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

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46

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50

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64 **Abstract:**

65 **The majority of vulnerability assessments of biodiversity to global changes have so far**  
66 **been applied to, and designed for, mainland systems, overlooking islands. However,**  
67 **islands harbour unique biodiversity and are epicentres of ongoing extinctions. We thus**  
68 **introduce a specific framework for quantifying the vulnerability of terrestrial insular**  
69 **biota to multiple threats. This framework uses markers of exposure, sensitivity, and**  
70 **adaptive capacity to account for the unique characteristics of island biodiversity. Our**  
71 **assessment framework involves five steps: (1) defining the scope of the vulnerability**  
72 **assessment, (2) selecting the most appropriate markers, (3) computing the vulnerability**  
73 **metric, (4) evaluating uncertainties, and (5) providing recommendations for**  
74 **conservation. The development of this vulnerability framework tailored for island**  
75 **systems is part of a larger initiative to meet international policy targets that better**  
76 **integrate biodiversity threats and dimensions. We thus discuss the need and urgency for**  
77 **applying this framework to guide evidence-based decisions for the conservation of**  
78 **insular biodiversity, and for increased attention to insular biota at the science-policy**  
79 **interface.**

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## 86 **Introduction**

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

95 Island ecosystems are disproportionately vulnerable to threats for three key reasons. Most  
96 insular species are intrinsically more sensitive to any given threat due to their specific traits  
97 deriving from the so-called 'island syndrome' (Benítez-López et al., 2021; Biddick et al.,  
98 2019; Lomolino, 1985; Rozzi et al., 2023). Second, insular species are less likely to adapt to  
99 new threats due to their inherent demographic features (e.g., small population sizes, naturally  
100 fragmented distribution ranges). Lastly, the physiography of islands, specifically in the case of  
101 isolated, small-sized ones, renders their biota more exposed to threats and also less able to  
102 escape compared to their mainland counterparts (Fernández-Palacios et al., 2021; see also the  
103 "uniqueness of insular biota" section for a more complete description of those inherent  
104 vulnerabilities).

105 Despite the urgency of the need to protect unique and highly vulnerable island biodiversity  
106 from ongoing global change, insular biota receives very cursory attention in international  
107 biodiversity policy frameworks. In December 2022, the 196 parties of the Convention on  
108 Biological Diversity (CBD) adopted the Kunming-Montreal Global Biodiversity Framework  
109 (KM-GBF), involving 23 action-oriented global targets. Among them, only one target (Target  
110 6) explicitly mentions islands as priority areas for conservation (though not explicitly

111 mentioning insular biota) and two other targets (Target 3 and 4, GBF) require strong actions  
112 on islands, reflecting a broader scientific bias toward mainland (Rodrigues et al., 2010). Since  
113 the emergence of climate change vulnerability assessments, the vast majority of those  
114 assessments have been designed for, and applied to, mainland ecosystems. When extended to  
115 islands, these frameworks are often mere adaptations of those developed for mainland  
116 systems. This approach is particularly problematic, as it fails to account for the unique  
117 complexities and specific characteristics of insular biota. Consequently, our knowledge of the  
118 vulnerability of insular ecosystems and species is incomplete and results most likely in an  
119 underestimation of their global vulnerability.

120 Here, our objective is to introduce an adapted framework that addresses the limitations of  
121 existing approaches by incorporating the idiosyncrasies of island biota. This framework is  
122 specifically designed to quantify the vulnerability of terrestrial insular biota to multiple threats  
123 and aims to capture the distinct challenges posed by the unique attributes of insular  
124 ecosystems, such as the island syndrome, the isolated nature of islands, and their high levels  
125 of endemism (see also Figure 1A). The goal is also to provide improvements of existing  
126 vulnerability frameworks that are not specific to island ecosystems (*e.g.*, the inclusion of  
127 multiple threats and dimension of diversity). We define vulnerability in this framework across  
128 multiple biodiversity dimensions, considering the exposure, sensitivity, and adaptive capacity  
129 of insular biota to multiple threats. The originality of our framework is that it is specifically  
130 designed for insular biodiversity, with the inclusion of multiple threats, taxa, and dimensions  
131 of diversity, such as functional and phylogenetic diversity, as well as the inclusion of  
132 vulnerability markers at species, community and assemblage levels of islands. We first  
133 describe the characteristics that contribute to the vulnerability of island biota to global  
134 changes, which need to be considered in insular biodiversity vulnerability assessments. Then,

135 we describe how our framework can address some of the ongoing questions on island biota  
136 vulnerability.

### 137 **Challenges**

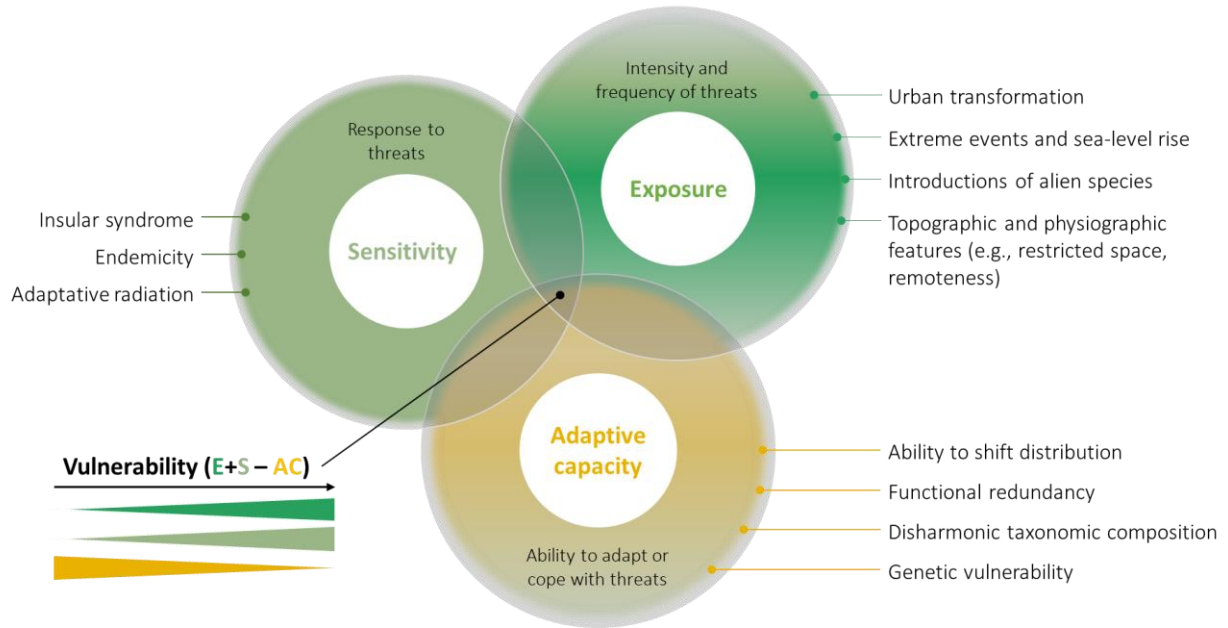
138 *The uniqueness of insular biota increases its vulnerability to threatening processes*

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

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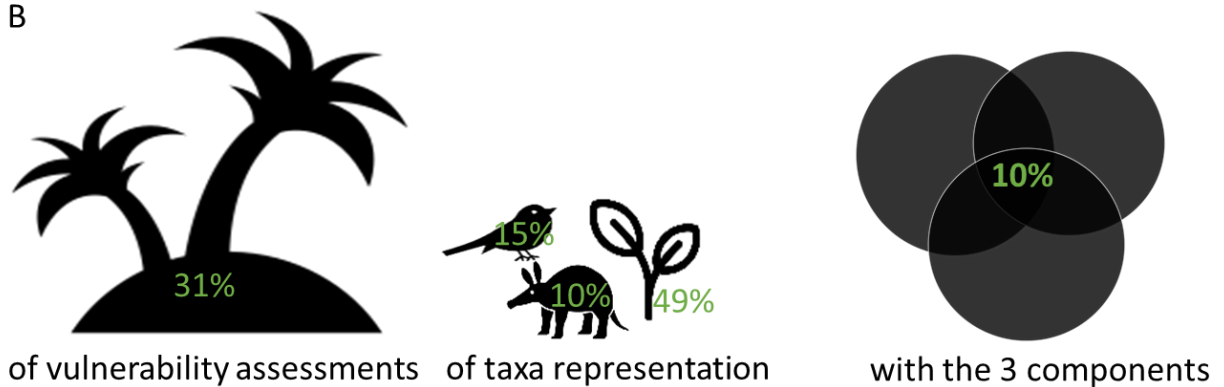
A

## Why is island biota particularly vulnerable to global changes?



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B



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

164

165 Insular species have also evolved distinct and unique ecological, physiological, behavioural,  
 166 morphological, and life-history traits, a phenomenon commonly referred to as the ‘island



167 syndrome' (Baeckens & Van Damme, 2020). These distinct and unique traits increase their  
168 sensitivity to current and future anthropogenic threats, including overharvesting, habitat loss,  
169 increased drought and invasive alien species (Rozzi et al., 2023; Zizka et al., 2022) (Figure  
170 1A). This phenomenon is for example associated with dioecy in plants, which makes them  
171 more likely to disappear if their mutualists' species go extinct, increasing their vulnerability to  
172 human-induced perturbations (Schrader et al., 2021). Reduced herbivory and predation  
173 pressure on islands leads to low levels of spinescence compared to their mainland  
174 counterparts, making island species more susceptible to the introduction of exotic herbivores  
175 (Burns, 2016; Clavero et al., 2009), unless they have coevolved with megafaunal herbivores  
176 (Barton et al., 2024). The loss of flight capacity renders insular birds unable to escape from  
177 anthropogenic threats (Sayol et al., 2021), making them more susceptible to extreme weather  
178 events, such as hurricanes, or gradual environmental change (e.g., progressive directional  
179 changes such as climate change) (Burns, 2019; Roff, 1990).

180 Additionally, the intricate and often rugged topography of oceanic islands frequently leads to  
181 long-term isolations of populations, which may result in genetic drift and then inbreeding  
182 when population sizes are small (Frankham, 2002). Furthermore, island colonization within  
183 archipelagos, and isolation by distance, could also lead to high population structure between  
184 islands (Parmakelis et al., 2015; White & Searle, 2007). These populations are thus more  
185 prone to genetic diversity loss, which may result in fewer opportunities to adapt to changes.  
186 Although this does not necessarily increase their vulnerability *per se*, the loss of genetic  
187 diversity due to natural or anthropogenic disturbances, especially when exposed to  
188 disturbances over a short time period (Inamine et al., 2022), can cause a demographic or  
189 genetic collapse. Thus, losing an insular population can cause demographic and genetic loss,  
190 whereas it causes only demographic loss on the mainland. Because of that, the effective

191 conservation units on islands should be, in most cases, at the population level, rather than  
192 species level (Melo & O’Ryan, 2007).

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

208 *State of the art of vulnerability assessments and their limitations when applied in insular*  
209 *contexts*

210 Despite the inherent vulnerability of insular biota to global changes, most biodiversity  
211 vulnerability assessments have been designed for and focused on mainland systems. Climate  
212 change vulnerability assessments emerged in the 1990s to anticipate impacts and prepare  
213 appropriate responses (Foden et al., 2018). Most definitions concur that a species’  
214 vulnerability to a threat is determined by three components: **exposure**, **sensitivity**, and  
215 **adaptive capacity** (Butt et al., 2022; Foden, Butchart, et al., 2013; Foden et al., 2019). Using

216 this definition, a recent review showed that among the 741 studies assessing climate change  
217 vulnerability, the majority focused on mainland systems, with less than one third ( $n = 231$ )  
218 including islands (de los Ríos et al., 2018) (see Table S2 for examples) and only 136 studies  
219 associated with a specific insular country. Although this would be representative of the small  
220 land surface area occupied by islands (6.7%), it falls short in terms of biodiversity  
221 representativeness, as island's biodiversity represents 20% of biodiversity worldwide, with  
222 50% of threatened and 75% of known extinctions (Fernández-Palacios, Otto, et al., 2021).  
223 These island vulnerability assessments are often geographically and taxonomically restricted  
224 towards high income countries (e.g., Australia, the UK, and the USA account for 60% of  
225 studies on insular biota) and plants (49% of studies) (Figure 1B), respectively. In addition, the  
226 large majority of vulnerability assessments do not consider the influence of multiple threats  
227 (but see Santos et al., 2021; Sousa et al., 2021; Ureta et al., 2022), which are likely to act  
228 together, interact with and be exacerbated by climate change (e.g., habitat loss,  
229 overexploitation, biological invasions), potentially leading to synergistic impacts (Leclerc et  
230 al., 2018). Note that these geographic, taxonomic, or conceptual biases occur in both  
231 mainland and insular assessments (see also de los Ríos et al., 2018). Finally, those  
232 assessments do not consider other dimensions of diversity, neither they take into account the  
233 specificities of insular biota (e.g., phylogenetic endemism, geographic isolation of islands,  
234 population size, etc.) that make them more likely to be vulnerable to global changes..

235 Assuming that exposure is a suitable proxy for the impact of a particular threat, most  
236 vulnerability assessments focused on exposure alone, with less than 10% considering the three  
237 components of vulnerability (Butt et al., 2016; de los Ríos et al., 2018). This assumes that all  
238 species will have equal responses to a threat, which is highly unlikely. In fact, the likelihood  
239 of species being impacted by certain threats, either on the mainland or on islands, is  
240 modulated by their traits (Fromm & Meiri, 2021; Leclerc, Villéger, et al., 2020; Marino et al.,

241 2022; Soares et al., 2022). We argue that trait-based vulnerability assessments, when applied  
242 to a range of taxa and threats, can provide a useful approach for (i) developing a more  
243 comprehensive index of vulnerability to threats, and (ii) informing effective management  
244 actions for conservation (Gallagher et al., 2021).

## 245 **An adapted framework for assessing the vulnerability of island biota to multiple threats**

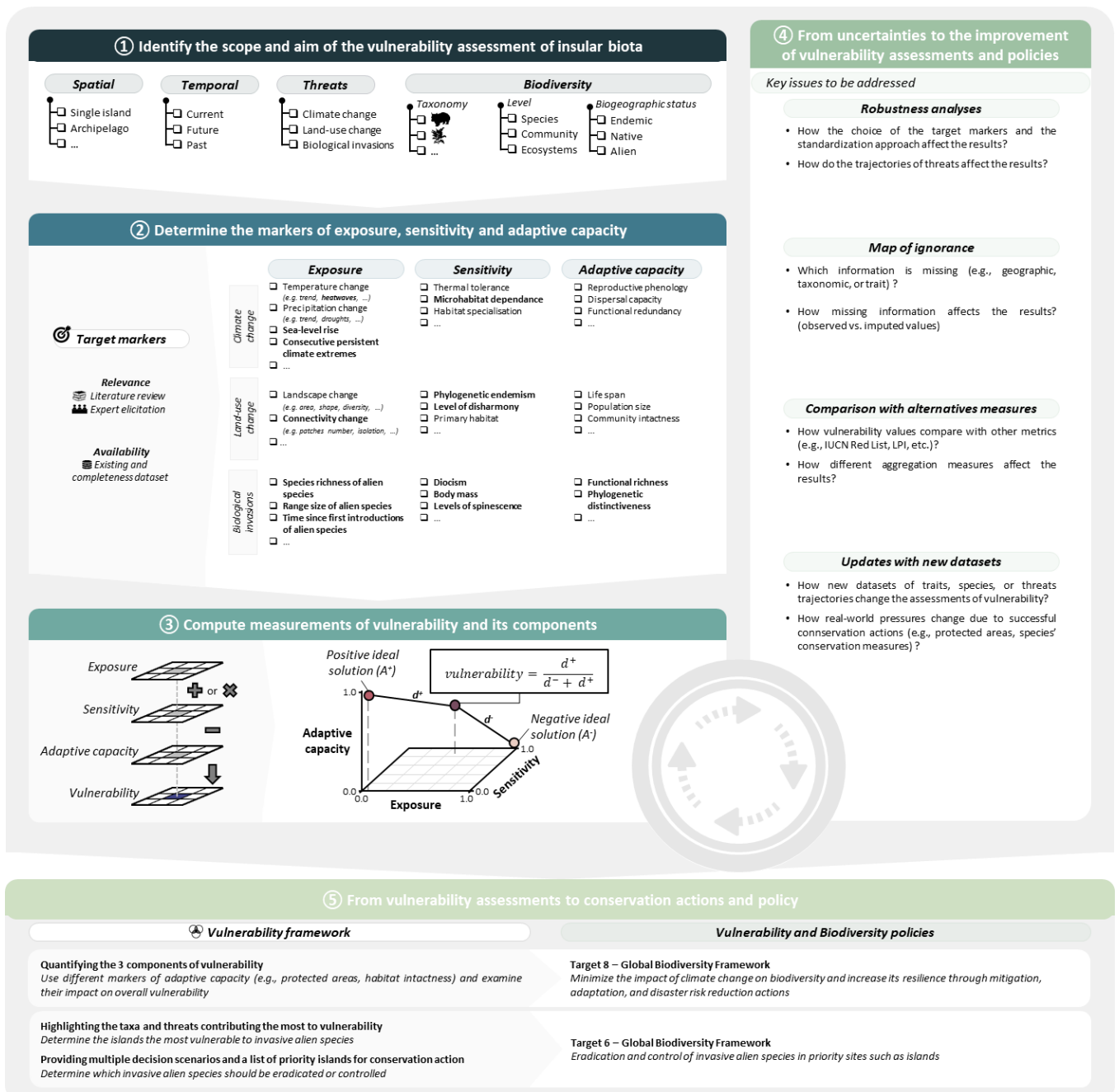
246 We present a conceptual framework divided into five steps that provides a roadmap for  
247 vulnerability assessments of biota in insular systems, considering the three components of  
248 vulnerability (i.e., exposure, sensitivity, and adaptive capacity; Figure 2). Here, we refer to  
249 islands as insular systems that have a landmass smaller than Greenland (i.e., < 2.17 million  
250 km<sup>2</sup>) and are surrounded by sea water. We built upon previous framework, which focused on  
251 taxonomic diversity in mainland ecosystems (Butt et al., 2022; Foden, Butchart, et al., 2013;  
252 Leclerc, Courchamp, et al., 2020; Parravicini et al., 2014) to include in our framework various  
253 markers for each vulnerability component that are specifically tailored to the inherent  
254 characteristics of island biota, as well as multiple threats, taxa, and dimensions of diversity  
255 (i.e., taxonomic, functional and phylogenetic diversity). The development of this vulnerability  
256 framework tailored for island systems is part of a larger initiative to meet international policy  
257 targets that better integrate biodiversity threats and dimensions (Box 2).

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

259 The first step is to define the scope of the vulnerability assessment in terms of spatial and  
260 temporal extent, relevant threats, and studied biota (taxonomic group, biological level,  
261 biogeographical origin of species). This challenging task is pivotal for the assessment design,  
262 which in turn affects the final step of informing conservation actions (Step 5). For example,  
263 broad-scale assessments (e.g., at the global extent or among several archipelagos) contribute

264 to strategic planning and to establish a common baseline of vulnerability information (e.g.,  
265 IPBES assessments), while local-scale assessments (e.g., within archipelagos or group of  
266 islands) are appropriate for informing site-level management decisions.

267 For instance, a vulnerability assessment could be conducted at the spatial extent of a national  
268 park, within an island of a few hectares only, with a restricted set of species (e.g., Harper et  
269 al., (2022), 24 ha in South Africa, 18 amphibian and 41 reptile species). This can inform  
270 management priorities at the landscape scale, such as defining park-use zones to help allocate  
271 restricted areas acting as corridors for species migration, or creating habitat conditions for  
272 breeding (Harper et al., 2022). At this level, an explicit treatment of population genetics  
273 and/or population viability analyses could also be conducted, and this may become more  
274 feasible in the future with the emergence of macrogenetics studies (Leigh et al., 2021). In  
275 parallel, studies focusing on the global extent are key to assessing vulnerability metrics,  
276 identifying geographical shortfalls in data coverage, and supporting the implementation of  
277 conservation policies to mitigate biodiversity losses. Note that, in all cases, controlling for  
278 island area or species richness is essential to mitigate biases towards larger islands when  
279 calculating vulnerability metrics. Finally, even if vulnerability assessments are conducted at  
280 the global scale, particular attention should be directed toward endemic species or insular  
281 populations, since island biodiversity conservation operates at the population level.



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284 **Figure 2:** Conceptual framework for assessing the vulnerability of island biodiversity to  
 285 global changes. The framework consists of five steps and can be iteratively applied (see  
 286 circular arrow). Markers in bold font represent the markers specifically designed for insular  
 287 biota but other markers are also important to indicate species' sensitivity, independently of  
 288 their occurrence on islands or mainland.

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

290 There are many different relevant markers associated with biodiversity' vulnerability to  
291 threats. Those markers may vary in type (e.g., ecological or demographic) and organisation  
292 level (e.g., population, species, community, or ecosystem), and could be considered either  
293 simultaneously or separately in the framework. Literature review and expert opinion can be  
294 used to identify and collect relevant markers for each component of vulnerability based on the  
295 scope and purpose of the assessment. For instance, assessing the vulnerability of terrestrial  
296 species to land-use change and climate change requires markers of exposure to climate change  
297 (e.g., change in precipitation regimes, sea-level rise) and land-use change (e.g., urbanization  
298 rates, forest conversion into agriculture, infrastructure development, shift from agriculture to  
299 tourism) (e.g., Bellard, Leclerc, & Courchamp, (2015); Bellard, Leclerc, Hoffmann, et al.,  
300 (2015)). Moreover, it is necessary to list the biological traits related to the sensitivity and  
301 adaptive capacity of the studied insular species to the above-listed threats (Figure 2 and Table  
302 S1 for examples of exposure, sensitivity, and adaptive capacity markers). Markers of exposure  
303 can be linked to the focal threat(s) or general global changes, and multiple markers should be  
304 used to capture the multiple dimensions of changes (e.g., temperature change, heat waves,  
305 droughts).

306 Species' sensitivities are determined by their intrinsic traits (e.g., climatic niche breadth,  
307 habitat specialization, diet breadth, body size, etc.). Despite the increasing availability of traits  
308 for birds, mammals, reptiles, and plants (Díaz et al., 2022; Faurby et al., 2018; Soria et al.,  
309 2021; Tobias et al., 2021), trait-based approaches have been barely applied to date or they are  
310 specific to some threats. Some sensitivity traits might be specific to a given threat (e.g.,  
311 temperature tolerance), while others are general and encompass sensitivity to different threats  
312 (e.g., body size, dependence on interspecific interactions). Note that for both sensitivity and  
313 adaptive capacity, a comprehensive list of markers is applicable at either the population or  
314 species level (see Thurman et al., 2020 for a list and Table S1). However, markers applicable

315 at the community level, such as functional or phylogenetic diversity, could also be included  
316 (Table 1 for example). Using extinctions scenarios for threatened species, we can also  
317 establish which islands are more likely to lose a significant share of their functional and  
318 phylogenetic diversity (richness, redundancy, originality, uniqueness) (Bellard et al., 2021;  
319 Llorente-Culebras et al., 2024). Those analyses could help identify islands that are more likely  
320 to be sensitive at the community level.

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

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	<b>Functional-based markers</b>	<b>Phylogeny-based markers</b>
<b>Sensitivity</b>	<p><b><i>Functional rarity</i></b></p> <p><i>Definition</i> - 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).</p>	<p><b><i>Phylogenetic endemism</i></b></p> <p><i>Definition</i> - 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.</p>

339 **Table 1:** Proposed functional-based and phylogeny-based metrics that could be used as  
340 markers at the community level in vulnerability assessments.  
341

	<i>Rationale</i> - 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.	<i>Rationale</i> - 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.
<b>Adaptive capacity</b>	<b><i>Functional redundancy</i></b>	<b><i>Phylogenetic distinctiveness</i></b>
	<i>Definition</i> - Functional redundancy of a given community reflects the tendency for the constituent species to perform similar functions (Mouillot et al., 2014).  <i>Rationale</i> - 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.	<i>Definition</i> - Phylogenetic distinctiveness reflects the degree of isolation of a species or a group of species within a phylogenetic tree (Pavoine & Ricotta, 2021).  <i>Rationale</i> - 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.

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

343 Once all the markers of the three components of vulnerability are collected, the next step is to  
344 combine them all into a composite vulnerability measure. However, aggregating multiple  
345 markers into a single value for each of the three vulnerability components is challenging. A  
346 recent review showed that most assessments use arithmetic mean for the aggregation  
347 (Tonmoy et al., 2014). In the case of quantitative markers, as here, one possibility is to  
348 normalise each marker to a 0-1 range. This transformation, which can be done with multiple  
349 methods (e.g. Leclerc et al., 2020), creates unitless metrics with equal weight. Then the  
350 markers of each component (i.e., exposure, sensitivity, and adaptive capacity) can be summed  
351 and re-scaled to obtain normalized values of exposure, sensitivity, and adaptive capacity. This

352 technique has the advantage of clearly identifying which components drive vulnerability, by  
353 ensuring that all markers are weighted equally, and can thus effectively guide the  
354 implementation of conservation actions at the island level. The different markers could also  
355 be weighted differently to put more emphasis on specific markers depending on the current  
356 level of island protection policy or biodiversity richness occurring on each island. Finally,  
357 when all the data have been aggregated, the majority of the current methods employ criteria-  
358 based approaches, classifying biota into categories of vulnerability from low to high by  
359 summing up the different components of vulnerability (hereby referred to as qualitative or  
360 semi-quantitative frameworks).

361 To avoid arbitrary thresholds, we propose a quantitative framework using multicriteria  
362 decision analysis (Leclerc, et al., 2020; Parravicini et al., 2014) that provides continuous  
363 vulnerability values and a relative ranking of islands or archipelagos. This method ranks  
364 alternatives according to their distance to positive (i.e., minimal vulnerability, with low  
365 exposure and sensitivity, and high adaptive capacity) and negative ideal solutions (see  
366 (Leclerc, Courchamp, et al., 2020) for details). Note that alternative approaches exist for  
367 combining the three vulnerability components, such as additive effect between components  
368 (Nyboer et al., 2021) or interacting effect between exposure and sensitivity for instance (Silva  
369 Rocha et al., 2024). In any case, we recommend carrying out robustness analyses to explore  
370 the uncertainty potentially introduced by the aggregation and weighting methods.

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

373 The outcomes of vulnerability assessments can be affected by several factors, including  
374 missing data, variation of underlying traits, aggregation techniques, and, in the case of future-  
375 looking assessments, by uncertainties about the future trajectories of anthropogenic threats.  
376 Techniques to estimate uncertainty from missing trait data (e.g., Hossain et al., 2019) as well

377 as uncertainty in modelling the future of biodiversity (e.g., Thuiller et al., 2019) are readily  
378 available. In addition, we recommended carrying out robustness analyses to assess the impact  
379 of including or excluding specific markers from the calculation, by rerunning the vulnerability  
380 assessment without the focal marker. Robustness analyses may also be used to explore the  
381 effects of alternative aggregation or standardisation techniques (Tonmoy et al., 2014, see also  
382 Boyce et al., 2022). We also recommend validating the outcome of the vulnerability  
383 assessments with data from another temporal period (e.g., by comparing with past data), or by  
384 comparing with other islands using documented vulnerability or other metrics of vulnerability  
385 (e.g., IUCN Red List of species, Living Planet Index). In this context, it is crucial to both  
386 estimate and communicate the uncertainty. This can be achieved by, for example, mapping  
387 the variance of the estimated vulnerability metric or using maps of ignorance (e.g., Rocchini  
388 et al., 2011; Tessarolo et al., 2021). As well as identifying future research priorities,  
389 uncertainty evaluations strengthen confidence in assessments, thereby increasing their uptake  
390 to support policy decisions, including by the IPCC and IPBES (Joyce et al., 2011; Vadrot,  
391 2020). We also advocate for the integration of interactive interfaces and dashboards with  
392 options to select threats, markers, taxa, or ecosystems of interest, as well as different  
393 narratives trajectories, which may complement the scientific message - particularly helpful  
394 for simplifying multidimensional information (McInerny et al., 2014). Finally, we recommend  
395 conducting an ongoing review and iteration to enable adaptation and improvement of the  
396 vulnerability framework including as new ecological data become available (Henry et al.,  
397 2024).

#### 398 *Step 5: From vulnerability assessments to conservation actions and policy*

399 Effective conservation practice under global change relies on identifying which species are  
400 most vulnerable and what drives their vulnerability (Box 2, Step 5). Vulnerability assessments  
401 such as those described in Steps 1-4 provide invaluable information for conservation priority-

402 setting for island biodiversity. For species of the highest vulnerability, for example, the  
403 specific traits and exposure measures driving the vulnerability can inform appropriate  
404 responses (e.g., captive breeding, assisted translocation, disease management). For species  
405 identified as sensitive to climate change, but which may escape impact through high adaptive  
406 capacity, such responses (e.g. range shifts, phenological change) should be facilitated and  
407 monitored (Foden et al 2013). Species with poor adaptive capacity but low sensitivity can be  
408 assumed to be able to withstand change *in situ*, but close monitoring should be maintained to  
409 ensure that this assumption is realised (Foden et al, 2013) and that unexpected thresholds are  
410 not crossed. Assessments can also inform landscape-level actions (e.g. protecting hotspots of  
411 high numbers of vulnerable species, habitat restoration, and increasing permeability) (Harper  
412 et al., 2022).

413 Biodiversity policy too may be guided by species vulnerability assessments. For instance, one  
414 of the targets of the KM-GBF is to minimize the impact of climate change on biodiversity  
415 through (among others) mitigation, adaptation and risk disaster reduction actions. Such aims  
416 require the use of different markers of adaptive capacity through multiple plausible scenarios  
417 (e.g., protected areas, habitat intactness) to inform how different management actions can  
418 impact the vulnerability. Vulnerability assessments can help target the control and eradication  
419 of invasive alien species in conservation priority sites such as islands (Target 6), and also  
420 identify the sites that need to be brought under protection (Target 3) to conserve biodiversity.  
421 Monitoring progress in meeting these conservation targets too may be informed by  
422 vulnerability assessments. Since species sensitivity and adaptive capacity traits change little  
423 over time, while exposure increases at an observable rate, changes in vulnerability may also  
424 provide a measure of the relative effectiveness of conservation actions implemented (Foden,  
425 Mace, et al., 2013). Because the exposure component is common across species (e.g.,  
426 inundation, invasive alien species), the threat itself may be managed through dedicated

427 programs, such as mangrove rehabilitation or invasive alien species control (Jones et al.,  
428 2016).

429 Finally, beyond its use in current policies, our vulnerability framework could be useful in  
430 drafting new policies. Indeed, to set relevant targets in the new policies being drafted (e.g., by  
431 [year], reduce by [number] % the number of species threatened by [threat]), policymakers  
432 need to know (i) the current situation and (ii) what could be the future situation under  
433 different scenarios. The proposed vulnerability framework has the potential to provide both  
434 aspects and may contribute to the establishment of policy targets that are ambitious and  
435 attainable for insular biodiversity.

436

#### 437 **Concluding remarks**

438 Given the proliferation of multiple threats that islands are facing, efforts must be made to  
439 study island biota in the light of global changes, and thus to embrace the whole of  
440 biodiversity, mainland and insular, in current vulnerability assessments in international arenas  
441 such as the IPCC or IPBES. Our comprehensive and detailed framework lays the foundations  
442 for understanding and predicting island biodiversity vulnerability to global change. Besides,  
443 this framework aims to integrate island-specific characteristics with enhancements to existing  
444 frameworks that may also prove beneficial for continental assessments. These enhancements  
445 include the consideration of multiple threats, indicators that reflect phylogenetic and  
446 functional diversities, and an uncertainty analysis that is essential for all types of vulnerability  
447 assessments. Because the outcomes of vulnerability assessments will be challenged by  
448 missing data and, in the case of future assessments, by uncertainties about the future  
449 trajectories of anthropogenic threats, it is crucial to both estimate and communicate

450 uncertainty (e.g., Hossain et al., 2019; Rocchini et al., 2011; Tessarolo et al., 2021; Thuiller et  
451 al., 2019), which will ultimately help to protect biodiversity with more robust information.

452

453

454 **Glossary:**

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

458

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

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

464

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

467

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

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

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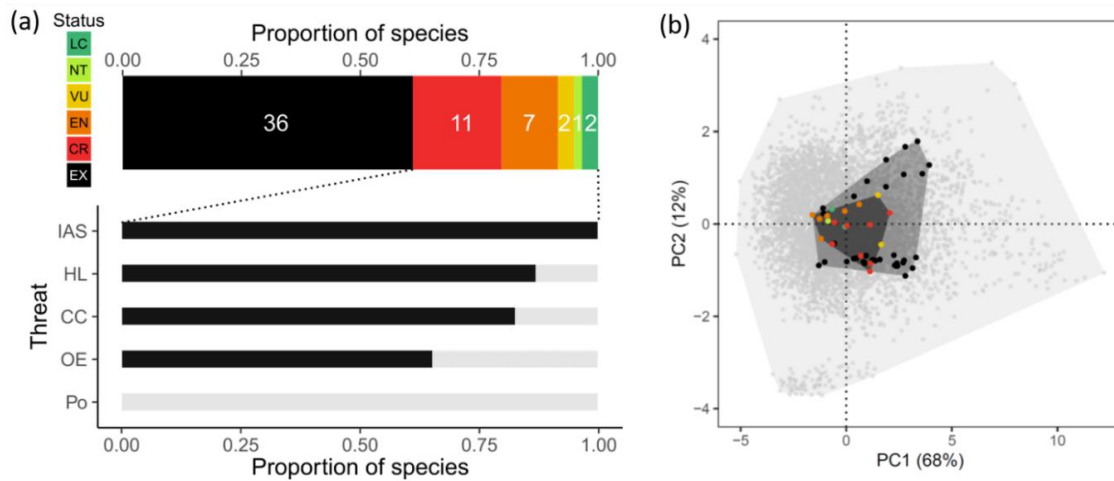
**Box 1 | The case of the Hawaiian honeycreepers**

483 The Hawaiian Islands constitute an archipelago of eight larger and numerous smaller volcanic  
484 islands in the North Pacific Ocean, in total about 16,644 km<sup>2</sup> in size, and about 3,200 km from  
485 the nearest mainland. The islands in the northwest are older and typically smaller, due to  
486 progressive erosion, while the islands in the southeast are larger and still volcanically active.

487 Thanks to the archipelago's extreme isolation and the islands' variation in size and  
488 environmental conditions, Hawaii has been a hotspot of speciation and adaptive radiation  
489 (Baldwin & Sanderson, 1998; Lerner et al., 2011; Price, 2004). The Hawaiian honeycreepers  
490 (Fringillidae: Drepanidinae) represent one of the most iconic examples of adaptive radiation,  
491 which has resulted in a striking variation in bill morphology (Lerner et al., 2011). This strong  
492 adaptive radiation, however, has also led to species with highly restricted ranges and naturally  
493 small population sizes. Hence, little is needed to push these species over the brink of  
494 extinction and, to date, at least 36 of the 59 known species of honeycreepers are extinct  
495 (Figure 3a). Extinction has been non-random with respect to the functional trait space of the  
496 species, with extinct species generally larger in size (Figure 3b). The main drivers of past  
497 extinction, as well as current threats to the species, are the introduction of invasive alien  
498 species and the loss of habitat due to conversion to agricultural land. Invasive species include  
499 predators of birds and eggs (rats, cats, dogs), herbivores that modify the habitat (for example,  
500 the extinction of Laysan Honeycreeper *Himatione fraithii* is ascribed to the introduction of  
501 rabbits, which eliminated virtually all vegetation cover from the Laysan islands), and vector  
502 species of infectious diseases, such as avian malaria (Benning et al., 2002). Habitat loss began  
503 with Polynesian colonists, who cleared much of the low-elevation and seasonally dry forest  
504 for agricultural purposes, and was continued by later European colonists, who additionally  
505 converted high-elevation forests for pasturage (Riper & Scott, 2017). Climatic factors  
506 contributed little to past extinctions but are a progressive threat to extant species, especially  
507 because it may lead to an upslope shift of infectious diseases (notably avian malaria)  
508 (Benning et al., 2002). Overall, this example illustrates the importance of (i) considering  
509 multiple threats when focusing on island biota, (ii) taking into account the singularities of  
510 insular syndrome such as restricted ranges, small population size or level of endemism, as  
511 well as (iii) the importance of considering markers of sensitivity at the community level given



512 the non-random distribution of species at risk of extinction and their potential higher  
513 vulnerability to global change.



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

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**Box 2 | Leveraging an insular vulnerability framework to inform multiple policies**

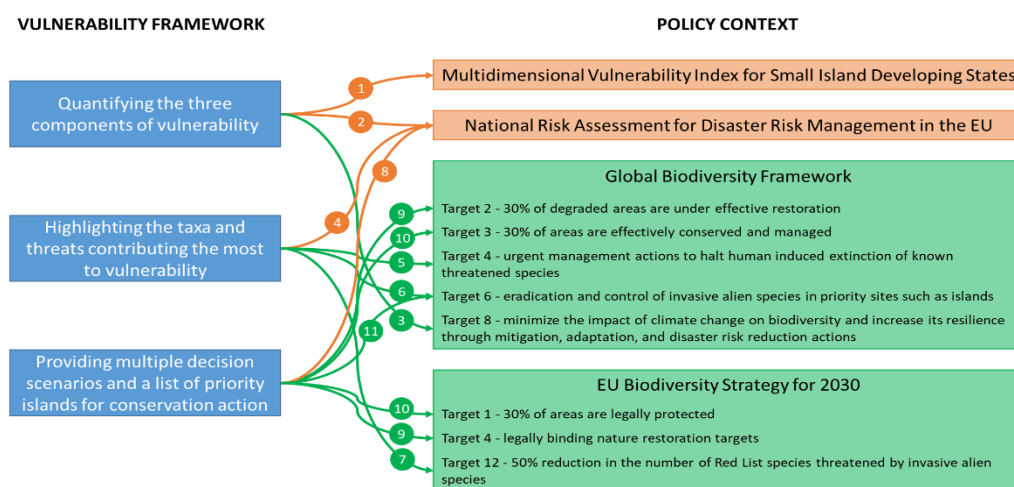
Our proposed vulnerability framework for insular biodiversity is particularly relevant for informing various vulnerability policies (in orange) and biodiversity policies (in green) at both international and supranational levels, such as the European Union (EU). First, by quantifying the three components of island vulnerability, our framework can directly provide 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/ohrls/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 disproportionately 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), which does not specifically cover insular biodiversity. In that context, our framework can provide real added value for EU Member States with insular territories. Our framework can also further help identify whether mitigation strategies for islands to reach Target 8 of the Kunming-Montreal Global

558 Biodiversity Framework (hereafter, GBF) should focus on limiting territories' exposure or  
 559 promoting the adaptive capacity of insular biota (3).

560 Second, our framework can help EU Member States with insular territories to identify and  
 561 prioritise which species, sites and threats they should use to best mitigate their vulnerability in  
 562 the context of their National Risk Assessment (4). Such actions can also contribute to (i)  
 563 identifying the species, sites and to focus the management measures to reach Target 4 of the  
 564 GBF (5), (ii) determining the most vulnerable islands to invasive alien species to reach Target  
 565 6 of the GBF (6), and helping EU Member States with insular territories to identify species  
 566 threatened by invasive alien species on which to focus conservation actions to reach Target 12  
 567 of the [EU Biodiversity Strategy for 2030](#) (EU BDS, (7)).

568 Finally, by providing multiple scenarios of threat trajectories and a list of priority islands for  
 569 conservation action, our framework can contribute to documenting the risk of biodiversity  
 570 loss under different conservation scenarios to EU Member States with insular territories (8).

571 The vulnerability framework could contribute to identifying priority islands for restoration  
 572 (target 2 of the GBF and target 4 of the EU BDS (9)) and for protection (target 3 of the GBF  
 573 and target 1 of the EU BDS (10)), as well as in determining which invasive alien species  
 574 should be eradicated or controlled (target 6 of the GBF (11)).



583 *This figure highlights some possible outcomes of the vulnerability framework and how they*  
 584 *are linked (arrows) to the vulnerability (in orange) and biodiversity policies (in green). Each*

585 | *circled number represents a concrete example of how our vulnerability framework can*  
586 | *support existing policies, and all those examples are detailed in the box.*

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# Supplementary material :

## 989 **Supplementary appendix 1 :**

990 **Islands as past, current, and future epicentres of extinctions:** The IUCN Red List  
991 documented nearly 1,000 species as globally extinct or extinct in the wild (IUCN Red list  
992 access 04/07/2023). Among the documented extinctions, the overwhelming majority occurred  
993 on oceanic or continental islands. Extinction hotspots are located in the Oceanic realm,  
994 followed by the Afrotropical realm (e.g., Madagascar) and the Nearctic. Among reported  
995 extinctions in the IUCN Red List, the large majority are represented by animal species  
996 (>80%), followed by plants. Most of these species have gone extinct because of invasive alien  
997 species, but a large number have also concurrently been harmed by wildlife exploitation  
998 and/or land use change due to cultivation (Leclerc et al., 2018; Maxwell et al., 2016). In fact,  
999 extinct species within insular regions have faced, on average, four threats within the IUCN  
1000 threat classification scheme (i.e., biological invasions, wildlife exploitation, cultivation, and  
1001 habitat modifications), and the exposure to threats for birds, mammals, amphibians, reptiles,  
1002 freshwater fish, plants, arthropods, and gastropods continue to increase, with an average of ten  
1003 threats being faced by threatened species (Leclerc et al., 2018).

1004 Moreover, islands makes a disproportionate contribution to global biodiversity in relation to  
1005 their surface area, especially regarding the percentage of threatened species (Fernández-  
1006 Palacios, Kreft, et al., 2021), the risk for future extinctions is thus high. The first extinction  
1007 due to climate change already occurred on an island near Papua New Guinea (Fulton, 2017).  
1008 In 2016, *Melomys rubicola* (also called the Bramble Cay melomys) became the very first  
1009 documented extinction due to climate change (Waller et al., 2017). This small rodent was  
1010 endemic to the low-island of Bramble Cay in Australia and was periodically recorded from

1011 1978 to 2009. Seven years after its last observation in the field and numerous efforts to trap  
 1012 the species in the years after, the species was officially declared extinct. Although only  
 1013 symbolic, compared to a thousand species already extinct since 1500, this is one of the first  
 1014 documented victims of climate change as a major factor of near-future extinctions in our  
 1015 contemporary history. Unfortunately, under future climate change, we can expect that islands  
 1016 will continue to contribute disproportionately to the loss of biodiversity, given that the  
 1017 overwhelming majority of critically endangered species are endemic to island systems, and  
 1018 their inherent characteristics make them extremely vulnerable to global change drivers

1019  
 1020 **Table S1:** Traits and characteristics from population to the community that may be included  
 1021 in the island biota vulnerability framework (IBVF) with examples of sources where to  
 1022 calculate or when directly available.  
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	<b>Traits</b>	<b>Level</b>	<b>Availability</b>
<b>Population dynamic</b>	Growth rate	population	Living planet database (vertebrates), FishGlob (fishes), German vegetation (plants)
	Population size	population	CoralTraits (corals), BIOTime (multiple taxa)
<b>Reproduction</b>	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-cycle	species	TRY (plants), GIFT (plants)
<b>Life-history</b>	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).
<b>Acclimatation or evolution</b>	Clonality	species	GIFT (plants), COMPADRE (plants)
	Phenotypic plasticity	population	Noble et al 2018 (reptiles)
	Genetic diversity	population	Miraldo et al. 2016 (mammals)
	Life span	species	AmphiBio (amphibians),
<b>Ecological characteristics</b>	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	
	Dispersal capacity	species	AVONET (birds), TRY (plants), GIFT (plants)
	Site fidelity	species	
<b>Movement and mobility</b>	Migration frequency and distance	species	AVONET (birds)
	Flight efficiency	species	AVONET (birds)
	Number of insular populations	population	
	Home range	population	HomeRange (mammals)
	Geographic rarity	population	
<b>Distribution</b>	Low local abundance	island level	
	Range size	population	IUCN
	Level of disharmony	community	
	Functional redundancy	community	TRY and GIFT(plants)
<b>Community properties</b>	Phylogenetic diversity	community	GIFT, WCVP (plants)
	Richness	community	GIFT, WCVP (plants)
	Intactness	ecosystem	
	Phylogenetic endemism	community	Vertlife (vertebrates), PhylomeDB (plants), GIFT, WCVP (plants)
	Behaviour (aggressiveness)	species	
<b>Inherent vulnerability</b>	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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1082 **Table S2:** An overview of studies assessing the vulnerability of species to climate change.  
1083 This is a non-exhaustive summary that focuses on studies that have been conducted recently,  
1084 use trait-based approaches, and include a full assessment of vulnerability with exposure,  
1085 sensitivity, and adaptive capacity markers.  
1086

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

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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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