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

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62 **Abstract:**

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

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83 **Introduction**

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

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

102 Despite the urgency needed to protect the unique and highly vulnerable island biodiversity
103 from ongoing global changes, insular biota is only briefly mentioned in international
104 biodiversity policy frameworks. In December 2022, the 196 parties of the Convention on
105 Biological Diversity (CBD) adopted the [Kunming-Montreal Global Biodiversity Framework](#)
106 (KM-GBF), involving 23 action-oriented global targets. Among them, only one target (Target
107 6) explicitly mentions islands as priority areas for conservation (though not explicitly

108 mentioning insular biota) and only two other targets (Target 3 and 4, GBF) require strong
109 actions on islands. This lack of attention reflects the scientific bias towards mainland or
110 charismatic species (Albert et al., 2018; Rodrigues et al., 2010), with remote islands receiving
111 less consideration in biodiversity assessments, hindering the development of effective
112 conservation plans on insular biota (Troudet et al., 2017).

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

126

127 **Challenges**

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

129 The inherent uniqueness of insular biota (e.g., endemic species, distinct traits, unique
130 lineages) increases their overall vulnerability to global change drivers (Fernández-Palacios,
131 Kreft, et al., 2021). Islands are isolated, have complex topographies, are prone to extreme

132 events (e.g. volcanic eruption, landslides, hurricanes), and harbour fewer potential refuge
133 (compared to mainland), making them more exposed to threats (Russell & Kueffer, 2019)
134 (Figure 1A). Islands also host a large number of endemic species: up to 90% of endemic non-
135 vagile taxa (i.e., non-flying vertebrates, seed plants, molluscs or arthropods) occur in islands
136 (e.g., Madagascar, Antonelli et al., 2022). Endemics have small population sizes and restricted
137 geographical ranges (sometimes limited to an archipelago, an island or even a single peak,
138 volcano or cliff) (Fernández-Palacios, Otto, et al., 2021), increasing their extinction
139 probability (Manes et al., 2021). An iconic example is the radiation of Hawaiian
140 Honeycreepers, a group of small birds particularly vulnerable to invasive alien species and
141 habitat modification (**Box 1**).

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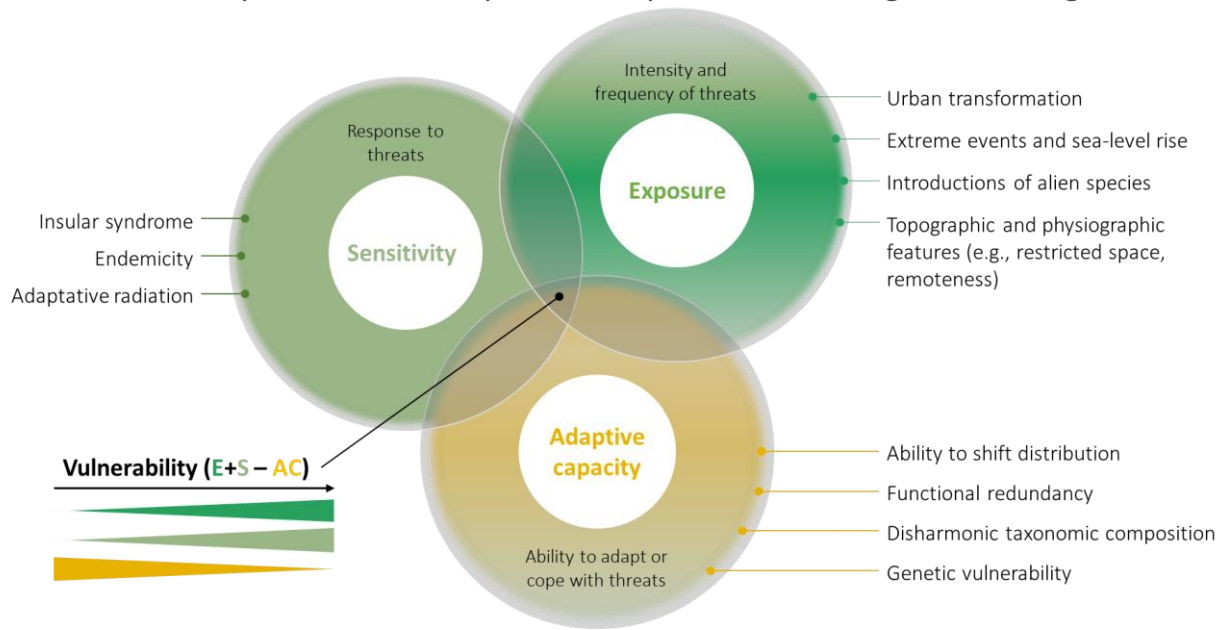
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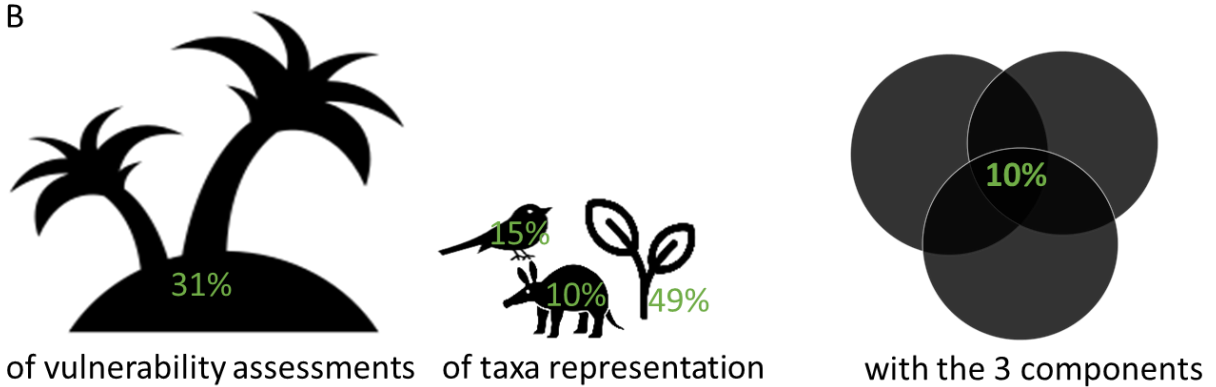
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Why is island biota particularly vulnerable to global changes?



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B



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

159

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

162 syndrome' (Baeckens & Van Damme, 2020). These distinct and unique traits increase their
163 sensitivity to current and future anthropogenic threats, including overharvesting, habitat loss,
164 increased drought and invasive alien species (Rozzi et al., 2023; Zizka et al., 2022) (Figure
165 1A). This phenomenon is for example associated with dioecy in plants, which makes them
166 more likely to disappear if their mutualists' species go extinct, presumably increasing their
167 vulnerability to human-induced perturbations (Schrader et al., 2021). Reduced herbivory and
168 predation pressure on islands leads to low levels of spinescence compared to their mainland
169 counterparts, making island species more susceptible to the introduction of exotic herbivores
170 (Burns, 2016; Clavero et al., 2009), unless they have coevolved with megafaunal herbivores
171 (Barton et al., 2024). The loss of flight capacities renders insular birds unable to escape from
172 anthropogenic threats (Sayol et al., 2021), making them more susceptible to extreme weather
173 events, such as hurricanes, or gradual environmental change (e.g., progressive directional
174 changes such as climate change) (Burns, 2019; Roff, 1990).

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

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

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

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

204 Despite the inherent vulnerability of insular biota to global changes, most biodiversity
205 vulnerability assessments have designed and focused on mainland systems. Climate change
206 vulnerability assessments emerged in the 1990s to anticipate impacts and prepare appropriate
207 responses (Foden et al., 2019; Solomon et al., 2007). Most definitions concur that a species’
208 vulnerability to a threat is determined by three components: **exposure**, **sensitivity**, and
209 **adaptive capacity** (Butt et al., 2022; Foden et al., 2013, 2019). Using this definition, a recent

210 review showed that among the 741 studies assessing climate change vulnerability, the
211 majority focused on mainland systems, with less than one third ($n = 231$) including islands (de
212 los Ríos et al., 2018) (see Table S2 for examples) and only 136 studies associated with a
213 specific insular country. Although this would be representative of the small land surface area
214 occupied by islands (6.7%), it falls short in terms of biodiversity representativeness, as
215 island's biodiversity represents 20% of biodiversity worldwide, with 50% of threatened and
216 75% of known extinctions (Fernández-Palacios, Otto, et al., 2021). These island vulnerability
217 assessments are often geographically and taxonomically restricted towards high income
218 countries (e.g., Australia, the UK, and the USA account for 60% of studies on insular biota)
219 and plants (49% of studies) (Figure 1B), respectively. In addition, the large majority of
220 vulnerability assessments do not consider the influence of multiple threats (but see Santos et
221 al., 2021; Sousa et al., 2021; Ureta et al., 2022), which are likely to act together, interact with
222 and be exacerbated by climate change (e.g., habitat loss, overexploitation, biological
223 invasions), potentially leading to synergistic impacts (Leclerc et al., 2018). Note that these
224 geographic, taxonomic, or conceptual biases occur in both mainland and insular assessments
225 (see also (de los Ríos et al., 2018).

226 Assuming that exposure is a suitable proxy for the impact of a particular threat, most
227 vulnerability assessments focused on exposure alone, with less than 10% considering the three
228 components of vulnerability (Butt et al., 2016; de los Ríos et al., 2018). This assumes that all
229 species will have equal responses to a threat, which is highly unlikely. In fact, the likelihood
230 of species being impacted by certain threats, either on the mainland or on islands, is
231 modulated by their traits (Fromm & Meiri, 2021; Leclerc, Villéger, et al., 2020; Marino et al.,
232 2022; Soares et al., 2022). We argue that trait-based vulnerability assessments, when applied
233 to a range of taxa and threats, can provide a useful approach for (i) developing a more

234 comprehensive index of vulnerability to threats, and (ii) informing effective management
235 actions for conservation (Gallagher et al., 2021).

236 **Perspective**

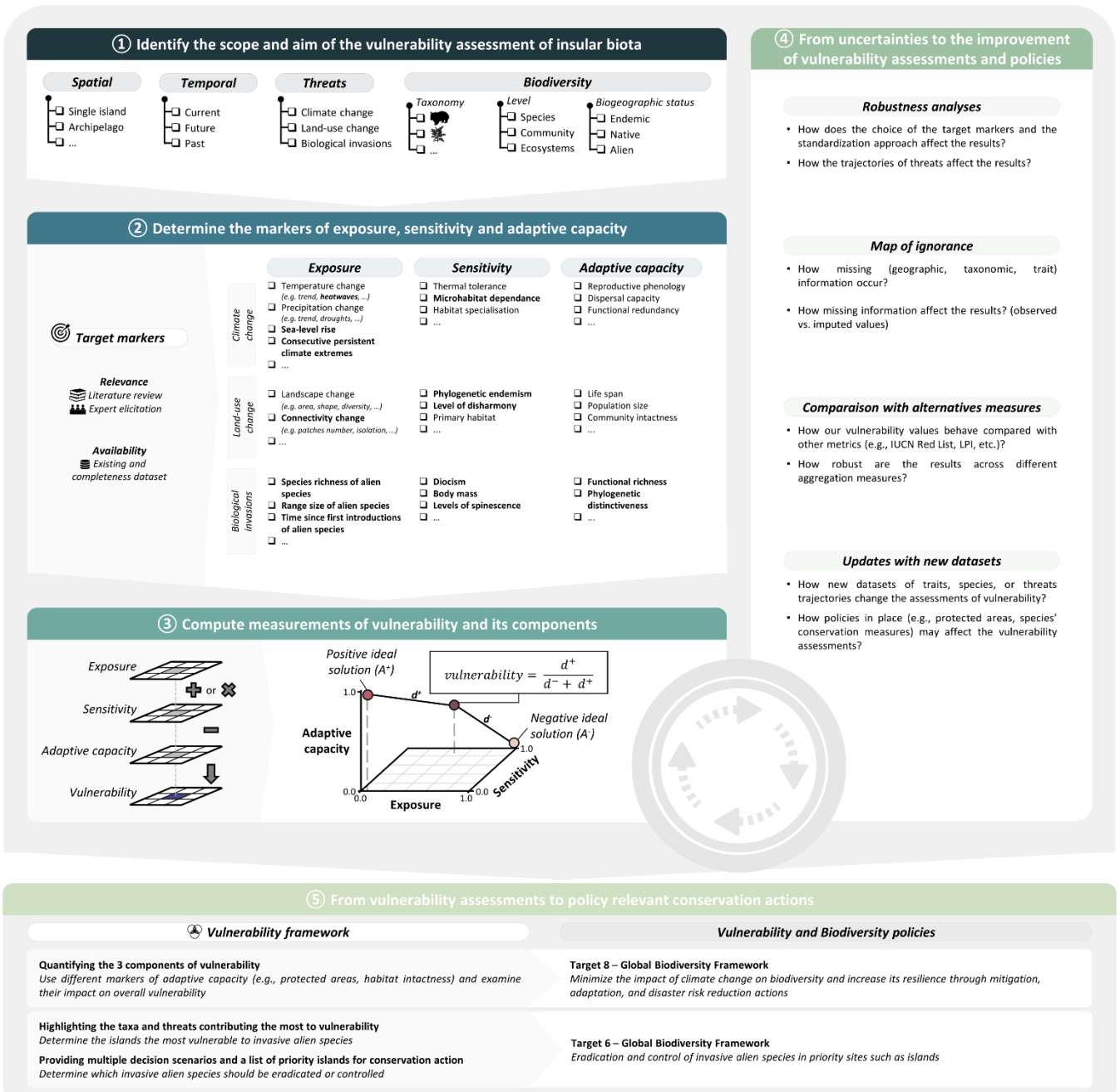
237 We present a conceptual framework divided in five steps that provides a roadmap for
238 vulnerability assessments of biota in insular systems, considering the three components of
239 vulnerability (i.e., exposure, sensitivity, and adaptive capacity; Figure 2). Here, we refer to
240 islands as insular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

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

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

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

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

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

280 There are many different relevant markers associated with species' vulnerability to threats.

281 Those markers may vary in type (e.g., ecological or demographic) and organisation level (e.g.,

282 population, species, community, or ecosystem). One can review the scientific literature and

283 elicit expert opinion to identify and collect relevant markers for each component of
284 vulnerability based on the scope and purpose of the assessment. For instance, assessing the
285 vulnerability of terrestrial species to land-use change and climate change requires markers of
286 exposure to climate change (e.g., change in precipitation regimes, sea-level rise) and land-use
287 change (e.g., urbanization rates, forest conversion into agriculture, infrastructure
288 development, shift from agriculture to tourism) (e.g., Bellard, Leclerc, & Courchamp, (2015);
289 Bellard, Leclerc, Hoffmann, et al., (2015). Moreover, it is necessary to list the biological traits
290 related to the sensitivity and adaptive capacity of the studied insular species to the above-
291 listed threats (Figure 2 and Table S1 for examples of exposure, sensitivity, and adaptive
292 capacity markers). Markers of exposure can be linked to the focal threat(s) or general global
293 changes, and multiple markers should be used to capture the multiple dimensions of changes
294 (e.g., temperature change, heat waves, droughts).

295 Species' sensitivity is determined by their intrinsic traits (e.g., climatic niche breadth, habitat
296 specialization, diet breadth, body size, etc.). Despite the increasing availability of traits for
297 birds, mammals, reptiles, and plants (Díaz et al., 2022; Faurby et al., 2018; Soria et al., 2021;
298 Tobias et al., 2021), trait-based approaches have been barely applied to date. Some sensitivity
299 traits might be specific to a given threat (e.g., temperature tolerance), while others are general
300 and encompass sensitivity to different threats (e.g., body size, dependence on interspecific
301 interactions). Note that for both sensitivity and adaptive capacity, a long list of markers is
302 applicable at either the population or species level (see (Thurman et al., 2020) for a list and
303 Table S1). However, markers applicable at the community level, such as functional or
304 phylogenetic diversity, could also be included (Table 1 for example). Using extinctions
305 scenarios for threatened species, we can also establish which islands are more likely to lose a
306 significant share of their functional and phylogenetic diversity (richness, redundancy,

307 originality, uniqueness) (Bellard et al., 2021; Llorente-Culebras et al., 2024). Those analyses
308 could help identify islands that are more likely to be sensitive at the community level.

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

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	Functional-based markers	Phylogeny-based markers
Sensitivity	<p><i>Functional rarity</i></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><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.</p>	<p><i>Phylogenetic endemism</i></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> <p><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.</p>
Adaptive capacity	<p><i>Functional redundancy</i></p> <p><i>Definition</i> - Functional redundancy of a given community reflects the tendency for the constituent species to perform similar functions (Mouillot et al., 2014).</p> <p><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.</p>	<p><i>Phylogenetic distinctiveness</i></p> <p><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).</p> <p><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.</p>

329 **Table 1:** Proposed functional-based and phylogeny-based metrics that could be used as
330 markers at the community level in vulnerability assessments.
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332

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

334 Once all the markers of the three components of vulnerability are collected, the next step is to

335 combine them all into a composite vulnerability measure. However, aggregating multiple

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

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

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

386 opportunity to adjust and update the vulnerability framework with new ecological data, such
387 as the IUCN perform periodic Red List (re-)assessments as new information becomes
388 available (Henry et al., 2024).

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

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

410

411 **Concluding remarks**

412 Given the proliferation of multiple threats that islands are facing, efforts must be made to
413 study island biota in the light of global changes, and thus to embrace the whole of
414 biodiversity, mainland and insular, in current vulnerability assessments in international arenas
415 such as the IPCC or IPBES. Our comprehensive and detailed framework lays the foundations
416 to understand and predict island biodiversity vulnerability to global changes. Yet, the
417 outcomes of vulnerability assessments will be challenged by missing data and, in the case of
418 future assessments, by uncertainties about the future trajectories of anthropogenic threats. In
419 this context, it is crucial to both estimate and communicate uncertainty (e.g., (Hossain et al.,
420 2019; Rocchini et al., 2011; Tessarolo et al., 2021; Thuiller et al., 2019)), which will
421 ultimately help to protect biodiversity with more robust information.

422

423

424 **Glossary:**

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

428

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

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

434

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

437

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

440

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

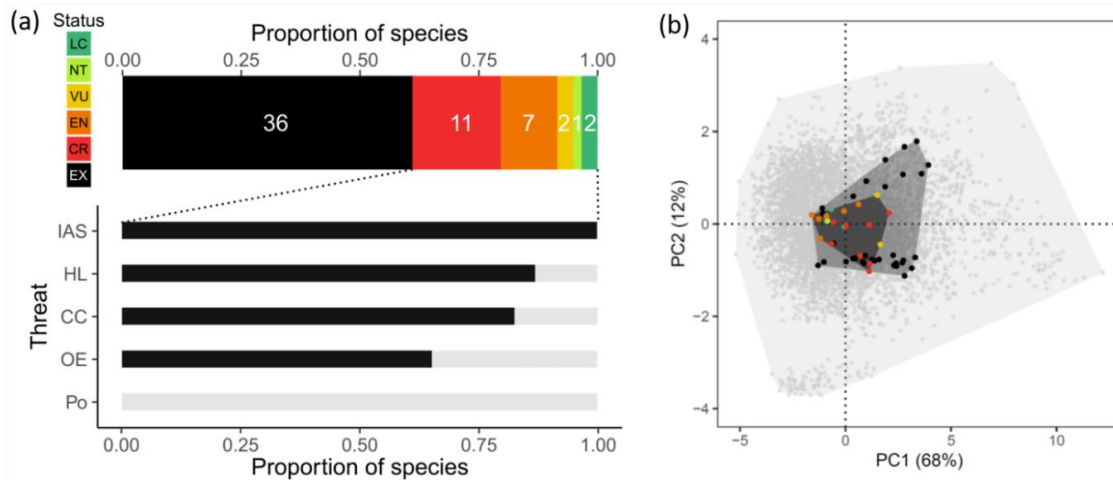
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Box 1 | The case of the Hawaiian honeycreepers

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

470 because it may lead to an upslope shift of infectious diseases (notably avian malaria)
 471 (Benning et al., 2002).



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

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Box 2 | A novel vulnerability framework fit for multiple policies

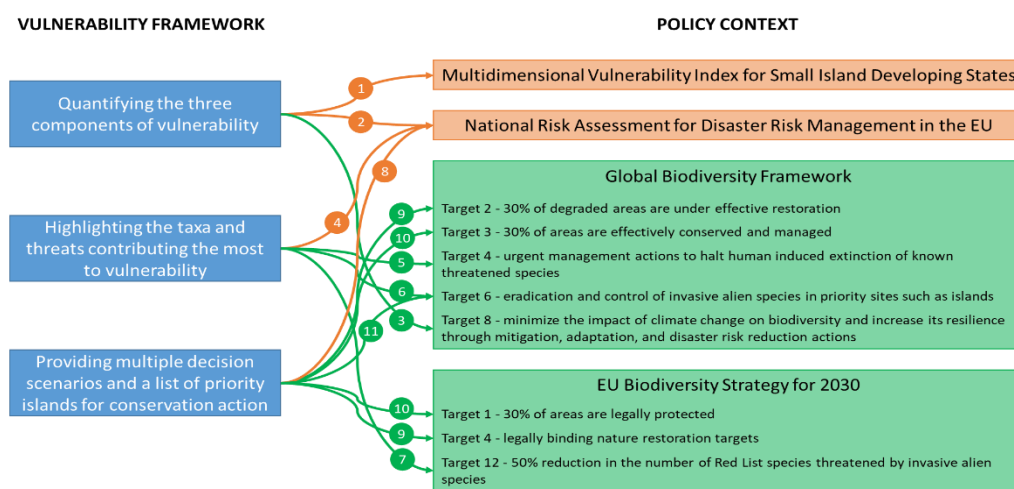
This novel vulnerability framework proposed for insular biodiversity is particularly relevant to inform various vulnerability policies (in orange) and biodiversity policies (in green) at both international and supranational levels such as the European Union (EU).

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 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), 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-

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

518 Second, our framework can help EU Member States with insular territories to identify and
519 prioritise which species, sites and threats they should use to best mitigate their vulnerability in
520 the context of their National Risk Assessment (4). Such actions can also contribute to (i)
521 identifying the species, sites and to focus the management measures to reach Target 4 of the
522 GBF (5), (ii) determining the most vulnerable islands to invasive alien species to reach Target
523 6 of the GBF (6), and helping EU Member States with insular territories to identify species
524 threatened by invasive alien species on which to focus conservation actions to reach Target 12
525 of the [EU Biodiversity Strategy for 2030](#) (EU BDS, (7)).

526 Finally, by providing multiple scenarios of threats trajectories and a list of priority islands for
527 conservation action, our framework can contribute to document the risk of biodiversity loss
528 under different conservation scenarios to EU Member States with insular territories (8). The
529 vulnerability framework could contribute to identifying priority islands for restoration (target
530 2 of the GBF and target 4 of the EU BDS (9)) and for protection (target 3 of the GBF and
531 target 1 of the EU BDS (10)), as well as in determining which invasive alien species should
532 be eradicated or controlled (target 6 of the GBF (11)).



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

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

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Supplementary material :

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958 **Supplementary appendix 1 :**

959 **Islands as past, current, and future epicentres of extinctions:** The IUCN Red List
960 documented nearly 1,000 species as globally extinct or extinct in the wild (IUCN Red list
961 access 04/07/2023). Among the documented extinctions, the overwhelming majority occurred
962 on oceanic or continental islands. Extinction hotspots are located in the Oceanic realm,
963 followed by the Afrotropical realm (e.g., Madagascar) and the Nearctic. Among reported
964 extinctions in the IUCN Red List, the large majority are represented by animal species
965 (>80%), followed by plants. Most of these species have gone extinct because of invasive alien
966 species, but a large number have also concurrently been harmed by wildlife exploitation
967 and/or land use change due to cultivation (Leclerc et al., 2018; Maxwell et al., 2016). In fact,
968 extinct species within insular regions have faced, on average, four threats within the IUCN
969 threat classification scheme (i.e., biological invasions, wildlife exploitation, cultivation, and
970 habitat modifications), and the exposure to threats for birds, mammals, amphibians, reptiles,
971 freshwater fish, plants, arthropods, and gastropods continue to increase, with an average of ten
972 threats being faced by threatened species (Leclerc et al., 2018).

973 Moreover, islands makes a disproportionate contribution to global biodiversity in relation to
974 their surface area, especially regarding the percentage of threatened species (Fernández-
975 Palacios, Kreft, et al., 2021), the risk for future extinctions is thus high. The first extinction
976 due to climate change already occurred on an island near Papua New Guinea (Fulton, 2017).
977 In 2016, *Melomys rubicola* (also called the Bramble Cay melomys) became the very first
978 documented extinction due to climate change (Waller et al., 2017). This small rodent was
979 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)
	Reproductive strategies	species	GIFT (plants), COMPADRE, GARD (reptiles)
	Mating system	species	
Reproduction	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),

Acclimatation or evolution			COMBINE (mammals), AVONET (birds).
	Clonality	species	GIFT (plants), COMPADRE (plants)
	Phenotypic plasticity	population	Noble et al 2018 (reptiles)
	Genetic diversity	population	Miraldo et al. 2016 (mammals)
Ecological characteristics	Life span	species	AmphiBio (amphibians), GARD (reptiles), GBIF (plants), BIEN (plants)
	Climatic niche breadth	species	Global Species Database (vertebrates, invertebrates, plants)
	Endemicity	species	IUCN
	Habitat specialization	species	IUCN
	Habitat condition	species	IUCN
	Diet breadth	species	GARD (reptiles), AmphiBio (amphibians), GARD (reptiles), Amphibio (amphibians)
	Nocturnality	species	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	
	Movement and mobility	Migration frequency and distance	species
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
	Level of disharmony	community	
Community properties	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)
	Behaviour (aggressiveness)	species	
Inherent vulnerability	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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1051 **Table S2:** An overview of studies assessing the vulnerability of species to climate change. This
 1052 is a non-exhaustive summary that focuses on studies that have been conducted recently, use
 1053 trait-based approaches, and include a full assessment of vulnerability with exposure, sensitivity,
 1054 and adaptive capacity markers.
 1055

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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