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The economic cost of protecting islands from invasive alien species

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1 **The economic cost of protecting islands from invasive alien species**

2

3 **Abstract**

4

5 Biological invasions represent a key threat to insular systems, with pronounced
6 impacts across environments and economies. The ecological impacts have received
7 substantial focus, but the socio-economic impacts are poorly synthesised across
8 spatial and temporal scales. We utilised the InvaCost database, the most
9 comprehensive assessment of published economic costs of invasive species, to
10 assess economic impacts on islands worldwide. We analysed socio-economic costs
11 across differing expenditure types, and examined temporal trends, across islands
12 that differ in their political geography – island nation states, overseas territories and
13 islands of continental countries. Over US\$36 billion in total costs (including both
14 damages and management) has occurred on islands between 1965-2020 due to
15 invasive species' impacts. Nation states incurred the greatest total and management
16 costs, with islands of continental countries incurring costs of similar magnitude, both
17 far higher than those in overseas territories. Damage-loss costs were significantly
18 lower, but with qualitatively similar patterns across differing political geographies.
19 The predominance of management spending differs from the pattern found for most
20 countries examined and suggests important knowledge gaps in the extent of many
21 damage-related socio-economic impacts. Nation states spent the greatest proportion
22 of their GDPs countering these costs, at least one order of magnitude higher than
23 other locations. Most costs were borne by authorities and stakeholders,
24 demonstrating the key role of governmental and NGO bodies in addressing island
25 invasions. Temporal trends revealed cost increases across all island types,
26 potentially reflecting efforts to tackle invasive species at larger, more socially

27 complex scales. Nevertheless, the already high total economic costs of island
28 invasions substantiate the role of biosecurity in reducing/preventing IAS arrivals to
29 reduce strains on limited financial resources and avoid threats to sustainable
30 development goals.

31
32 **Keywords:** biodiversity; economic impact; government; InvaCost; overseas territory;
33 political geography; socio-economic

34

35

36 **Introduction**

37

38 Alien species, species that have been distributed beyond their natural range through
39 human agency, are one of the driving forces of the restructuring of global and
40 regional species communities (Capinha et al. 2015, Russell et al. 2017, IPBES
41 2019). A subset of these alien species, so termed invasive alien species (IAS), can
42 have severe, multi-faceted impacts in their novel environments (Reaser et al. 2007,
43 Blackburn et al. 2011, Diagne et al. 2021a). IAS regularly cause ecological
44 devastation within island ecosystems, and are a leading cause of native species
45 extinctions (Blackburn et al. 2004, Bellard et al. 2016, Doherty et al. 2016), a
46 particular problem in these hotspots of global biodiversity with high endemism and
47 evolutionary distinctiveness (Whittaker 2007, Kier et al. 2009, Fernandez-Palacios et
48 al. 2021). IAS also severely degrade ecosystem integrity (e.g. loss of ecosystem
49 services such as inshore productivity [Graham et al. 2018]); cause declines in local
50 economies (e.g. through crop damage or reduced availability of wild food stocks
51 [Naylor 1996, Ballew et al. 2016]); and can have significant impacts on human health
52 by increasing welfare costs and hospitalisations (Mwebaze et al. 2010).

53

54 These severe impacts are particularly problematic as, simultaneously, islands are
55 global hotspots of invasion (Bellard et al. 2016, 2017, Dawson et al. 2017, Turbelin
56 et al. 2017), and have experienced dramatic increases in alien species richness
57 (Seebens et al 2017, Moser et al. 2018). Increased recognition of the importance of
58 islands to global biodiversity has resulted in substantial IAS management efforts and
59 expenditure on islands worldwide (Veitch et al. 2011, Russell et al. 2017, Bellard et
60 al. 2017, Veitch et al. 2019); with the smaller spatial scale of many islands increasing

61 the feasibility and implementation of many management options (Jones et al. 2011,
62 2016). Indeed, the removal of IAS from islands has proven a successful way to
63 support many endangered species globally (Veitch et al. 2011, Russell et al. 2017,
64 Veitch et al. 2019). In addition, islands have proven to be the ideal testing ground for
65 developing management strategies and techniques across invasion stages, from
66 early detection (e.g. using environmental DNA [Takahara et al. 2013]), through
67 population management of target IAS (e.g. using stable isotope techniques [Bodey et
68 al. 2011]), to complete eradication (Veitch et al. 2011, Carter et al. 2021).

69
70 While islands have proven fertile locations for such work, many of these
71 management and mitigation strategies for IAS remain costly to implement, requiring
72 significant resources, strategic planning and workforce capability to enact effectively.
73 Such efforts come with unavoidable economic costs (Diagne et al. 2021a), which
74 means that the political and administrative situation of an island can substantially
75 influence the implementation of IAS management efforts. For example, the vast
76 majority of independent small island states are considered to be developing
77 economies. However, in contrast, island overseas territories (OTs) that may be
78 geographically nearby and topographically similar, are exclusively administered by
79 economically more developed nations with greater financial resources to potentially
80 deploy (Dawson et al. 2014, Churchyard et al. 2014, Vaas et al. 2017, Sieber et al.
81 2018). Examples within the Caribbean include a range of islands comprising OTs
82 (e.g. US Virgin Islands, Montserrat, Martinique), independent nation states (e.g.
83 Jamaica, Haiti, Dominica) and offshore islands of otherwise continental countries
84 (e.g. the Swan Islands, Honduras, and Islas Caracas, Venezuela). This discrepancy
85 in the availability of internal and external financial resources may create differences
86 in the ability or incentives to control IAS. Political and administrative differences may

87 thus significantly affect investment in IAS management when other contrasting, and
88 often urgent, societal needs exist.

89
90 An economic rationale is often considered necessary for justifying action (or inaction)
91 concerning IAS. Cost-benefit analyses are often conducted as prioritisation exercises
92 (Dawson et al. 2014, Holmes et al. 2019, Carter et al. 2021), but typically these tend
93 to be on a case-by-case basis, either operationally or within a specific location.

94 However, more generalised modelling suggests that prevention of IAS impacts is
95 less cost-intensive than post-establishment responses (Leung et al. 2002, Ahmed et
96 al. 2022, Cuthbert et al. 2022). Beyond island-, or even country-specific contexts, we
97 are currently lacking a global synthesis of the economic costs of IAS on islands
98 (Reaser et al. 2007). This is concerning, as such a synthesis would help to identify
99 knowledge and/or management gaps locally and internationally, and would highlight
100 actions that can produce synergies across sectors or locations. Such win-win
101 scenarios are particularly important for addressing multi-faceted threats such as
102 those posed by invasive species.

103
104 To address the need for a more thorough understanding of the costs of invasive
105 species on islands, we used the recently developed 'InvaCost' database (Diagne et
106 al. 2020) to synthesise total reported costs of IAS on islands worldwide. Total costs
107 are split between management costs (e.g. control, biosecurity, eradication) and
108 damages incurred (e.g. agricultural losses, health costs). Specifically, we compared
109 management costs and damage losses associated with IAS among islands differing
110 in political geography (independent nation states, overseas territories or islands of
111 continental countries). We hypothesised that i) the cost ratio between management
112 and damage will be equivalent across the three island categories. That is, the

113 economic impacts of IAS are felt, and responded to, similarly on all islands.
114 However, ii) there should be a positive relationship between GDP and total
115 expenditure, and iii) as a result of their ties to larger and often financially wealthier
116 states, overseas territories and islands of continental countries should have
117 proportionately greater expenditures compared to independent island nation states.
118 Finally, in light of the ongoing increase in, and recognition of, IAS impacts, we
119 hypothesised that iv) all islands would experience increases in total costs over time.
120

121 **Methods**

122

123 *InvaCost dataset*

124 We used the InvaCost database v.4.0 (published June 2021), a publicly available
125 living repository that compiles the monetary impacts of invasive species globally
126 (<https://doi.org/10.6084/m9.figshare.12668570>). Diagne et al. (2020) developed
127 InvaCost via standardised literature searches (using the Web of Science platform,
128 Google Scholar and Google search engines), coupled with opportunistic, targeted
129 searches where data gaps were identified. Analogous searches were conducted in
130 >10 non-English languages (Angulo et al. 2021). Additional targeted searches,
131 specifically for economic costs on islands were made within the grey literature and
132 global funding databases (e.g. www.thegef.org). These costs (n = 100) were added
133 and all costs were standardised to a common currency (2017 US\$, hereafter \$)
134 using the annual average market exchange rate incorporating inflation factors
135 (<https://data.worldbank.org/indicator/FP.CPI.TOTL?end=2017&start=1960>).

136 *Data selection*

137 We considered only highly reliable (entries assigned as *high* in the 'Method reliability'
138 column of the database) and observed (entries assigned as *observed* in the
139 'Implementation' column) costs. This excludes entries that were not from the peer-
140 reviewed literature or official reports, or were otherwise not reproducible (*low*
141 reliability within the database); and those that were predicted on future organism
142 invasions (labelled as *potential* within the database). However, we retained studies
143 where costs observed at a small scale within an island were extrapolated to larger
144 areas of the same location. We also omitted the handful of database entries with no

145 value in the 'Cost_estimate_per_year_2017_USD_exchange_rate' column. We
146 consider this approach to provide minimum but robust estimates of the economic
147 costs of IAS.

148 To derive island costs specifically (Fig 1), we screened the full database by the
149 metadata columns 'Official country' and 'Location'. The 'Official_country' column was
150 used to locate all costs associated with island nation states i.e. all countries
151 surrounded by oceanic waters smaller than continental Australia. The column
152 'Location' was then used to capture all costs associated with islands of continental
153 nations e.g. the Galapagos (offshore islands of continental Ecuador) or Reunion (OT
154 of France). Costs without geographic precision ('islands') or that spanned islands of
155 differing political geographies were excluded (n = 3 in total).

156 *Cost and Island descriptors*

157 All costs were categorised by cost type, impacted sector and species. For cost type,
158 we used the three categories from the 'Type_of_cost_merged' column for
159 categorisation: *Management* (e.g. control, biosecurity, eradication effort), *Damage*
160 (e.g. infrastructure repairs/losses, health impacts) and *Mixed* (costs not accurately
161 separable between the two aforementioned categories, plus a small quantity of
162 unspecified costs [~0.05% of the total]). The 'Impacted_sector' column was
163 combined into broad socio-economic categories, namely: *Authorities-Stakeholders*
164 (principally governmental and NGO actors), *Primary Industries* (agriculture, forestry
165 etc.), *Health and Social Welfare*, *Environment* and *Mixed* (costs without definable
166 detail across multiple sectors, including a small unspecified amount [~\$270 000]).
167 We used the 'Species' column to determine costs for individual species, with this
168 analysis necessarily restricted to costs with precise values attributable to specific

169 species i.e. a multi-species eradication without detailed breakdowns would be
170 excluded as costs could not be accurately assigned among multiple targeted
171 species.

172 We then added additional classifiers to identify island types and metrics suggested to
173 predict IAS impacts. 'Island Type' distinguishes islands among three distinct political
174 geographic categories: island *Nation States* (NSs, all sovereign island nations
175 present); *Islands of Continental Countries* (ICCs, any island, from small rock stacks
176 to large archipelagos, that is an insular component of an otherwise continental
177 political entity, regardless of distance from continental to insular location e.g. Corsica
178 is an ICC of France); and *Overseas Territories* (OTs, islands that are politically an
179 integral part of, or in some way dependent on, another state for ultimate governance
180 while being geographically separated from that country's territorial waters e.g. New
181 Caledonia is an OT of France). We added multiple metrics hypothesised to influence
182 the likelihood of IAS impacts or arrival rates (IUCN 2018), namely: i) the World
183 Bank's gross national income (GNI) assessment (four groups: lower, lower middle,
184 upper middle, upper [World Bank Development Indicators 2019]); ii) average GDP of
185 the relevant country from 1970-2018 (World Bank 2021) for the NS itself, or the
186 continental country or ultimate administrative country of the ICC/OT in question.
187 Such estimates are typically not available for individual OTs due to their complex
188 administrative relationships, and other measures, such as the UN World Economic
189 Situation and Prospects reports, do not provide country-level resolutions; iii)
190 Population density (people per sq. km of land area); iv) merchandise imports (current
191 USD) v) number of international tourist arrivals (all from World Bank 2021,
192 <https://data.worldbank.org/>)

193 Finally, to allow for comparisons across locations, costs were standardised to a
194 common metric of \$ km⁻² using the information provided by the 'Spatial_scale'
195 column, where necessary returning to the primary source for clarification. Scaling to
196 the areas over which costs are recorded facilitates comparisons in the same way as
197 scaling to a standard currency. For example, spending \$1 million on IAS
198 management over an entire country would potentially produce very different
199 outcomes than the same value directed to a smaller within-country location. When
200 no more precise scale was available, we adopted a conservative approach by
201 assuming that the cost applied either to the entire named island(s) or, if even this
202 level of precision was lacking, the entire country. This resulted in the exclusion of a
203 few references with indeterminable areas (e.g., 'three ponds' or 'fishing boat'). As \$
204 km⁻² can be skewed by large costs over very small areas, e.g. eradication of fire ants
205 *Solenopsis invicta* from <1 ha of Auckland airport, New Zealand at a scaled cost km⁻²
206 of >US\$40 million (Kean et al. 2021), we examined how costs varied when
207 considering i) all cost estimates ii) cost estimates only for areas > 1 ha iii) cost
208 estimates only for areas > 5 ha.

209 *Estimating Total Costs*

210 To assess the total economic impact of average annual and cumulative costs, we
211 annualised the cost values to standardise the temporal frame of occurrence. We
212 expanded entries that reported costs spanning multiple years to determine annual
213 expenditure, providing an accurate assessment of the economic costs of IAS through
214 time. Multi-year costs were expanded using the the *expandYearlyCosts* function from
215 the 'invacost' package in R v4.0.3 (R Core Team 2020, Leroy et al. 2022), based on
216 the 'Probable_starting_year_adjusted' and 'Probable_ending_year_adjusted'

217 columns. When this information was unclear within the primary source (n = 4 entries
218 totalling ~\$120 000), we conservatively considered that these costs occurred in a
219 single year, using the ending year determined from the source as the relevant time
220 period. This resulted in an expanded dataset of 4881 annual costs covering 1965-
221 2021 (Fig 1).

222 We used this expanded dataset (Table S1) to assess area-corrected costs across
223 islands of differing political geographies in relation to cost types, socio-economic
224 sectors, geographic location, species and economic indicators. For ICCs and OTs,
225 we also compared the costs in these locales with the total documented costs (using
226 the same filtering steps in *Data Selection*) for the relevant continental country.
227 Relationships between the total costs recorded for IAS and the four metrics of GDP,
228 population density, imported goods and international arrivals were examined using
229 linear models with a Gaussian error structure, with IAS costs as the independent
230 variable and each metric in turn as the dependent variable. While other potential
231 island-level classifiers are available (e.g. total number of IAS present, arrival dates or
232 extent of spread), such data are temporally and spatially incomplete relative to each
233 other and to the economic cost timeframes, as well as being occasionally
234 contradictory. For example, cost data may relate to a specific IAS but be reported
235 many years before/after any assessment of its distribution. Given these widespread
236 temporal and spatial discrepancies and absences across sources, interpretation of
237 results would be highly problematic, with the potential to conflate genuine absences
238 of relationships with absences of relevant data.

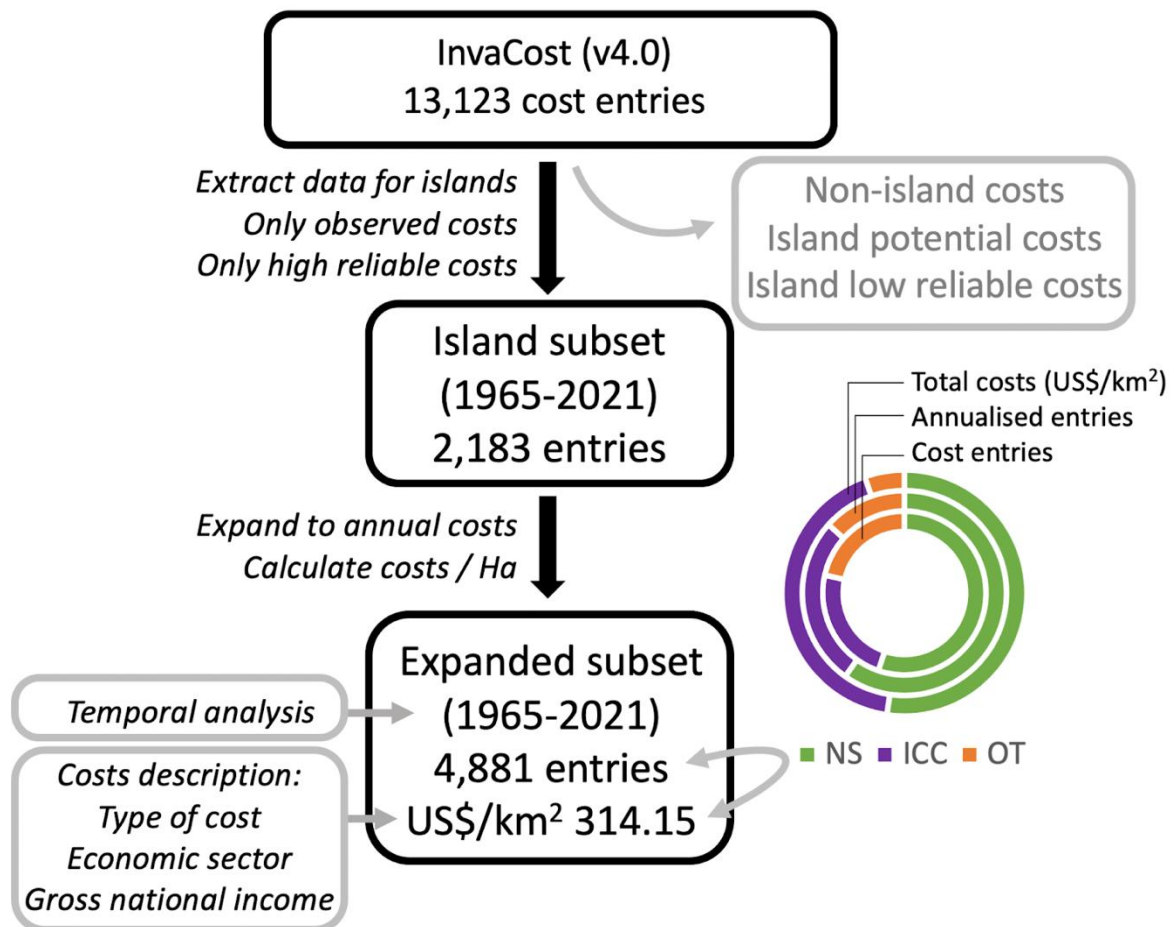
239 *Analysing temporal Trends*

240 For the analysis of temporal trends, we excluded costs that occurred after 2020 (n =
241 13), although publication lags will result in incomplete data for other recent years as
242 well (median publication lag, the difference between publication and impact years of
243 the study, was three years). Temporal trends were assessed using the
244 *summarizeCosts* function of the 'invacost' package (Leroy et al. 2022), and were
245 examined independently for total reported costs across all islands, across island
246 types separately, and for exclusive management and damage costs.

247

248

249



250

251

252 Fig 1. Workflow showing the extraction and filtering processes to determine the
253 reported economic costs of invasive alien species on islands. The graphic depicts
254 the proportion of entries, the proportion of annualised costs records and the
255 proportion of total costs in US\$ millions km⁻² spent by island type (ICC: Island of
256 Continental Country, OT Overseas Territory and NS: Nation State).

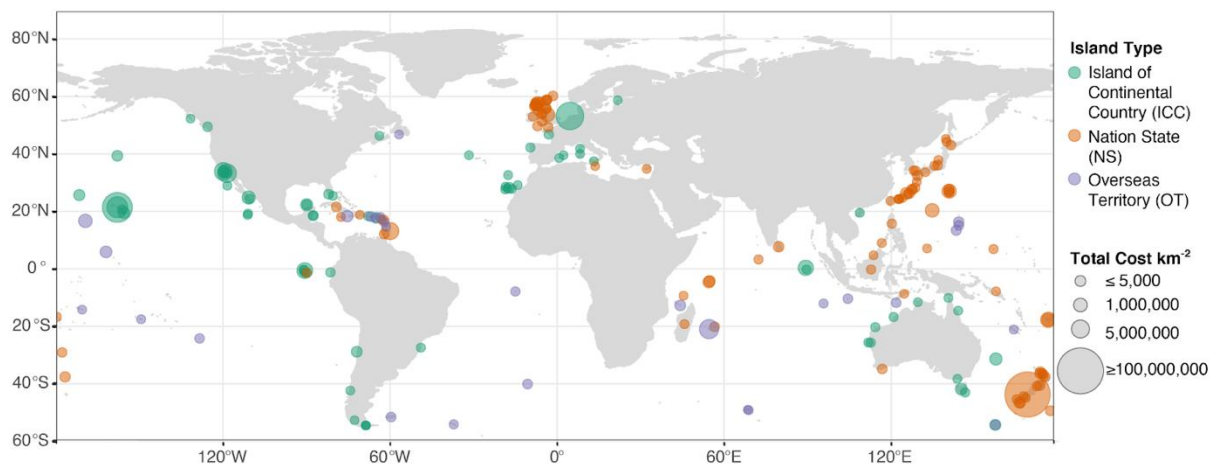
257 **Results**

258

259 The complete islands dataset consisted of 2183 unique cost entries on a minimum of
260 246 different islands, resulting in 4881 annualised costs spanning the years 1965-
261 2021. These costs summed to \$36.6 billion from a cumulative 396 million km². Total
262 area-corrected costs summed to US\$km⁻² 316 million. Over half of the cost entries
263 and annualised costs were reported from NSs (Fig 1, 2, Table S2). This translated
264 into 52% of the total reported economic costs occurring in NSs, with ICCs reporting
265 43% of the total costs, and only 5% recorded on OTs (Fig 1, 2, Table S2).

266 Geographically, spending on islands was dominated by the Pacific region (69% of
267 the total costs), including NSs and islands with political ties to countries in Oceania,
268 Europe, South and North America (Fig 2).

269



270

271
272 Fig 2. Global locations and magnitudes of total reported economic costs of invasive
273 alien species on islands. All costs are in 2017 US\$ km⁻²

274

275 *Spending ratios across island types*

276 Across all islands, the large majority of the total costs (88%) were classified as

277 management related, with exclusive damage costs comprising a mere 1% of the

278 total, and the remainder (11%) classified as mixed (Fig 3a). Across political
279 geographies, NSs spent the most on the management of IAS (>160 million \$km⁻²),
280 with almost 100 million \$km⁻² spent on management on ICCs, and ICCs also
281 accounted for the vast majority of mixed costs (Fig 3a, Table S3). The ratio of
282 management spending to damage incurred was of a similar magnitude in NSs and
283 OTs (at over 100:1), although this halved in ICCs, likely as a result of greater
284 quantities of mixed costs (Table S3). While the area over which costs were incurred
285 significantly affected the total amounts recorded, with rapid responses to early IAS
286 detections leading to effective but expensive eradications at small scales of 1-5 ha, it
287 did not qualitatively change comparisons (Fig S1).

288
289 *Influence of GDP on recorded costs*

290 As a proportion of GDP, there were large differences between island types. NSs lost
291 the greatest fraction of their income to total IAS costs, an order of magnitude higher
292 than the proportion of GDP lost in ICCs, and two orders of magnitude higher than
293 that of any OT's administrative country (Fig 3b, Table S3). In addition, reported costs
294 in OTs were an order of magnitude lower for both management and damage than for
295 other island types (Fig 3a, Table S3). A comparison of the raw 2017 US\$ costs on
296 ICCs and OTs to costs of the relevant continental/ultimate governing nations showed
297 that sums incurred on islands are consistently lower (with the exception of Ecuador).
298 Although the degree of disparity varies considerably, it remains much greater
299 between countries and OTs than ICCs (Table S4). Costs from upper and upper-
300 middle income countries dominated the dataset, with lower and lower-middle income
301 countries representing only 0.3% of all annualised costs (Table S5). Lower-middle
302 income nations are particularly poorly represented, as relatively few lower income
303 countries are island states or coastal nations. The extent of foreign visitors and

304 population density, hypothesised predictors of IAS emergence risk, were not
305 significant predictors of the reported economic costs of IAS across NSs and ICCs
306 (Fig S2). While GDP and import quantities were significantly correlated with IAS
307 costs, these relationships were driven by values for the USA, and were not present if
308 this datapoint was removed (Fig S2).

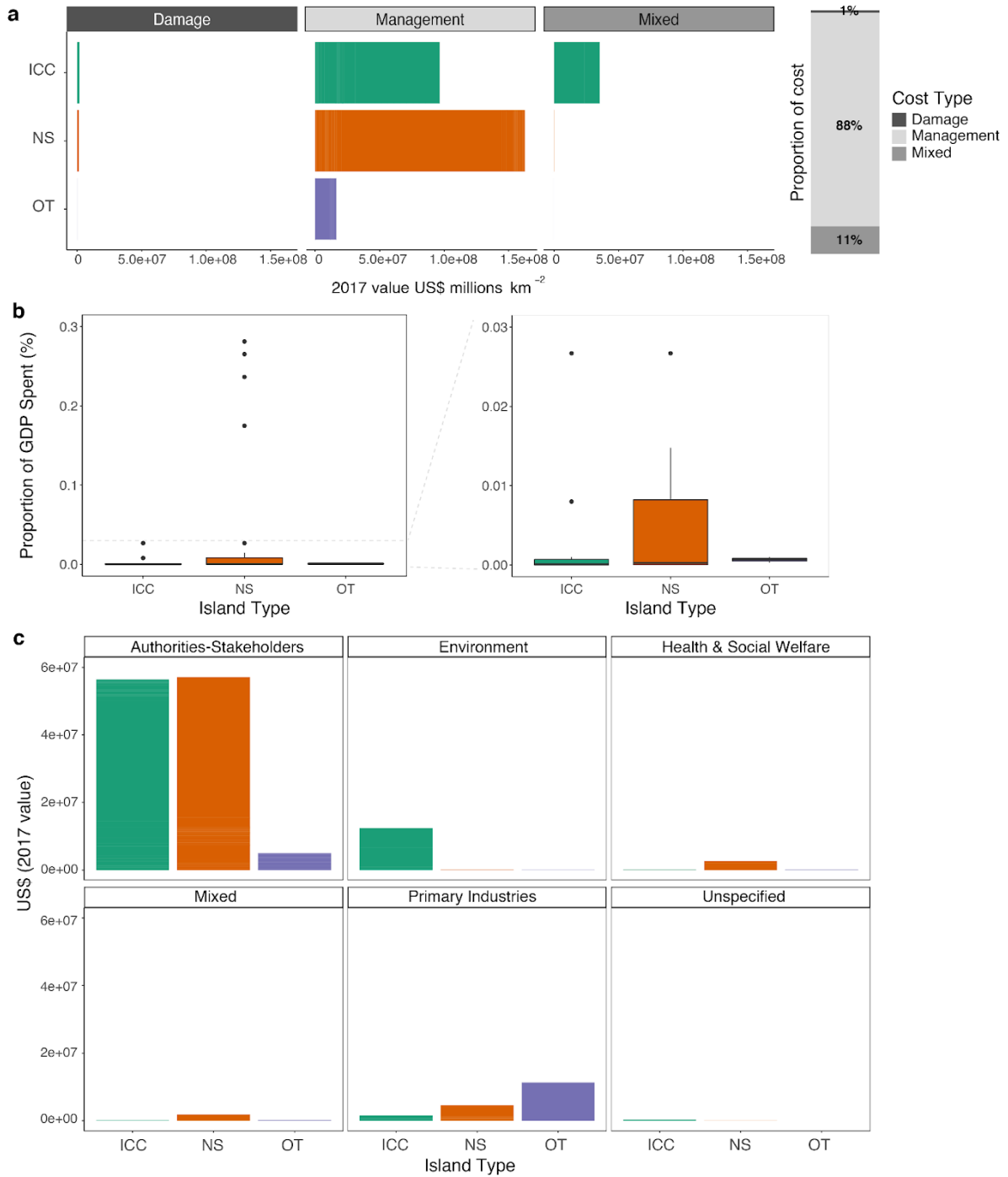
309
310 Considering all costs by socio-economic sectors revealed dominant spending
311 incurred by the Authorities-Stakeholders category (89% of annualised costs) i.e.
312 governmental departments and/or non-governmental organisations including
313 conservation agencies or charities concerned with the management of IAS. Most
314 remaining costs were attributed to Primary Industries (6%) and Environment (4%)
315 categories (Fig 3c). However, sector-related impacts were not spread evenly across
316 island types. Primary industry costs were mainly incurred by OTs, with some
317 spending in NSs; environmental costs were dominated by ICCs; and the limited
318 health and social welfare costs were documented overwhelmingly in NSs (Fig 3c).

319
320 In terms of the costliest species (considering only costs clearly assignable to single
321 species), there was almost no overlap among island types, or between the principal
322 species incurring management or damage costs (Fig 4). Taxonomic groups incurring
323 the highest costs across all island types chiefly comprised insects (137 US\$ millions
324 km⁻²) - particularly ants (98% of insect costs) - flowering plants (50 US\$ millions km⁻²)
325 ²), and mammals (16 US\$ millions km⁻²) (Fig 4). For example, some of the highest
326 total costs on ICCs and OTs are spent on tropical kudzu (*Pueraria phaseoloides*),
327 swamp stonecrop (*Crassula helmsii*) and the little fire ant (*Wasmannia*
328 *auropunctata*); whereas in NSs major costs were incurred in relation to the stoat
329 (*Mustela erminea*), the small Indian mongoose (*Urva auropunctata*), the African tulip

330 tree (*Spathodea campanulata*) and, most significantly, the red fire ant (*Solenopsis*

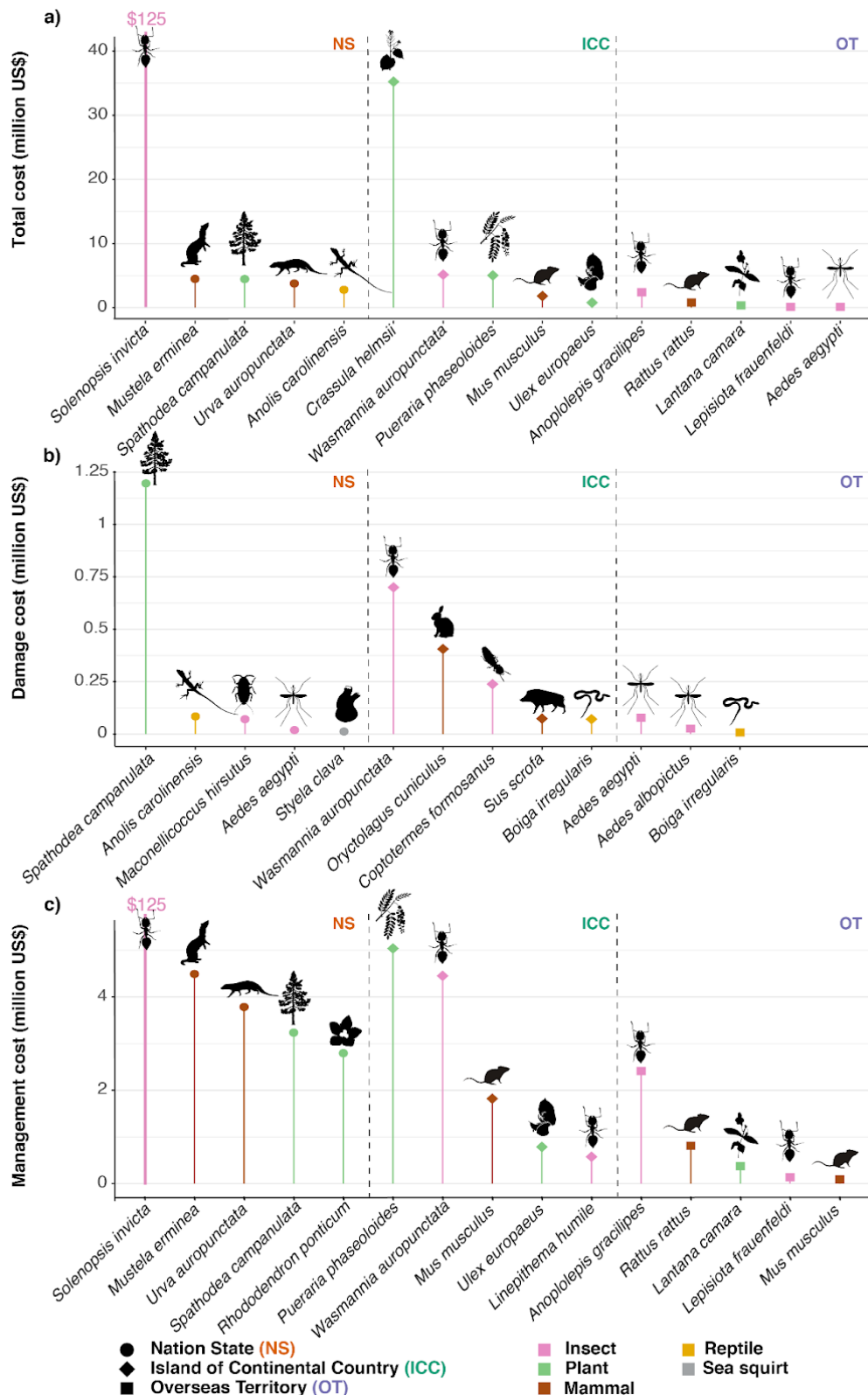
331 *invicta*).

332



334

335 Fig 3. Distribution of costs in 2017 US\$ millions km⁻² a) proportion across main cost
 336 types and in value by island categories b) Spend as a proportion of GDP by island
 337 type showing both 0 to 0.3% and 0 to 0.03% scale c) Principal socio-economic
 338 sectors (NS - Nation State, ICC - Island of Continental Country, OT - Overseas
 339 Territory)



340

341

Fig 4. Top five most costly species by island type for (a) all costs, (b) damage costs

342

and (c) management costs. All costs are in US\$ km⁻². There was almost no overlap

343

between species across different island types: NS (circles), ICC (diamonds), OT

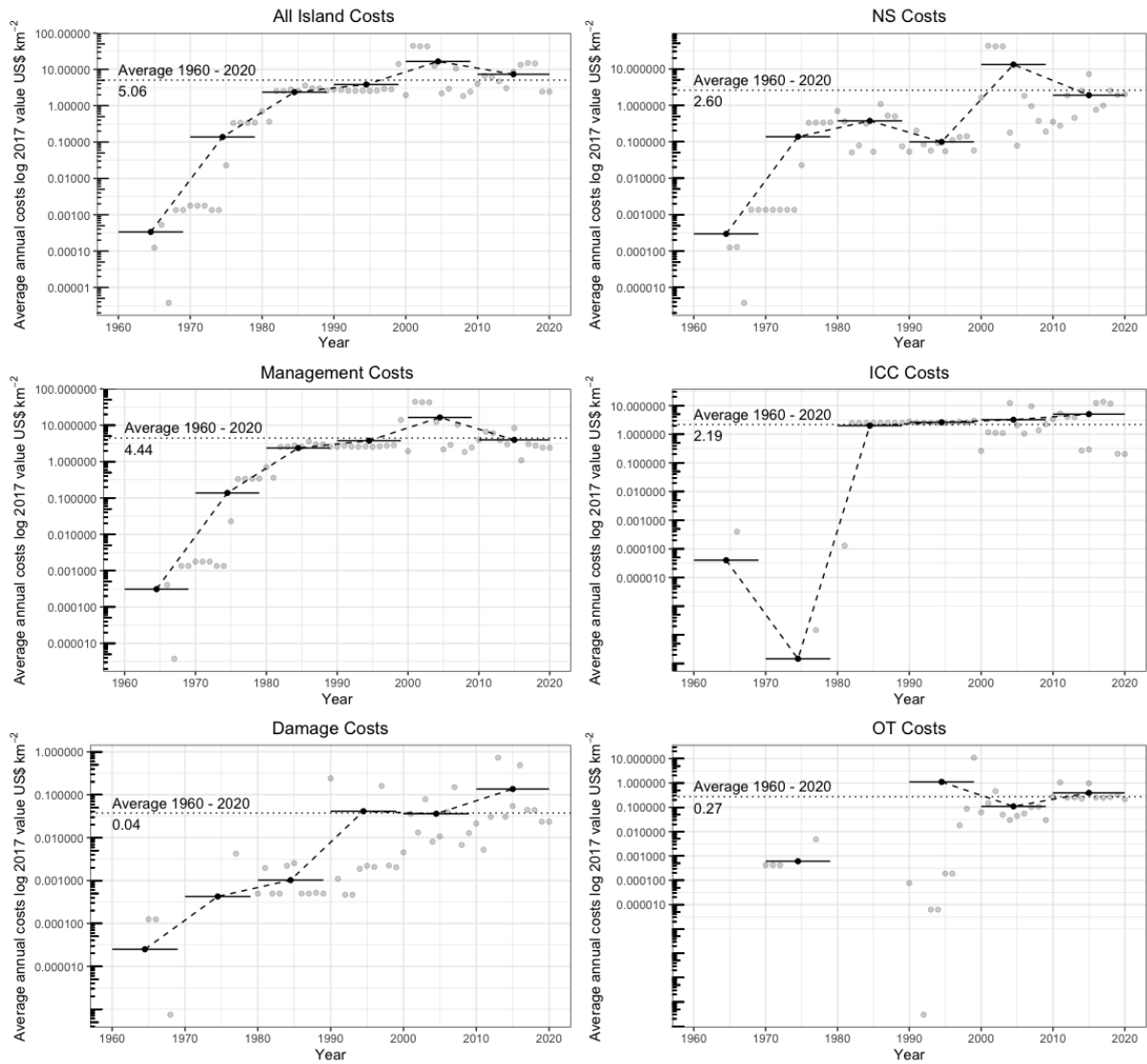
344

(squares). Note that only three species had reported damage costs within OTs.

345

346 *Temporal trends*

347 When all islands are considered together, there is an increasing trend in total
348 reported costs through each decade from 1960 to 2010, with step changes in orders
349 of magnitude in each decade except for the 1990s. The trend for the 2010s remains
350 incomplete due to publication lags, but may not see as substantial an increase (Fig
351 5). As management spending comprises the overwhelming majority of reported
352 costs, trends for management exhibit the same pattern as total costs. Damage costs
353 have similarly made exponential leaps through time while remaining at lower overall
354 values (Fig 5). Trends by island type are far less consistent. NSs show an overall
355 increasing trend but with marked drops in some decades, meaning that it is only
356 since the 2000s that they have consistently surpassed costs on ICCs. In contrast,
357 ICC costs initially rose rapidly but have only increased comparatively incrementally
358 since the 1980s. Costs on OTs are less consistently reported through time, but have
359 been similarly static, even declining in recent decades (Fig 5).



360

361 Fig 5. Temporal trends in reported total spending on the economic impacts of IAS
 362 across different types of costs and island types. Note the differing and logarithmic
 363 scales of the y-axes when comparing expenditure. Grey dots represent annual
 364 values, and solid lines represent decadal means.

365

366 **Discussion**

367

368 We found that total reported costs in insular systems to address the economic
369 impacts of IAS comprised over US\$ 300 million in area-corrected costs over the past
370 60 years. However, there was significant variation in the types of costs across island
371 types in relation to their political geography. Our hypothesis that the cost ratio
372 between management and damages would be equivalent across island types was
373 not supported by our results. Reported costs for management were always far
374 greater than those for damages, but ratios were lowest for ICCs (e.g., the
375 Galapagos, Hawai'i and Tierra del Fuego), with management spending
376 approximately twice as high as that in NSs (e.g., Japan, the Seychelles and Palau)
377 and almost three times as high as that in OTs (e.g. Christmas Island, French
378 Polynesia and Guam). In addition, reported costs in OTs substantially lagged behind
379 both ICCs and NSs under any comparisons despite the formers' connections to
380 wealthier governing countries. Similarly, although globally costs have increased
381 through time, trends across island types are very different; with broadly exponential
382 costs in NSs but only incremental changes in ICCs and OTs in recent decades. This
383 likely reflects a complex combination of factors, including the continuing emergence
384 of alien species, variation in capacity or political willingness to address these threats,
385 and changes in available technologies.

386

387 *Cost Patterns*

388

389 Management spending dominated the total costs incurred, comprising almost 90% of
390 all reported costs on islands. This proportion may well be even higher given the

391 extent of mixed management-damage costs reported within ICCs (Fig 3). This
392 substantial management-related burden contrasts with most individual countries
393 where costs due to IAS damages consistently outstrip management expenditure
394 (e.g. Haubrock et al. 2021a, Diagne et al. 2021a but see Ballesteros-Mejia et al.
395 2021). Geographically, management costs were greatest in the Pacific region, with a
396 notable lack of reported management costs from island-rich, biodiverse hotspots
397 such as the Caribbean, SE Asia and the western Indian Ocean (Kier et al. 2009).
398 This suggests that such regions would likely benefit substantially from direct
399 knowledge sharing, facilitating the coordination of approaches in such areas as
400 biosecurity, management techniques and legal instruments. While such island-rich
401 regions would benefit, the same is also true of continental countries given that
402 management costs are already proportionately greater on islands than continental
403 areas.

404
405 NSs incurred the greatest costs in total and for management, with damage costs
406 marginally highest on ICCs, and with much lower to negligible figures recorded for
407 OTs. NSs also incurred costs to a far greater (although still extremely small) extent
408 as a proportion of their GDP. This may reflect the greater impact of IAS on NS
409 economies where, for example, damage to crops directly impacts the country's
410 capacity to feed itself and increases its import requirements (Reaser et al. 2007,
411 Mwebaze et al. 2010). NSs may also have an increased reliance on their unique
412 biodiversity for generating income from natural capital or ecotourism than either ICCs
413 or OTs (Fotiou et al. 2002, Dolins et al. 2010, Ballesteros-Mejia et al. 2021).
414 However, the low value of damage costs appears to run counter to these
415 suggestions. There are three non-exclusive explanations for this trend. Firstly, lower
416 damage costs may reflect stricter biosecurity protocols that successfully reduce IAS

417 arrivals to some NSs (e.g. New Zealand [Bodey et al. 2022]) and ICCs managed as
418 strict nature reserves (e.g. the Galapagos [Ballesteros-Mejia et al. 2021]). While
419 such biosecurity expenditure has costs, these chiefly comprise salaries and
420 infrastructure costs that are unlikely to be readily captured by literature searches
421 targeted at costs specifically attributable to invasive species. Such values are not
422 regularly included in reporting of management expenditure (for example because of
423 concerns around confidentiality) in the way that equipment purchases typically are.
424 Secondly, damage costs may be less frequently recorded to precise areas - for
425 example damage may be reported simply as occurring to forestry within a country,
426 whereas management costs may be more regularly reported in detail e.g. removal of
427 a pest species from a discrete area. As we standardised costs by area, this could
428 mean that imprecisely reported costs in the primary source became comparatively
429 smaller if they were applied over larger spatial scales. Lastly, the limited damage
430 costs reported in ICCs and OTs may reflect combinations of under-reporting and low
431 investment in mitigating such losses due to their relatively small contribution to the
432 governing country's overall economy. This, coupled with small proportions of national
433 populations resident in ICCs and OTs, their spatial isolation and limited electoral
434 influence, may affect the extent of reporting for these island types in comparison to
435 NSs.

436
437 The mitigation of the costs of IAS in NSs has a clear economic rationale, but we
438 found that costs were broadly comparable between NSs and ICCs for both
439 management and damage. However, ICCs comprise many sparsely
440 inhabited/uninhabited locations (e.g., Dirk Hartog Island, Australia; the Aleutian
441 Island chain, United States; many tropical motu) where significant recordable
442 economic damages or primary industry-driven management requirements are far

443 less likely. Rather, costs in these locations probably reflect losses to biodiversity and
444 ecosystem function. While placing monetary values on such losses is complex,
445 particularly where human livelihoods are negligibly affected (Nunes et al. 2001, Kallis
446 et al. 2013, Roberts et al. 2018), ICCs provided virtually all costs incurred by the
447 environmental sector. This reflects significant management spending on IAS control
448 or eradication for ecological restoration following conservation models established in
449 New Zealand (Jones et al. 2016, Veitch et al. 2011, 2019). Under this model, islands
450 can be used to either sustain relict populations or to receive translocations of
451 endangered endemics that may receive exceptional financial support due to their
452 charismatic status and high extinction risk (Russell et al. 2015, 2017). This is
453 demonstrated by the dominance of reported costs from ICCs with successful IAS
454 eradication campaigns including: Macquarie Island, Australia (Helmstedt et al. 2016),
455 the California Channel Islands, USA (Parkes et al. 2010), the Galapagos, Ecuador
456 (Carrion et al. 2011) and islands and islets around Mexico (Samaniego-Herrera et al.
457 2018). In addition, many islands are also protected areas of varying legislative
458 status, making IAS management either a legal requirement or a high priority. The
459 significance of such programmes is seen in the extent to which governmental
460 organisations and other stakeholders, such as conservation NGOs, incurred most of
461 the reported costs across all islands.

462
463 However, it was also apparent that there was almost no consistency in the species
464 generating management or damage costs across island types. Although a number of
465 known impactful species surprisingly do not feature, for example invasive *Rattus*
466 species (Diagne et al. 2021b), this analysis was restricted to costs specifically
467 attributed to individual species, so multi-species eradication campaigns and non-
468 specific damages were excluded. Potentially this variation also reflects a lack of

469 uniformity in the reporting of costs incurred by specific species, and no doubt it also
470 reflects some regional, latitudinal or societal differences. It does, though, strongly
471 suggest there is a need for greater information sharing to synergise experience
472 across locales, particularly because many islands support a relatively homogenous
473 suite of IAS (Capinha et al. 2015). Otherwise, such gaps (whether actual absences
474 or simply a lack of data resolution) have the potential to produce erroneous
475 conclusions on how to effectively allocate budgets and prioritise efforts in what is,
476 necessarily, a triage process (Holmes et al. 2019, Carter et al. 2021). Consideration
477 of approaches from similar contexts, such as neighbouring islands, should be used
478 to further guide management approaches, including informing horizon scanning
479 exercises to identify potential threats and actions at local and regional scales
480 (Ricciardi et al. 2017).

481
482 While the number of foreign visitors or extent of imports have been considered high-
483 risk factors for the spread of IAS (Pretto et al. 2012, IUCN 2018), they did not act as
484 significant predictors of IAS costs incurred on islands. Significant relationships for
485 GDP and imports were only present when the USA (an outlier for both) was included,
486 providing little support for these as predictive metrics. There were also quantitative
487 differences in the reported area-corrected expenditure depending on how small an
488 area was included. Eradications within very small areas (e.g. the detection and
489 removal of red imported fire ants from Auckland airport [Kean et al. 2021]) produced
490 substantial area-corrected costs, but then such rapid reactive responses likely also
491 substantially reduced ongoing chronic costs, highlighting the importance of pre-
492 invasion management (Ahmed et al. 2022, Cuthbert et al. 2022). Importantly,
493 qualitative patterns, for example the dominance of management over damage costs,
494 remained unaltered if these highly localised costs are omitted.

495
496 *Underfunded or under-reported?*

497

498 Regardless of the type of cost, it is clear that costs from OTs are substantially lower
499 than on other island types. This is seen not just in absolute terms, but exceedingly so
500 in relation to the ultimate administrative countries' GDP (Table 2). As these costs are
501 corrected for area, this is not simply a reflection of the smaller landmass OTs
502 occupy. Instead, it appears that countries maintaining OTs either do not record, or do
503 not invest in, IAS mitigation at any significant scale in these locales. There is no
504 evidence to suggest that IAS presence is lower in OTs and, indeed, this would be
505 contrary to the global trend (Seebens et al. 2017). While the InvaCost database
506 collected data only on economic values, and so may fail to capture costs of
507 biodiversity or cultural losses such as the ecosystem functions of extinct island
508 endemics (Wood et al. 2017), this absence should occur across all island types.
509 However, adding additional fields to the database and expanding the search terms
510 used to attempt to capture such costs and losses, would aid with assessing these
511 absences across locations. Indeed, future developments could broaden the scope of
512 the InvaCost database to link monetary values with ecological and socio-economic
513 impacts through integration of information from other standardized global
514 assessments (e.g. EICAT and SEICAT [Hawkins et al. 2015, Bacher et al. 2017]),
515 improving our understanding of IAS impacts in the round. Nevertheless, the effort
516 expended on recording IAS costs in OTs appears to vary substantially by location.
517 For example, New Caledonia represents the highest number of cost entries, though
518 not the highest spending, within the French *départements*, with Réunion also
519 recording substantial numbers of IAS costs. However, there is simultaneously little
520 data on costs for other French OTs including Mayotte and Martinique (Renault et al.

521 2021). This pattern of limited reporting reflects a common theme of undervaluing of
522 the ecological richness of these locations, as also seen in the UK (Churchyard et al.
523 2014). Most OTs are relatively small islands with limited internal resources, and
524 without substantial external support, they will struggle to mitigate IAS impacts or
525 conserve the biodiversity they support (Churchyard et al. 2014, Vaas et al.
526 2017). While targeted funding to tackle the impacts of IAS exists (e.g. Darwin
527 Initiative www.darwininitiative.org.uk), there is clear potential for international
528 collaboration and cooperation to enhance efficacies and synergies, particularly when
529 adjacent countries are likely experiencing similar threats. This can include best
530 practice resources and training courses such as those provided under the Pacific
531 Invasives Initiative (www.pacificinvasivesinitiative.org).

532

533 Our work also highlights the almost complete lack of reported costs from lower and
534 lower-middle income economies, with <1% of all reported island costs occurring in
535 such countries. This trend is seen in many aspects of ecology and conservation
536 (Christie et al. 2021, Lynch et al. 2021, Nunez et al. 2021), and remains here despite
537 significant efforts to incorporate non-English resources (Angulo et al. 2021). It should
538 be noted that very few lower income countries are either islands or even countries
539 with a coastline, and the only lower income island NS - Madagascar - does have
540 some costs recorded. However, there is a clear absence of costs from lower-middle
541 income island nations in SE Asia and the Pacific (Haubrock et al. 2021b). This
542 absence is likely a product of true knowledge gaps resulting from limited studies due
543 to reduced capacity or competing priorities, and an urgent need to increase the
544 ability to disseminate publications and information (Wallace et al. 2020). Given that
545 several of the largely/completely absent countries represent populous and mega-

546 diverse biological hotspots (e.g., Indonesia and the Philippines), there is an urgent
547 need to improve our understanding of both the damage and management costs of
548 IAS in these locations (Kier et al. 2009, Fernandez-Palacios et al. 2021). Likely
549 impacts include severe negative effects on a range of indicators including health,
550 productivity, happiness, child poverty, ecosystem services and biodiversity (United
551 Nations 2017).

552

553 *Temporal trends*

554

555 Given the importance of islands to global diversity, temporal trends in management
556 costs exhibit a potentially encouraging trend. Average decadal management
557 spending showed an increase of over three orders of magnitude between 1960 and
558 2010, and this value is consistently at least one order of magnitude greater than
559 damage costs (Fig 3). While total management expenditure for the 2010s shows a
560 small decline, this is likely to reflect publication lag times rather than a genuine
561 reduction. However, it appears that the reasons for spending may differ across
562 islands of differing political geography, as trends differ between the three island
563 types. Costs for NSs broadly reflect the overall pattern of increasing magnitudes of
564 expenditure, reflecting a combination of ongoing increases in rates of introductions
565 (Seebens et al. 2017), increased recognition of the importance of island biodiversity
566 (Fernandez-Palacios et al. 2021) and attempts to tackle IAS at larger, more socially
567 complex scales (Oppel et al. 2011, Russell et al. 2015, Veitch et al. 2019). However,
568 costs on ICCs have largely flatlined since the 1980s. This likely reflects early
569 eradication efforts on small uninhabited islands (Jones et al. 2011, 2016,
570 Samaniego-Herrera et al. 2018) that have led to substantial conservation gains

571 (Jones et al. 2016, Holmes et al. 2019). However, translating approaches from
572 uninhabited to inhabited islands is challenging. Logistical difficulties, social
573 challenges and current higher risk of failure can constrain the capacity or willingness
574 of governments to commit to such endeavours (Oppel et al. 2011, Russell et al.
575 2015, Carter et al. 2021). A largely stable, although substantially lower, level of
576 recorded costs is also seen on OTs, again suggesting comparative underinvestment
577 or under-recording, although the complications associated with inhabited ICCs are
578 also relevant here (Dawson et al. 2014, Renault et al. 2021).

579

580 *Conclusion*

581

582 Using the most comprehensive available data on the economic impacts of IAS on
583 islands, we demonstrate that political geography is central to the type and quantity of
584 economic costs and/or the likelihood of reporting costs. Across all locations, losses
585 due to damage were far smaller than costs accrued managing IAS, likely revealing
586 knowledge gaps as to the extent of many socio-economic impacts of IAS and
587 difficulties in estimating damages across all locations (*cf* Crystal-Ornelas and
588 Lockwood 2020). There is also a clear disparity in the effective dissemination of cost
589 reporting within the literature, whether through differences in terminology,
590 classifications, or access to publishing (Wallace et al. 2020, Nunez et al. 2021).
591 Importantly, despite islands from all political geographies supporting unique
592 biodiversity, costs were especially low in OTs. Given the importance of OTs to
593 country-specific biodiversity, a greater focus of attention on IAS impacts would
594 improve progress towards the Convention on Biological Diversity's Post-2020 Global
595 Biodiversity Framework (www.cbd.int/conferences/post2020/wg2020-03/documents)

596 in these countries. Such an approach would benefit all islands given the strong
597 overlap between this framework and sustainable development goals (Schultz et al.
598 2016). Nevertheless, the predominance of spending on management approaches
599 (incorporating all stages from biosecurity through to long-term control/eradication),
600 and the continuing temporal increase in costs across all islands, suggests that there
601 continues to be the potential to make substantial conservation and development
602 gains in insular ecosystems. This includes the generation of efficiencies of scale over
603 wider management units, regional cooperation and knowledge sharing, and
604 innovations to enhance efficacy (Perrings et al. 2011, Russell et al. 2015). In
605 particular, the high economic burden imposed by biological invasions on islands
606 adds to the weight of evidence that prevention rather than management will better
607 protect insular biodiversity (Ahmed et al. 2022). International cooperation and the
608 strengthening of legal instruments would also improve approaches, for example by
609 addressing key invasion pathways (Turbelin et al. 2022), in turn reducing the
610 financial costs biological invasions continue to accrue worldwide.

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