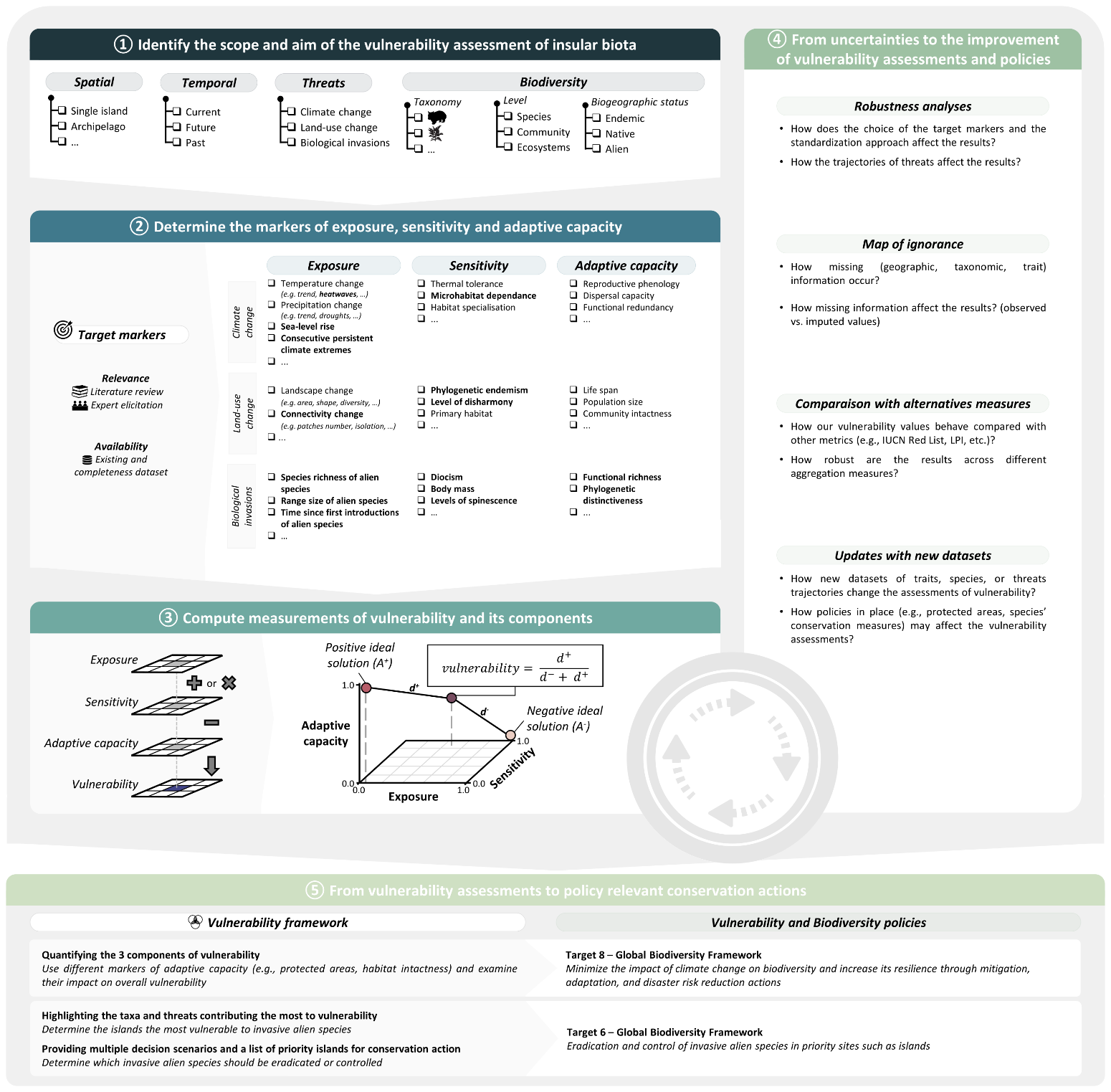
**Perspective**

We present a conceptual framework divided in five steps that provides a roadmap for vulnerability assessments of biota in insular systems, considering the three components of vulnerability (i.e., exposure, sensitivity, and adaptive capacity; Figure 2). Here, we refer to islands as insular systems that have a landmass smaller than Greenland (i.e., < 2.17 million km²) and are surrounded by sea water. We built upon previous work focused on taxonomic diversity in mainland ecosystems (Foden et al., 2013; Parravicini et al., 2014) to include in our framework various specific markers for each vulnerability component that are specifically tailored to the inherent characteristics of island biota, as well as multiple threats, taxa, and dimensions of diversity (i.e., taxonomic, functional and phylogenetic diversity). The development of this vulnerability framework tailored for island systems is part of a larger initiative to meet international policy targets that better integrate biodiversity threats and dimensions (Box 2).

*Step 1: Identify the scope and aim of the vulnerability of insular biota*

The first step is to define the scope of the vulnerability assessment in terms of spatial and temporal extent, relevant threats, and studied biota (taxonomic group, biological level, species biogeographical origin). This challenging task is pivotal for the assessment design, which in turn affects the final step of informing conservation actions (Step 5). For example, broad-scale assessments (e.g., at the global extent or among several archipelagos) contribute to strategic planning and to establish a common baseline of vulnerability information (e.g., IPBES assessments), while local-scale assessments (e.g., within archipelagos or group of islands) are appropriate for informing site-level management decisions.

For instance, a vulnerability assessment could be conducted at the spatial extent of a national park, within an island of a few hectares only, with a restricted set of species (e.g., Harper et al., (2022), 24 ha in South Africa, 18 amphibian and 41 reptile species). This can inform management priorities at the landscape scale, such as defining park- use zones to help allocate restricted areas acting as corridors for species migration, or creating habitat conditions for breeding (Harper et al., 2022). At this level, an explicit treatment of population genetics and/or population viability analyses could also be conducted, this may become more feasible in the future with the emergence of macrogenetics studies (Leigh et al., 2021). In parallel, studies focusing on the global extent are key to assess vulnerability metrics, identify geographical shortfalls in data coverage, and support the implementation of conservation policies to mitigate biodiversity losses. Note that in all cases, it is important to control by island area or species richness, to avoid biases towards larger islands when calculating vulnerability metrics. Finally, even if vulnerability assessment are conducted at the global scale, special attention should be paid to endemic species or insular populations, since island biodiversity conservation operates at the population level.

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***Figure 2:*** *Conceptual framework for assessing the vulnerability of island biodiversity to global changes. The framework consists of five steps and can be iteratively applied (see circular arrow). Markers in bold font represent the markers specifically designed for insular biota.*

*Step 2: Determine the markers of exposure, sensitivity, and adaptive capacity*

There are many different relevant markers associated with species’ vulnerability to threats. Those markers may vary in type (e.g., ecological or demographic) and organisation level (e.g., population, species, community, or ecosystem). One can review the scientific literature and elicit expert opinion toidentify and collect relevant markers for each component of vulnerability based on the scope and purpose of the assessment. For instance, assessing the vulnerability of terrestrial species to land-use change and climate change requires markers of exposure to climate change (e.g., change in precipitation regimes, sea-level rise) and land-use change (e.g., urbanization rates, forest conversion into agriculture, infrastructure development, shift from agriculture to tourism) (e.g., Bellard, Leclerc, & Courchamp, (2015); Bellard, Leclerc, Hoffmann, et al., (2015). Moreover, it is necessary to list the biological traits related to the sensitivity and adaptive capacity of the studied insular species to the above-listed threats (Figure 2 and Table S1 for examples of exposure, sensitivity, and adaptive capacity markers). Markers of exposure can be linked to the focal threat(s) or general global changes, and multiple markers should be used to capture the multiple dimensions of changes (e.g., temperature change, heat waves, droughts).

sensitivity to different threats (e.g., body size, dependence on interspecific interactions). Note that for both sensitivity and adaptive capacity, a long list of markers is applicable at either the population or species level (see (Thurman et al., 2020) for a list and Table S1). However, markers applicable at the community level, such as functional or phylogenetic diversity, could also be included (Table 1 for example). Using extinctions scenarios for threatened species, we can also establish which islands are more likely to lose a significant share of their functional and phylogenetic diversity (richness, redundancy, originality, uniqueness) (Bellard et al., 2021; Llorente-Culebras et al., 2024). Those analyses could help identify islands that are more likely to be sensitive at the community level.

Adaptive capacity refers to the ability of insular biota to respond to stressors, caused by multiple threats or novel conditions, by either persisting *in situ* or by shifting in space or time (following (Thurman et al., 2020)). Acclimation, behavioural change, phenotypic plasticity and evolutionary adaptation may all contribute to adaptive capacity (Foden et al., 2018; Royer-Tardif et al., 2021). For mobile animals, especially the most vagile organisms, markers of adaptive capacity can incorporate traits linked to movement or mobility (e.g., migration frequency and distance, site fidelity) (Butt et al., 2022; Thurman et al., 2020). Note that for plants and sessile organisms in general, adaptive capacity mostly refers to their capacity to persist *in situ* with mating system and fecundity as strong markers (Thurman et al., 2020). Finally, adaptive capacity of insular biota can also be modulated by extrinsic factors, such as habitat quality, habitat availability, habitat connectivity and level of protection, factors which could therefore be included in the measure of adaptive capacity at the system level (i.e., island, archipelago).

***Table 1:*** *Proposed functional-based and phylogeny-based metrics that could be used as markers at the community level in vulnerability assessments.*

|  |  |  |
| --- | --- | --- |
|  | **Functional-based markers** | **Phylogeny-based markers** |
| **Sensitivity** | ***Functional rarity*** | ***Phylogenetic endemism*** |
| *Definition* - Functional rarity, expressed at the species level, is the combination of the functional distinctiveness/uniqueness (based on traits), the scarcity and the geographic restrictedness of a species (Violle et al., 2017). | *Definition* - Phylogenetic endemism is the spatial restriction of the phylogenetic diversity of a community (Rosauer et al., 2009). This is a relative measure of endemism that represents the degree to which lineages or branches of the tree of life are spatially restricted. |
| *Rationale* - Given that functionally rare species can play a critical role in ecosystem functioning, and because rarity is linked to species’ sensitivity (Davies et al., 2004; Loiseau et al., 2020), areas with a high proportion of functionally rare species are particularly susceptible to threats. | *Rationale* - Areas with high phylogenetic endemism are characterized by the presence of species that have diversified and evolved in response to specific environmental conditions within a particular location. Among the drivers shaping phylogenetic endemism, climate plays a significant role (Guo et al., 2023), implying that changes in climate conditions may disproportionately affect these species, potentially leading to their decline or extinction. |
| **Adaptive capacity** | ***Functional redundancy*** | ***Phylogenetic distinctiveness*** |
| *Definition* - Functional redundancy of a given community reflects the tendency for the constituent species to perform similar functions (Mouillot et al., 2014). | *Definition* - Phylogenetic distinctiveness reflects the degree of isolation of a species or a group of species within a phylogenetic tree (Pavoine & Ricotta, 2021). |
| *Rationale* - Functional redundancy has been theoretically and empirically linked to the concepts of resistance and resilience of ecosystem functioning to species loss (Biggs et al., 2020) via the insurance hypothesis (McCann, 2000). Hence, more functionally redundant systems should show greater resilience to perturbations (Mouillot et al., 2014) become role of extinction species can be fulfilled by functionally close species. This indirectly reflects the ability of the system to adapt to disturbances. | *Rationale* - Evolutionarily distinct species or group of species represent uniquely divergent genomes (Warren et al., 2008). Consequently, sets of evolutionarily distinct species are expected to encompass a large proportion of the parental clade’s total phylogenetic diversity, which may play a crucial role in ensuring long-term stability and resilience (Cadotte et al., 2012). This implies that communities exhibiting higher phylogenetic distinctiveness are more likely to harbor increased evolutionary potential, enabling them to adapt. |

*Step 3: Compute measures of vulnerability and its components*

Once all the markers of the three components of vulnerability are collected, the next step is to combine them all into a composite vulnerability measure. However, aggregating multiple markers into a single value for each of the three vulnerability components is challenging. A recent review showed that most assessments use arithmetic mean for the aggregation (Tonmoy et al., 2014). In the case of quantitative markers, as here, one possibility is to normalise each marker to a 0-1 range. This transformation, which can be done with multiple methods (e.g. Leclerc et al.2020), creates unitless metrics with equal weight. Then the markers of each component (i.e., exposure, sensitivity, and adaptive capacity) can be summed and re-scaled to obtain normalized values of exposure, sensitivity, and adaptive capacity. This technique has the advantage of clearly identifying which components drive vulnerability, by ensuring that all markers are weighted equally, and can thus effectively guide the implementation of conservation actions at the island level. The different markers could also be weighted differently to put more emphasis on specific markers depending on the current level of island protection policy or biodiversity richness occurring on each island. Finally, when all the data has been aggregated, most of the current approaches are criteria-based, classifying biota into categories of vulnerability from low to high by summing up the different components of vulnerability (hereby referred to as qualitative or semi-quantitative frameworks). To avoid arbitrary thresholds, we propose a quantitative framework using multicriteria decision analysis (Leclerc , et al., 2020; Parravicini et al., 2014) that provides continuous vulnerability values and a relative ranking of islands or archipelagos. This method ranks alternatives according to their distance to positive (i.e., minimal vulnerability, with low exposure and sensitivity, and high adaptive capacity) and negative ideal solutions (see (Leclerc, Courchamp, et al., 2020) for details). Note that alternative approaches exist for combining the three vulnerability components such as additive effect between components (Nyboer et al., 2021) or interacting effect between exposure and sensitivity for instance (Silva Rocha et al., 2024). In any case, we recommend carrying out robustness analyses to explore the uncertainty potentially introduced by the aggregation and weighting methods.

*Step 4: From uncertainty assessments to the improvement of vulnerability assessments and policy recommendations*

The outcomes of vulnerability assessments can be affected by several factors, including missing data, variation of underlying traits, aggregation techniques, and, in the case of future-looking assessments, by uncertainties about the future trajectories of anthropogenic threats. Techniques to estimate uncertainty from missing trait data (e.g., Hossain et al., (2019)) as well as uncertainty in modelling the future of biodiversity (e.g., Thuiller et al., (2019)) are readily available. In addition, we recommended carrying out robustness analyses to assess the impact of including or excluding specific markers from the calculation, by rerunning the vulnerability assessment but without the focal marker. Robustness analyses could also be used to explore the effects of alternative aggregation or standardisation techniques (Tonmoy et al., (2014), see also Boyce et al., (2022)). We also recommended to validate the outcome of the vulnerability assessments with data from another temporal period (e.g., by comparing with past data), or by comparing with other islands with documented vulnerability or other metrics of vulnerability (e.g., IUCN Red List of species, Living Planet Index). In this context, it is crucial to both estimate the uncertainty and communicate it. This can be achieved by, for example, mapping the variance of the estimated vulnerability metric or using maps of ignorance (e.g., (Rocchini et al., 2011; Tessarolo et al. 2021)). These uncertainty evaluations will help to identify future research priorities, but also to build authority and increase the relevance of the vulnerability assessments for policy makers, as it is the case for the climate and biodiversity assessments of the IPCC and IPBES, respectively (Joyce et al., 2011; Vadrot, 2020). We also advocate for the integration of interactive interfaces with options to select threats, markers, taxa, or ecosystems of interest, as well as different narratives trajectories, which may complement the scientific message, which is particularly helpful for simplifying multidimensional information (McInerny et al., 2014). Finally, we recommend conducting an iteration and review step as an opportunity to adjust and update the vulnerability framework with new ecological data, such as the IUCN perform periodic Red List (re-)assessments as new information becomes available (Henry et al., 2024).

*Step 5: From vulnerability assessments to policy relevant conservation actions*

The vulnerability framework developed here could be used to increase the fundamental knowledge on biodiversity vulnerability (Step 1-4) and to guide the implementation of biodiversity conservation policies (Step 5, Box 2). For instance, one of the targets of the KM-GBF is to minimize the impact of climate change on biodiversity through (among others) mitigation, adaptation and risk disaster reduction actions. Such aims require to use different markers of adaptive capacity through multiple plausible scenarios (e.g., protected areas, habitat intactness) to inform on how different management actions can impact the overall vulnerability. Vulnerability assessments can help target the control and eradication of invasive alien species in conservation priority sites such as islands (Target 6), but also identifying the sites that need to be under protection (Target 3) to conserve biodiversity. Effective conservation practice relies on understanding biodiversity’s vulnerability through multiple lenses. Where the exposure component is common across species (e.g., inundation, invasive alien species), the threat itself may be managed through dedicated programs, like mangrove rehabilitation or invasive alien species control (Jones et al., 2016). Finally, beyond its use in current policies, our vulnerability framework could be useful in drafting new policies. Indeed, to set relevant targets in the new policies being drafted (e.g., by [year], reduce by [number] % the number of species threatened by [threat]), policy makers need to know (i) what is the current situation and (ii) what could be the future situation under different scenarios. Our vulnerability framework may provide both, and can contribute to the setting of policy targets that are simultaneously ambitious and reachable for insular biodiversity.

**Concluding remarks**

Given the proliferation of multiple threats that islands are facing, efforts must be made to study island biota in the light of global changes, and thus to embrace the whole of biodiversity, mainland and insular, in current vulnerability assessments in international arenas such as the IPCC or IPBES. Our comprehensive and detailed framework lays the foundations to understand and predict island biodiversity vulnerability to global changes. Yet, the outcomes of vulnerability assessments will be challenged by missing data and, in the case of future assessments, by uncertainties about the future trajectories of anthropogenic threats. In this context, it is crucial to both estimate and communicate uncertainty (e.g., (Hossain et al., 2019; Rocchini et al., 2011; Tessarolo et al., 2021; Thuiller et al., 2019)), which will ultimately help to protect biodiversity with more robust information.

**Glossary:**

**Adaptive capacity**: the ability of a population or species or community to adapt to changing conditions; this may be via ecological (i.e., physiological and/or behavioural plasticity) or evolutionary adaptation (i.e., through natural selection acting on traits).

**Adaptive radiation**: the rapid diversification of a single evolutionary lineage into multiple ecologically or morphologically distinct species.

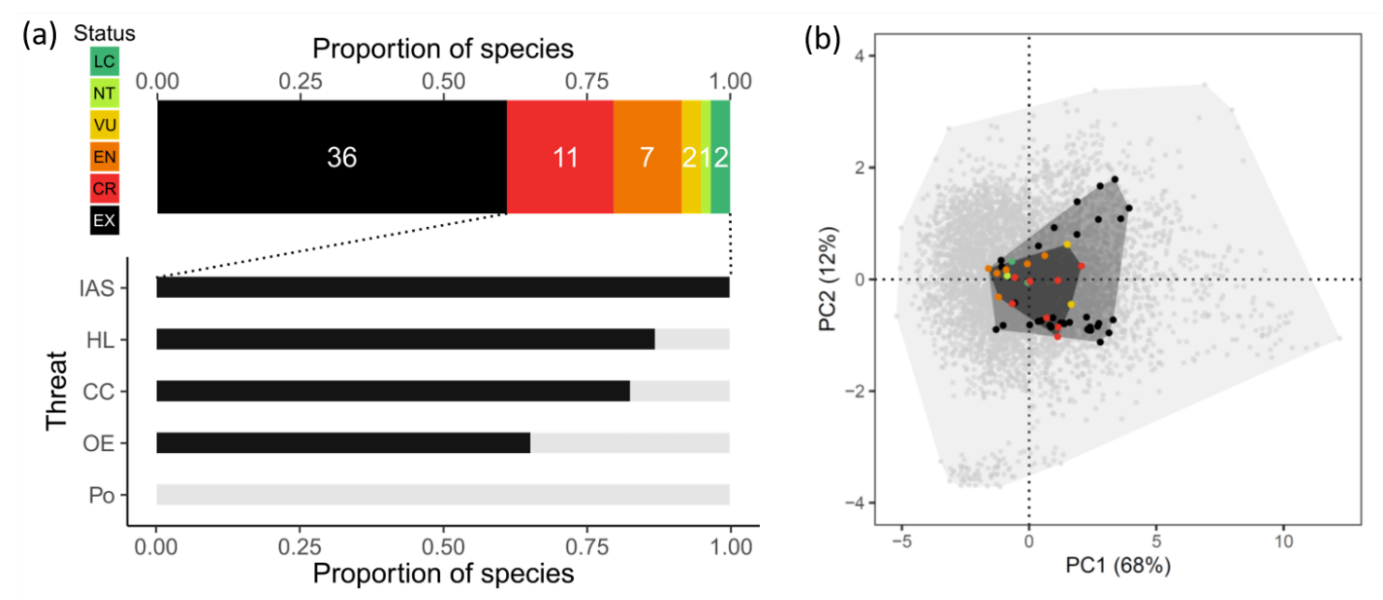
**Exposure**: the extent to which each population or species’ or a community physical environment changes due to global threats (Foden et al., 2013). It includes the intensity, magnitude and frequency of the threat(s).

**Sensitivity**: the intrinsic capacity of population or species or community to cope with threats, based on their life-history, ecology, morphology, or behaviour.

**Threat**: external factor that has the potential to impact the viability, abundance, distribution, or behaviour of a population, species, or community.

**Vulnerability**: susceptibility of a system/species to a negative impact following exposure to a threat

**Box 1 | The case of the Hawaiian honeycreepers**

The Hawaiian Islands constitute an archipelago of eight larger and numerous smaller volcanic islands in the North Pacific Ocean, in total about 16,644 km2 in size, and about 3,200 km from the nearest mainland. The islands in the northwest are older and typically smaller, due to progressive erosion, while the islands in the southeast are larger and still volcanically active. Thanks to the archipelago’s extreme isolation and the islands’ variation in size and environmental conditions, Hawaii has been a hotspot of speciation and adaptive radiation (Baldwin & Sanderson, 1998; Lerner et al., 2011; Price, 2004). The Hawaiian honeycreepers (Fringillidae: Drepanidinae) represent one of the most iconic examples of adaptive radiation, which has resulted in a striking variation in bill morphology (Lerner et al., 2011). This strong adaptive radiation, however, has also led to species with highly restricted ranges and naturally small population sizes. Hence, little is needed to push these species over the brink of extinction and, to date, at least 36 of the 59 known species of honeycreepers are extinct (Figure 3a). Extinction has been non-random with respect to the functional trait space of the species, with extinct species generally larger in size (Figure 3b). The main drivers of past extinction, as well as current threats to the species, are the introduction of invasive alien species and the loss of habitat due to conversion to agricultural land. Invasive species include predators of birds and eggs (rats, cats, dogs), herbivores that modify the habitat (for example, the extinction of Laysan Honeycreeper *Himatione fraithii* is ascribed to the introduction of rabbits, which eliminated virtually all vegetation cover from the Laysan islands), and vector species of infectious diseases, such as avian malaria (Benning et al., 2002). Habitat loss began with Polynesian colonists, who cleared much of the low-elevation and seasonally dry forest for agricultural purposes, and was continued by later European colonists, who additionally converted high-elevation forests for pasturage (Riper & Scott, 2017). Climatic factors contributed little to past extinctions but are a progressive threat to extant species, especially because it may lead to an upslope shift of infectious diseases (notably avian malaria) (Benning et al., 2002).

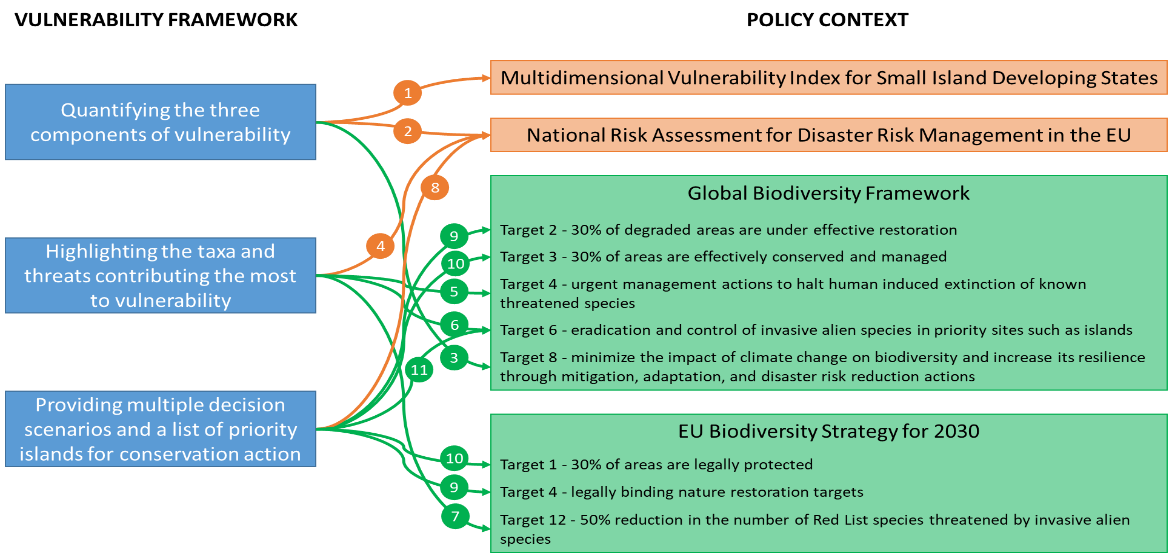
**Figure 3: Conservation status, threats, and morphospace of Hawaiian Honeycreepers.** (a) IUCN Red List threat status of 59 known species of honeycreeper (LC: least concern, NT: near threatened, VU: vulnerable, EN: endangered, CR: critically endangered, EX: extinct) and prevalence of threats among the 23 extant species (IAS: invasive alien species, HL: habitat loss, CC: climate change and extreme events, OE: over-exploitation, Po: pollution) (b) The morphospace compared between 19 extant honeycreepers (dark grey polygon), 54 extant and extinct honeycreepers (medium grey polygon), and 5,974 extant passerine birds globally (light grey polygon). The morphospaces for the extant species were obtained through a principal component analysis (PCA) based on the eight morphological traits available in AVONET (Tobias et al., 2021), namely bill length from tip to skull along the culmen, bill length to nostrils, bill width and depth to nostrils, tarsal, wing and tail length, and Kipp’s distance. The same eight traits for the extinct honeycreepers were taken from (Matthews et al., 2023). The first two principal components explain 80% of the total morphological variance and are primarily associated with body size (PC1) and tarsus length versus Kipp’s distance (PC2), respectively. The list of species was obtained from (Matthews et al., 2023; Ricklefs, 2017) and the IUCN Red List of Threatened Species v2023-1. Species names were harmonized and duplicates removed based on the taxonomic backbone used by the IUCN.

**Box 2 | A novel vulnerability framework fit for multiple policies**

This novel vulnerability framework proposed for insular biodiversity is particularly relevant to inform various vulnerability policies (in orange) and biodiversity policies (in green) at both international and supranational levels such as the European Union (EU).

First, by quantifying the three components of island vulnerability, our framework can directly feed the currently missing biodiversity component of the Multidimensional Vulnerability Index (MVI). The MVI is developed by the Small Island Developing States (SIDS; https://www.un.org/ohrlls/mvi/) with the United Nations to characterise the financing help for sustainable development needed in the face of global changes (1). The SIDS is a political coalition of 39 low-lying islands that are united by the threat posed by climate change to their survival. In climate change negotiations, they are a loud and powerful voice for upscaling climate action since they are disproportionally affected by climate change consequences. To understand and respond, several assessments of these islands’ climate change vulnerability have been carried out (e.g., (UN High Level Panel, 2022; UNFCCC, 2007)), but none includes more than a broad mention of biodiversity impacts. A scientifically rigorous assessment of climate change vulnerability of island biodiversity is, therefore, both extremely important and critically urgent. Our framework can also help EU Member States, which are required to report to the European Commission on their disaster risk management activities, to identify which components of vulnerability they can act on (2). The risk of biodiversity loss has been recently included in the Recommendations for National Risk Assessment for Disaster Risk Management in the EU (Poljansek et al., 2021), but it does not cover specifically insular biodiversity. In that context, our framework can provide a real additional value for the EU Member States with insular territories. Our framework can also further help identifying whether mitigation strategies for islands to reach Target 8 of the Kunming-Montreal Global Biodiversity Framework (hereafter, GBF) should focus on limiting territories’ exposure or promoting the adaptive capacity of insular biota (3).

Second, our framework can help EU Member States with insular territories to identify and prioritise which species, sites and threats they should use to best mitigate their vulnerability in the context of their National Risk Assessment (4). Such actions can also contribute to (i) identifying the species, sites and to focus the management measures to reach Target 4 of the GBF (5), (ii) determining the most vulnerable islands to invasive alien species to reach Target 6 of the GBF (6), and helping EU Member States with insular territories to identify species threatened by invasive alien species on which to focus conservation actions to reach Target 12 of the [EU Biodiversity Strategy for 2030](https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A52020DC0380" \o "https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A52020DC0380) (EU BDS, (7)).

****Finally, by providing multiple scenarios of threats trajectories and a list of priority islands for conservation action, our framework can contribute to document the risk of biodiversity loss under different conservation scenarios to EU Member States with insular territories (8). The vulnerability framework could contribute to identifying priority islands for restoration (target 2 of the GBF and target 4 of the EU BDS (9)) and for protection (target 3 of the GBF and target 1 of the EU BDS (10)), as well as in determining which invasive alien species should be eradicated or controlled (target 6 of the GBF (11)).

*This figure emphasizes some possible outcomes of the vulnerability framework and how they are linked (arrows) to the vulnerability (in orange) and biodiversity policies (in green). Each circled number represents a concrete example of how our vulnerability framework can support existing policies, and all those examples are detailed in the box.*

**Funding :** This publication is a product of the RIVAGE group  
funded by the synthesis center CESAB of the French Foundation for Research on Biodiversity  
(FRB; www.fondationbiodiversite.fr)**.**

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**Supplementary material :**

**Supplementary appendix 1 :**

**Islands as past, current, and future epicentres of extinctions:** The IUCN Red List documented nearly 1,000 species as globally extinct or extinct in the wild (IUCN Red list access 04/07/2023). Among the documented extinctions, the overwhelming majority occurred on oceanic or continental islands. Extinction hotspots are located in the Oceanic realm, followed by the Afrotropical realm (e.g., Madagascar) and the Nearctic. Among reported extinctions in the IUCN Red List, the large majority are represented by animal species (>80%), followed by plants. Most of these species have gone extinct because of invasive alien species, but a large number have also concurrently been harmed by wildlife exploitation and/or land use change due to cultivation (Leclerc et al., 2018; Maxwell et al., 2016). In fact, extinct species within insular regions have faced, on average, four threats within the IUCN threat classification scheme (i.e., biological invasions, wildlife exploitation, cultivation, and habitat modifications), and the exposure to threats for birds, mammals, amphibians, reptiles, freshwater fish, plants, arthropods, and gastropods continue to increase, with an average of ten threats being faced by threatened species (Leclerc et al., 2018).

Moreover, islands makes a disproportionate contribution to global biodiversity in relation to their surface area, especially regarding the percentage of threatened species (Fernández-Palacios, Kreft, et al., 2021), the risk for future extinctions is thus high. The first extinction due to climate change already occurred on an island near Papua New Guinea (Fulton, 2017). In 2016, *Melomys rubicola* (also called the Bramble Cay melomys) became the very first documented extinction due to climate change (Waller et al., 2017). This small rodent was endemic to the low-island of Bramble Cay in Australia and was periodically recorded from 1978 to 2009. Seven years after its last observation in the field and numerous efforts to trap the species in the years after, the species was officially declared extinct. Although only symbolic, compared to a thousand species already extinct since 1500, this is one of the first documented victims of climate change as a major factor of near-future extinctions in our contemporary history. Unfortunately, under future climate change, we can expect that islands will continue to contribute disproportionately to the loss of biodiversity, given that the overwhelming majority of critically endangered species are endemic to island systems, and their inherent characteristics make them extremely vulnerable to global change drivers

**Table S1:** Traits and characteristics from population to the community that may be included in the island biota vulnerability framework (IBVF) with examples of sources where to calculate or when directly available.

|  |  |  |  |
| --- | --- | --- | --- |
|  | Traits | Level | Availability |
| Population dynamic | Growth rate | population | Living planet database (vertebrates), FishGlob (fishes), German vegetation (plants) |
| Population size | population | CoralTraits (corals), BIOTime (multiple taxa) |
| Reproduction | Reproductive strategies | species | GIFT (plants), COMPADRE, GARD (reptiles) |
| Mating system | species |  |
| Clutch size | species | GARD (reptiles), AmphiBio (amphibians), COMBINE (mammals), AVONET (birds) |
| Fecundity | species | GARD (reptiles), AmphiBio (amphibians) |
| Temperature-dependent sex determination | species |  |
| Generation time | species | Amniote (vertebrates), CoralTraits (corals), COMPADRE (plants) |
| Parental care (time) | species | Amniote (vertebrates) |
| Life-history | Life-cycle | species | TRY (plants), GIFT (plants) |
| Life stage | population | FishBase (fishes), CoralTraits (corals), Amphibio (amphibians) |
| Dependence on intraspecific interactions | species | CoralTraits (corals) |
| Body dimension (mass/size?) | species | Amniote (vertebrates), TRY (Plants), GIFT (plants), COMBINE (mammals), AVONET (birds). |
| Clonality | species | GIFT (plants), COMPADRE (plants) |
| Acclimatation or evolution | Phenotypic plasticity | population | Noble et al 2018 (reptiles) |
| Genetic diversity | population | Miraldo et al. 2016 (mammals) |
| Life span | species | AmphiBio (amphibians), |
| Ecological characteristics | Climatic niche breadth | species | GARD (reptiles), GBIF (plants), BIEN (plants) |
| Endemicity | species | Global Species Database (vertebrates, invertebrates, plants) |
| Habitat specialization | species | IUCN |
| Habitat condition | species | IUCN |
| Diet breadth | species | GARD (reptiles),  AmphiBio (amphibians), |
| Nocturnality | species | GARD (reptiles), Amphibio (amphibians) |
| Dependence on water for reproduction, foraging, shelter | species | Amphibio (amphibians) |
| Competitive ability | species |  |
| Movement and mobility | Dispersal capacity | species | AVONET (birds), TRY (plants), GIFT (plants) |
| Site fidelity | species |  |
| Migration frequency and distance | species | AVONET (birds) |
| Flight efficiency | species | AVONET (birds) |
| Number of insular populations | population |  |
| Home range | population | HomeRange (mammals) |
| Distribution | Geographic rarity | population |  |
| Low local abundance | island level |  |
| Range size | population | IUCN |
| Community properties | Level of disharmony | community |  |
| Functional redundancy | community | TRY and GIFT(plants) |
| Phylogenetic diversity | community | GIFT, WCVP (plants) |
| Richness | community | GIFT, WCVP (plants) |
| Intactness | ecosystem |  |
| Phylogenetic endemism | community | Vertlife (vertebrates), PhylomeDB (plants), GIFT, WCVP (plants) |
| Inherent vulnerability | Behaviour (aggressiveness) | species |  |
| Dwarfism-gigantism | species | Meiri et al 2008 (mammals), Meiri 2007 (reptiles), Benítez-López et al. 2021, Rozzi et al. 2023 GIFT (plants) |
| Vulnerability to diseases | species | GABiP (amphibians) |
| Tolerance to drought |  | try-db.org (plants), Le Galliard et al. 2021 (reptiles) |
| Tolerance to fire | species | try-db.org (plants) |
| Endemism | community | GIFT and WCVP (plants) |

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**Table S2:** An overview of studies assessing the vulnerability of species to climate change. This is a non-exhaustive summary that focuses on studies that have been conducted recently, use trait-based approaches, and include a full assessment of vulnerability with exposure, sensitivity, and adaptive capacity markers.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| **Study** | **Ecosystems** | **Spatial coverage** | **Taxa** | **n** | **Main findings** |
| (Barry et al., 2023) | Freshwater | Ireland | Fish | 32 | All species were vulnerable to some effect of climate change with cold water species more vulnerable to climate change than warm water species. |
| (Boyce et al., 2022) | Marine | Global | Multiple groups (animals, plants, chromists, protozoans, and bacteria) | 24,975 | Almost 90% of all species are at high or critical risk under high emissions, with exploited species in low-income countries with heavy dependence on fisheries at greatest risk. |
| (Nyboer et al., 2021) | Freshwater / Marine | Global | Fish | 415 | Over 20% of recreationally fishes are vulnerable under a high emission scenario, with 72% of vulnerable freshwater fish and 33% of vulnerable diadromous fish being without conservation effort, compared to only 19% for vulnerable marine species. |
| (Vaz-Canosa et al., 2023) | Terrestrial | Uruguay | Amphibians  Reptiles | 112 | 14.6% of amphibians and 10.9% of reptiles were identified as highly vulnerable to climate change. |
| (Bueno-Pardo et al., 2021) | Marine | Portugal | Fish  Invertebrates | 74 | Under the RCP 8.5 scenario, only two species were classified as at very high vulnerability. Overall vulnerability scores were low, likely due to the high adaptive capacity of species from temperate ecosystems. |
| (Leclerc, Courchamp, et al., 2020) | Terrestrial – insular | Global | Mammals | 873 | All islands have some degree of vulnerability to future climate change, especially those in the Pacific Ocean. Among endemic mammals, those with long generation times and high food specialization are predicted to be most vulnerable to climate change. |
| (Ramírez-Bautista et al., 2020) | Terrestrial | Oaxaca State, Mexico | Rodents | 55 | Under the higher impact (MPI-RCP 8.5) climate scenarios, some level of threat was predicted for all species assessed, with 4 species predicted to be highly vulnerable. |

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