Historic collections as a tool for assessing the global pollinator crisis

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1 **Historic collections as a tool for assessing the global pollinator**

2 **crisis**

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16 *Running head:* Historic collections and pollinators

Abstract

18 There is increasing concern about the decline of pollinators worldwide. However, despite

19 reports that pollinator declines are widespread, data are scarce and often geographically

20 and taxonomically biased. These biases limit robust inference about any potential

21 pollinator crisis. Non-structured and opportunistic historical specimen collection data

22 provide the only source of historical information which can serve as a baseline for

23 identifying pollinator declines. Specimens historically collected and preserved in museums

24 not only provide information on where and when species were collected, but also contain

25 other ecological information such as species interactions and morphological traits. Here,

26 we provide a synthesis of how researchers have used historical data to identify long-term

27 changes in biodiversity, species abundances, morphology and pollination services. Despite

28 recent advances, we show that information on the status and trends of most pollinators is

29 absent, but we highlight opportunities and limitations to progress the assessment of

30 pollinator declines globally. Finally, we demonstrate different approaches to analysing

31 museum collection data using two contrasting case studies from distinct geographical

32 regions (New Zealand and Spain) for which long-term pollinator declines have never been

33 assessed. There is immense potential for museum specimens to play a central role in

34 assessing the extent of the global pollination crisis.

35 *Keywords*: Museums, biodiversity, global change, bees, hoverflies, butterflies.

36   ## Introduction

37   Animal pollinators are a critical component of both natural and agricultural ecosystems

38   worldwide, given their role in plant reproduction [1] and food security [2]. As with many

39   other taxa, pollinators are vulnerable to a range of anthropogenic disturbances, which can

40   cause local and regional population declines or even extinctions. The vulnerability of

41   pollinators was identified several decades ago, and was popularized in 1996 by the

42   influential book "The forgotten pollinators" [3]. However, early accounts of pollinator

43   declines were somewhat anecdotal, given the lack of pollinator population data at that

44   time. These initial claims triggered the first efforts to assess this potential issue and

45   included the formation of a US National Academy of Science (NAS) panel in 2006, which

46   was commissioned to assess the extent of pollinator declines. The NAS report concluded

47   that "For most pollinator species […] the paucity of long-term population data and the

48   incomplete knowledge of even basic taxonomy and ecology make definitive assessment of

49   status exceedingly difficult" [4]. Since then, studies on pollinator responses to various

50   global change drivers have multiplied rapidly. Researchers have now developed strong

51   consensus that disturbances such as habitat destruction, land-use intensification, chemical

52   exposure, exotic species and climate change are causing pollinator declines, and often act

53   synergistically [5,6]. Yet, the current status and population trends of most pollinator

54   species worldwide remain unknown. For example, a recent IUCN report concluded that

55   even for Europe's comparatively well-studied bee fauna, greater than 55% of bee species

56   fell into the "Data Deficient" category [7]. For countries outside of Europe and the US, data

57   on pollinator populations is almost non-existent.

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58 One of the main barriers to identifying long-term pollinator population trends is that

59 pollinators are incredibly taxonomically diverse and include bees, flies, butterflies, beetles,

60 birds, bats and lizards [8]. Additionally, many pollinators are highly mobile, short-lived and

61 small, which makes monitoring their populations difficult. Bees are generally regarded as

62 the most important pollinator group due to their abundance, pollination efficiency and

63 widespread distribution [9]. However, bees are diverse, with more than 20,000 species

64 currently described worldwide, and often require expert taxonomists for identification.

65 Furthermore, the uneven distribution of researchers has resulted in geographical biases in

66 bee decline research [10], as well as taxonomic biases toward species that are easier to

67 identify, such as bumblebees [11,12].

68 One solution to overcoming these barriers is the use of space-for-time substitutions, where

69 researchers compare pollinator populations across environmental gradients. Despite

70 critiques on the robustness of this approach [13,14], these studies currently provide the

71 most extensive source of pollinator population data. For example, researchers have

72 recently estimated bee richness declines for every country in Europe using predictions

73 from models of pollinator associations with different land-use types [15]. A second

74 important method is the use of data collected from pollinator monitoring programs, which

75 are often driven by citizen scientists. This approach was inspired by successful butterfly

76 monitoring programs [16] and is currently being extended to other pollinator taxa.

77 However, these programs require significant time to generate long-term datasets and

78 cannot be used to assess historic pollinator populations. Finally, the most practical

79 approach for assessing long-term historical pollinator population trends is to use historical

80   information on species occurrences, which is often archived in museum collections [e.g.

81   17].

82   In this review, we first assess current evidence for pollinator richness declines and present

83   a roadmap outlining a strategy for using historical collection data to fill current knowledge

84   gaps. We highlight the major technical difficulties involved in using historical collection

85   data and demonstrate several approaches for analysing different types of collection data to

86   assess long-term pollinator population trends. Finally, we highlight the need to move

87   beyond simple biological diversity descriptors and unleash the power of historical data to

88   assess changes in species interactions, ecosystem functioning and evolutionary changes

89   through time.

Current evidence on pollinator declines

91   At a global scale, current evidence of pollinator declines is highly limited with most data

92   restricted to the US and Europe. It is unsurprising that studies on pollinator declines are

93   biased towards developed western countries, which have also been subject to extensive

94   anthropogenic disturbance. For example, in the UK and the Netherlands, a citizen science

95   based study using both observations and museum collection data detected strong richness

96   declines for bees, hoverflies and flowering plants [18]. In the Netherlands, museum data

97   have also revealed simultaneous plant and pollinator declines [19]. Specifically, bee species

98   with the strongest host plant preferences (i.e., specialists) displayed the strongest declines

99   and thus, were most threatened with extinction. However, it is important to note that even

100 for these two countries, local estimates of pollinator richness are biased toward large cities

101 and regions dominated by agriculture, and thus lack data for well-preserved natural areas.

102 Further exploration of this dataset revealed that for declining pollinator taxa, the trend has

103 attenuated in recent decades [20].

104 Although studies of local pollinator communities often detect richness declines, regional

105 richness may remain relatively stable. For example, regional estimates for bee species

106 richness changes in the eastern US show moderate declines [17] and very few regional

107 extinctions [21]. This is a pattern also detected in the UK, where relatively few regional bee

108 extinctions have been reported [22]. These regional findings are in stark contrast with the

109 widespread local extinctions reported in local studies. For example, Burkle et al. [23]

110 compared historical observations of bee species' occurrences in a large forested ecosystem

111 with remaining forest remnants and reports several local extinctions. However, it is

112 important to note that there is strong concordance between local extinctions and regional

113 declines [24], suggesting that local extinctions are indicators of regional population

114 declines.

115 Reported declines for bumblebees are the most severe of all pollinator taxa. For example,

116 declines of up to 18% in local bumblebee richness have been reported for Belgium and the

117 Netherlands [20]. In other parts of Europe, local richness declines range from 5% in Great

118 Britain [20] to 42% in Denmark [25]. In the USA, reported bumble decline are also severe

119 with estimates ranging between 25% [26] and 30% [17]. However, studies on species

120 richness changes for other pollinator taxa are both scarce and geographically restricted.

121 For butterflies, the only evidence of richness declines comes from Europe. Butterfly species

122 richness has declined substantially in the Netherlands and Belgium since the 1950's,

123   although declines in Great Britain have been less severe [20]. In Belgium, another study

124   [27] found that richness declines have been severe (approximately 30%), although this

125   study assessed richness changes over a longer time period (early 1900's to 2000)

126   compared to [20] (1950-69 vs. 1970-80 and 1970-89 vs. 1990-2009). In parts of Germany,

127   up to 70% declines in local butterfly richness have been reported [28]. Compared with

128   other insect pollinator taxa, there are very few studies on hoverfly species richness

129   changes, which are all restricted to Europe. In Belgium, Great Britain and the Netherlands,

130   hoverfly richness changes have been modest [20]. In the Netherlands, moderate increases

131   in hoverfly species richness have been shown, whereas in Great Britain no significant

132   directional changes were detected [18]. Furthermore, directionality (richness increase or

133   decrease) varies depending on the time period assessed. For example, hoverfly richness

134   decreased in Belgium by approximately 6% from 1950-69 to 1970-80, but increased by

135   approximately 10% between 1970-89 and 1990-2009 [20].

136   For illustrative purposes, we mapped the findings of this studies in Figure 1 to show the

137   strong contrast between bee species richness worldwide, with bee diversity hotspots in

138   Mediterranean countries, against the paucity of countries for which we have any local or

139   regional data on bee, hoverfly or butterfly declines (see raw data in Sup Mat 1). Despite

140   outside of Europe and the US and for non-insect taxa, there are very few or no studies on

141   pollinator declines using historical records, there are species-specific examples of historical

142   losses from different parts of the world (e.g., *Bombus dalbhomi*; [29]).

a) Bee species richness

b) Average bee richness change

c) Average syrphid richness change

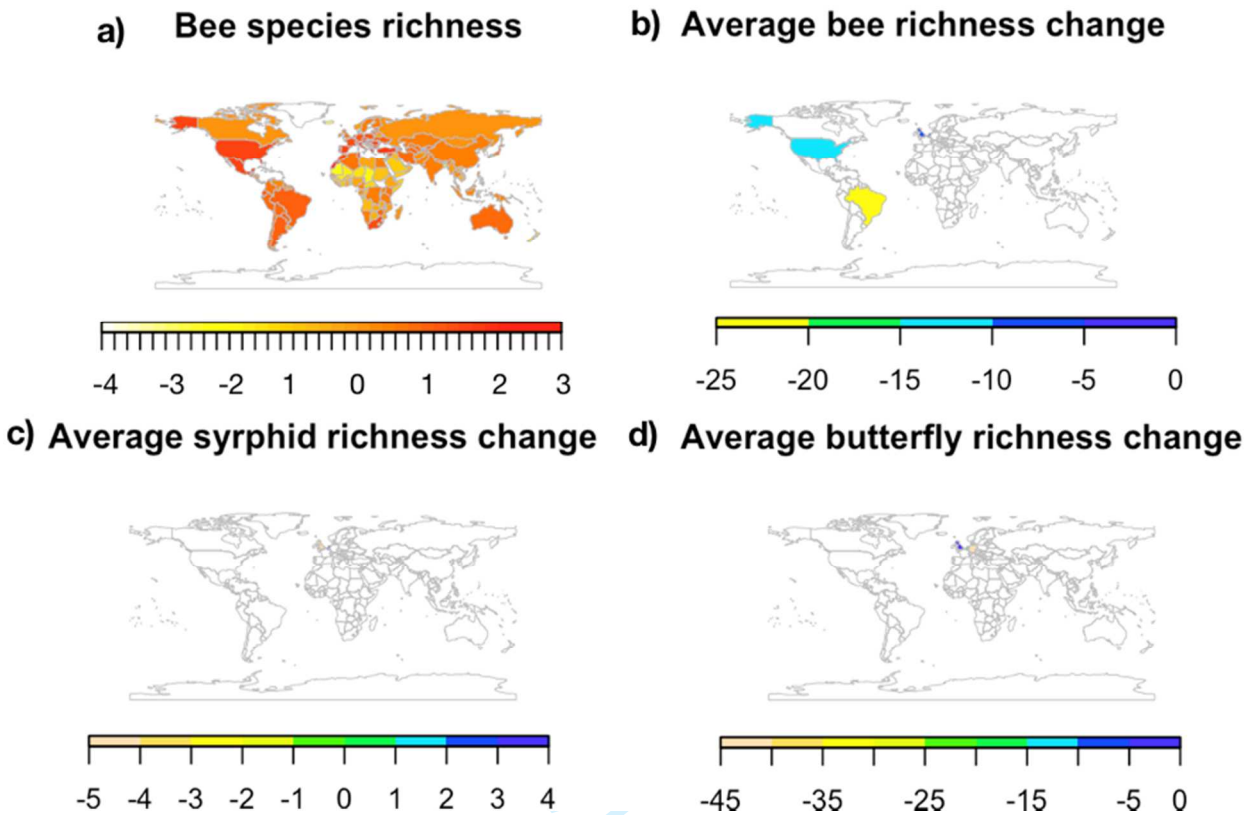d) Average butterfly richness change

143

Figure 1: Global map showing a) bee species richness per area (Data from

www.discoverlife.org) calculated as the residuals of the log-log regression between bee

species richness per country and country size. This correction accounts for the species-area

relationship. Warmer colours indicate higher bee diversity. Note that some African

countries may have incomplete listed faunas and that Alaska is included with USA values.

Countries with available historical changes in (b) bee, (c) hoverfly and (d) butterfly

richness within the last 100 years. Warmer colours indicate steeper average declines.

Countries without data are coloured in white.

Using historical collection specimen records to fill knowledge gaps

153 Estimates of pollinator declines are lacking for most countries worldwide (Figure 1). The

154 use of historic collection data may be the most effective tool for filling these gaps. The core

155 aim of museums is to conserve and curate historic collections. Thus, they serve as a

156 precious repository for specimens, and at the same time, often ensure higher quality

157 taxonomic identification. Yet, the major bottleneck for researchers wanting to use these

158 data is the lack of digitization. Digitizing old collection specimens is not a trivial task and

159 requires expertise to (i) ensure proper taxonomic identification [30–32], (ii) geo-locate the

160 coordinates of collection events (e.g. http://www.geonames.org) and (iii) store the data in

161 a properly curated database [33]. Undertaking this process for tens or hundreds of

162 thousands of museum collection specimens can be a daunting task and requires specialized

163 personnel. While some tasks can only be undertaken by people with specialist skills (e.g.,

164 taxonomists), new technologies and citizen science can speed up the collection digitization

165 process. High resolution photos of specimens and associated labels can be uploaded to the

166 internet, where the task of image transcription can be distributed across hundreds or

167 thousands of volunteers (e.g., https://www.zooniverse.org/). In addition, new algorithms

168 have been created that allow location geo-referencing based on vernacular names (e.g.

169 https://geoparser.io). However, achieving this requires adequate funding [34].

170 Where digitization has been completed, the data provide a rich source of information,

171 allowing assessment of the current status and long-term trends of pollinator populations

172 [17,19,35]. This is despite the fact that museum collections often have a number of biases,

173 including unknown sampling effort, personal interests of collectors and the curatorial

174 techniques used. For example, collectors tend to target rare or unusual over common taxa,

175 discard damaged individuals or only accession a certain number of individuals. In addition,

176 collections are often made opportunistically, leading to a spatial biases where difficult to

177 access areas are under-sampled or conversely, where samples are biased towards easily

178 accessed locations (e.g., towns/cities and/or roadsides). Further, museum collection data

179 can only be used to determine where species are present and not where they are absent.

180 However, given adequate sample sizes and appropriate statistical techniques, most biases

181 can be accounted for [e.g. 17,36,37].

182 **The way forward: Prioritizing the low hanging fruit.**

183 As we have shown, there is a paucity of countries for which historical data is available

184 (Figure 1), and hence can be used as baseline for assessing pollinator population declines.

185 While ideally one would aim to digitize all museum collection records, this is unlikely in the

186 near future, predominantly due to funding constraints. Here we show how researchers can

187 optimize the use of historical collection data to assess long-term pollinator population

188 changes.

189 GBIF (https://www.gbif.org/) is a central repository for global species occurrence data.

190 Much of these data come from museums, private collections and government research

191 institutes, but several other sources are also integrated. In combination with the popular

192 statistical language R [38], GBIF can be directly queried into your computer [39] and data

193 availability can be checked for the region of interest. Focusing on bee taxa, we show here

194 the number of modern and historic bee records currently available for different countries

195 (Figure 2a, Sup mat 2). Thirty-seven countries have more than 1800 records in each time

196 period, making these data potentially analyzable without further data collection effort (see

197 Figure 2b and c for an initial exploration). However, a proper analysis of this dataset would

198 require a careful inspection of the data, as we detail below for two specific countries (Spain

199 and New Zealand). In contrast, most countries fall short in one or both axes of Figure 2a.

200 For example, a variety of countries located in different continents such as Switzerland, Sri

201 Lanka, Nicaragua or Zimbabwe have a decent number of recent records, but lack historical

202 collections. In this cases, researchers should prioritize the digitalization of old material

203 before embarking on data analyses. For this end, it is also important to note that historical

204 records are not always vouchered in local museums (i.e., many European and USA

205 museums contain large collections of pollinators from other countries). Finally, it's

206 remarkable that more than 192 countries have less than 1000 records for each of both time

207 periods, making them poor candidates for analysing long-term pollinator population

208 trends. Aside from bees, similar exploratory analyses can easily be conducted for other

209 taxa.

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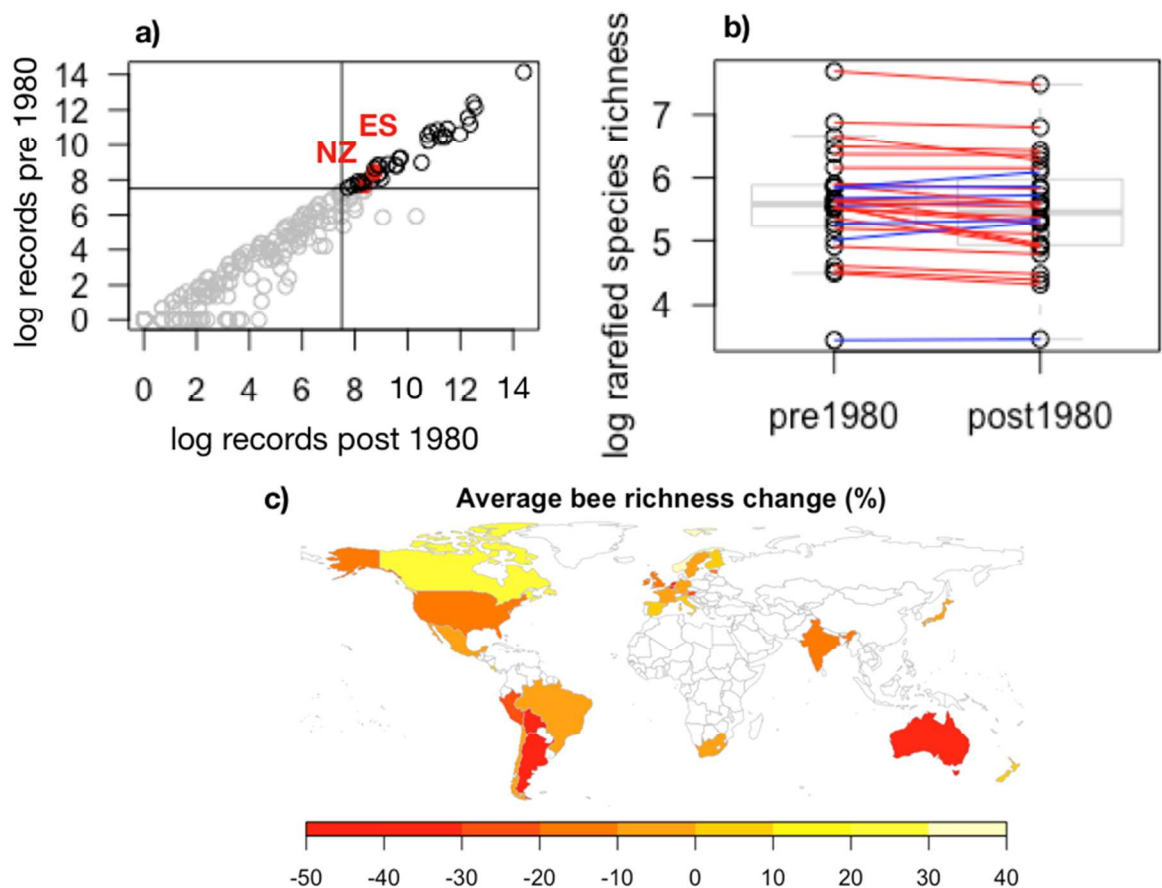
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Figure 2. Exploration of available data for bee records showing: (a) The number of bee

occurrences before 1980 and after 1980 in GBIF for each country. The upper right quadrat

(records in black) contains well covered countries with New Zealand (NZ) and Spain (ES)

marked in red (see below). For well covered countries, we show preliminary comparisons

of the rarefied number of species in both time periods and show that for most countries (21

out of 28) the number of species recorded is slightly lower (average of 10% richness

decline; red lines) for recent time periods (b). Data is log transformed for visualization

purposes. A more careful analysis of this data would help complete the map of global

declines (c). In this map we plot the % change in species recorded in GFIF for the available

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222   countries to show the potential geographic coverage. Note that this data is likely to contain

223   strong undetected biases, as we explore below.

224   As stated above, once historical collection datasets are made available, researchers must

225   identify any potential biases. We explore this process with two contrasting dataset

226   examples (Spain and New Zealand). In the Spanish dataset, most of the data comes from a

227   few specific locations and was collected by a few specific teams. Hence, the geographical

228   coverage is not representative. Even worst, historical and modern collections do not

229   overlap spatially, making any inference impossible to interpret. In this case, we contacted

230   the original collectors of the historical data to define their sampling protocols. We then

231   resurveyed the same sites (35 years after the original surveys) using the same sampling

232   protocols. In contrast, the New Zealand dataset includes a wide suite of collectors and

233   collection locations but shows no obvious biases in geographical and taxonomic coverage

234   through time. We complemented GBIF data with further museum collections for bees and

235   flies and analyze the regional richness changes through time. For these two case studies,

236   we provide annotated R scripts as examples of analysis for different dataset types (Sup Mat

237   3). These different analytical approaches allow us to reveal long-term trends in pollinator

238   populations for regions with contrasting sampling histories. We hope this resource will

239   encourage researchers to analyse data for regions where current information on pollinator

240   declines is lacking.

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Case study one: Spain

Spain provides an interesting study system because its natural habitats have been

transformed extensively by humans over a long time period, but land-use is not as

intensive compared with many other European countries. In addition, Spain is a bee

diversity hotspot (Figure 1a) and maintains a relatively heterogeneous landscape. Spain

has already digitalized a large amount of pollinator occurrence data for both historic and

recent periods (Figure 2a). However, visual inspection of the data revealed clustering

around a few localities. Further, historic records did not spatially match recent records,

making comparisons difficult. For this dataset, most of the historic records were located

around Valladolid and were collected by Enrique Asensio and collaborators. There has

been no recent sampling of bees in this area. However, we found that Enrique

systematically sampled six independent locations and that additional historical data were

available at the "Museo Nacional de Ciencias Naturales" and other minor collections.

Digitization of these records, along with a re-survey of the original sampling locations

provided an excellent dataset for a before and after comparison of bee communities.

In brief, after cleaning taxonomic names for possible typos and synonyms using the taxize

package [40], we checked for sampling completeness for both time periods and compared

rarefied species richness for each site before and after 1980 with a paired t-test

(rarefication at 1000 specimens). We found that there were a reduced number of species at

sites after 1980 (mean difference 20.27 species; 95 percent confidence interval: -1.03,

41.58; t = 2.44, df = 5, P = 0.06). However, this trend was highly dependent on site identity,

as two out of six sites showed no richness declines. Interestingly, these two localities were

263   the two that has experienced less land use changes (both are natural areas embedded into

264   agro-ecosystems). In contrast the other 4 localities suffered large urban or agricultural

265   intensification. In addition, species lost in the re-surveys are not a random selection of

266   species, but are clustered in a few genera. For example, Andrenidae and their parasites (e.g.

267   *Nomada*) showed the strongest declines whereas Halictidae tend to be more stable (Sup

268   mat 4). This pattern of winners and losers of land use intensification is in accordance with

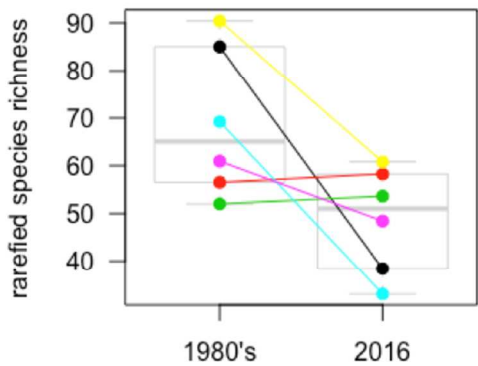269   findings elsewhere [17], indicating that some clades are more sensitive to disturbance than

270   others.

271

272   **Figure 3.** Comparison of historic collections (1980's) and modern re-surveys (2016) of the

273   rarefied richness of bees at six Spanish localities.

Case study two: New Zealand

275   In contrast to Spain, New Zealand is an isolated oceanic archipelago, with a distinctive

276   pollinator biota and a unique history of human occupation. Much of New Zealand's

277   pollinator fauna is also relatively depauperate. For example, New Zealand has only 27

278   native bee species [41], which is a fraction of nearby Australia's c. 1600 species [42].

279   However, New Zealand has a surprisingly high diversity of flies (Diptera), which are

280   important pollinators in many ecosystems [43]. Thus, New Zealand provides a unique

281   system to study long-term changes in pollinator communities, and is unlike continental

282   Europe and the US, which have been the focus of an overwhelming majority of pollinator

283   decline studies.

284   In global terms, human colonisation of New Zealand was relatively recent (c. 740 y) [44].

285   Before human arrival, New Zealand was predominately forested, but has since been

286   dramatically altered by people. Early Māori settlers cleared forests by burning and more

287   recently, European colonists cleared large tracts of remaining forests and drained low-lying

288   wetlands for agriculture, mostly before 1900 [45]. Therefore, human activity likely affected

289   pollinator communities in New Zealand long before surveys and specimen collections

290   began. Nevertheless, we can use museum records to identify trends in pollinator

291   communities during New Zealand's more recent history.

292   We used New Zealand bee collection records gathered from multiple sources, including

293   university, research institute, museum and private collections. Collection records from the

294   New Zealand Arthropod Collection (NZAC) are freely available online

295   (https://scd.landcareresearch.co.nz/). Fly pollinator data was obtained from three

296   participating New Zealand museums and covers two families (Calliphoridae and Syrphidae)

297   that contain important fly pollinators. Collections for the bee and fly datasets span over 100

298   years (early 1900s to late 2000s).

299   We followed protocols outlined in [17] to analyse the New Zealand data at the regional

300   level. First, we filtered our original datasets so that data used for analyses only included

301   independent collection events. To do this, we removed specimens collected at the same

302   location, on the same date, and by the same collector. We found our data had reasonable

303   coverage across time periods, although there was a peak in collection occurrences from

304   1960-1980. Further exploration of the New Zealand native bee data raised doubts on

305   collection completeness in records prior to 1970, so we removed these records from

306   further analyses. We accounted for differences in collection effort through binning

307   collection records by time so that each bin had a similar number of records but a different

308   number of years. We then estimated richness for each time period bin by rarefying all bins

309   to an equal number of specimens and calculated the mean species richness ±SE for each

310   bin. Finally, we estimated the significance of change in richness using a permutation test

311   that randomly reordered time periods and calculated the correlation between time period

312   and species richness. Thus, reported P-values were the proportion of permutations that

313   had higher or lower correlations compared to the correlation between richness and the

314   actual chronological time period sequence.

315   Second, to determine if the probability of finding a species in the collection changed over

316   time, we used a general linear model with a binomial distribution and a logit link. For

317   species that showed overdispersion, we used a quasi-binomial distribution. Further, we

318   only included species in this analysis for which we had 30 or more records. To account for

319   differences in sampling effort between years, we weighted each year by the total number of

320   samples collected that year.

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321 We found that rarefied richness for native bees was stable through time. Exotic bees

322 showed an increase in rarefied richness, but this trend was non-significant (P-value for

323 both natives and exotic bees > 0.05). In contrast, native fly richness declined, whereas

324 exotic fly richness increased, although results for these groups were also non-significant

325 (P-values for both groups > 0.05). Note that rarefied richness is sensitive to species

326 evenness, so increases in rarefied richness over time may actually indicate increased

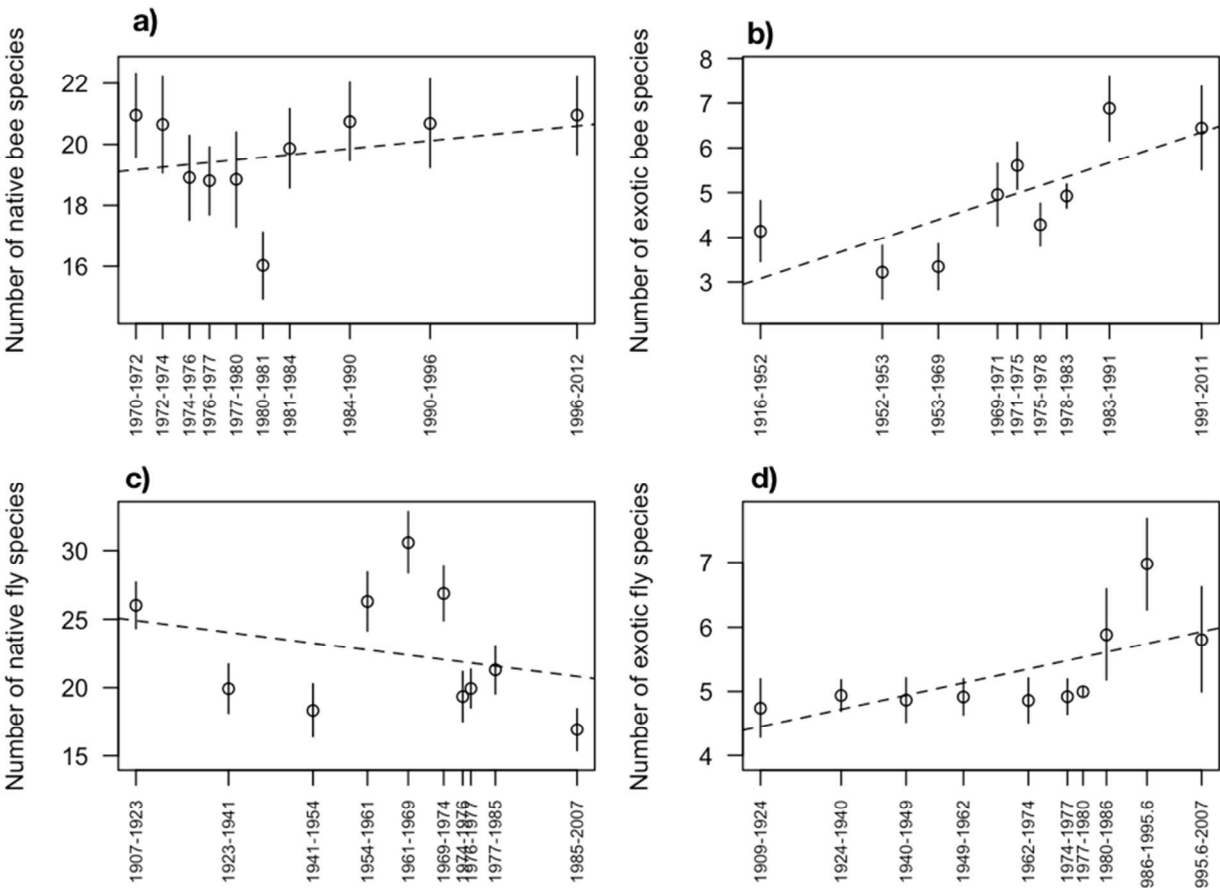327 species evenness and vice-versa for decreased richness.

329 **Figure 4**. Changes in rarefied species richness for different pollinator groups in New

330 Zealand over time. All trends were non-significant (α = 0.05).

331   However, at the species level, we found that 11 out of 27 bee species increased in relative

332   occurrence over time (10 native and one exotic) and three bee species declined in relative

333   occurrence (one native and two exotic) (Figure 3). Interestingly, the two exotic bee species

334   that declined in relative occurrence were both in the genus *Bombus*, which were

335   intentionally introduced into New Zealand for the pollination of crops. Native bees that

336   increased in relative occurrence were mostly from the genus *Leioproctus*, which are

337   medium sized, ground nesting solitary bees. Only one out of 14 fly species increased in

338   relative occurrence, which was exotic, whereas four species decreased in occurrence (three

339   native and one exotic). Native flies that decreased in relative occurrence were all Syrphidae

340   in the genus *Helophilus*.

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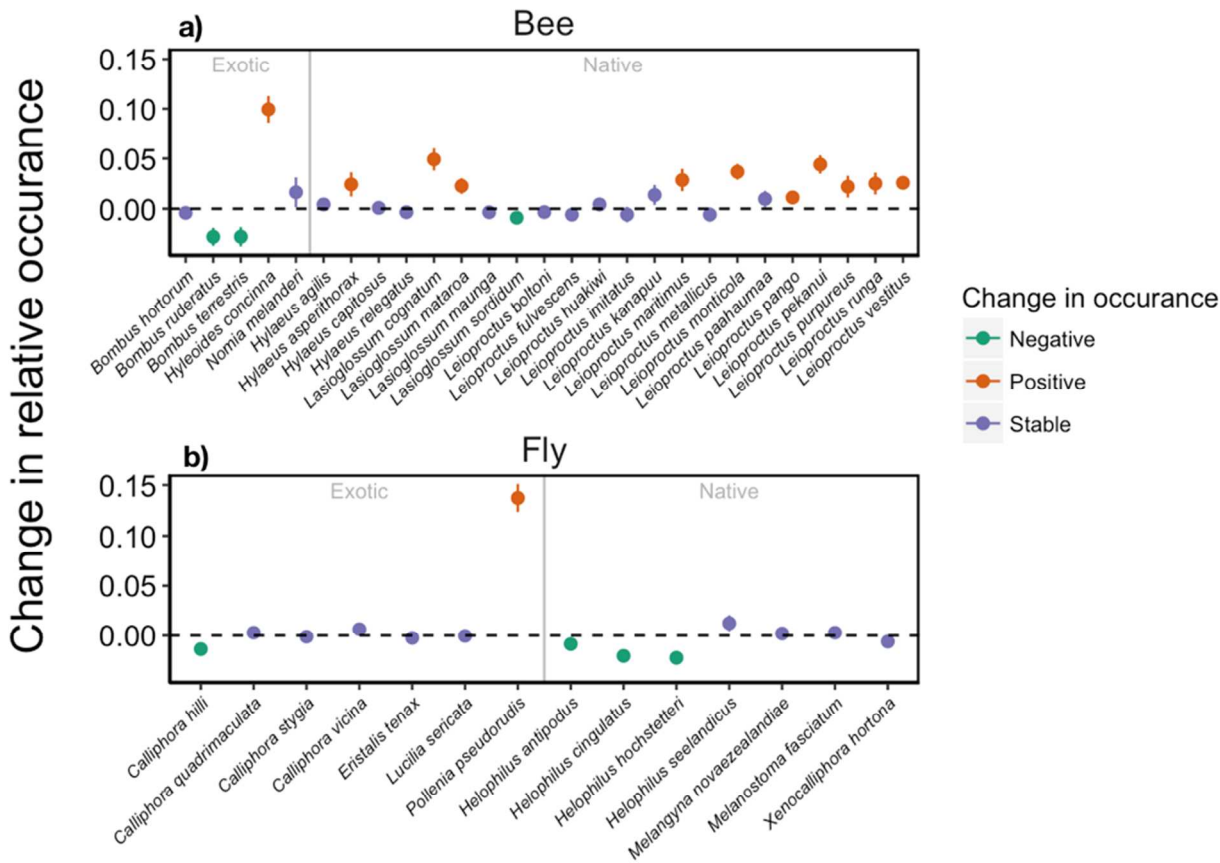
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342 **Figure 5**. Model estimated changes (± 1 SE) in the relative occurrence frequency of

343 different New Zealand bee and fly species in museum collections over time.

Beyond species occurrences

345 A recent study found that more than 90% of the papers investigating pollinator responses

346 to land-use change focused solely on richness and abundance descriptors [9]. But in

347 addition to local (*alpha*) diversity and regional (*gamma*) diversity, researchers need to

348 assess changes in turnover between sites (*beta* diversity). Environmental changes often

349 result in a few "winner" species and many "losers" species [17]. Identifying winners and

350 losers is critical as the few winners are often exotic and represent a subset of traits that

351 facilitate survival in highly modified environments [46]. These changes can have important

352 effects for pollination of native plant species and crops [47].

353 In addition, museum specimen collections can provide much more information besides

354 species occurrence records, given that such information is recorded when digitizing

355 collections. This is particularly important for identifying mechanisms of decline and

356 adaptation. For example, recording the date of collection is particularly important for

357 tracking of phenological advances congruent with contemporary climate change [48]. In

358 addition, pollinator specimen labels often include information about the host plant on

359 which the specimen was collected. This information critical for understanding past and

360 present species interactions [49]. Aside from this information, bee specimens often contain

361 pollen loads trapped on hairs, from which past visitation events can be identified [50].

362 Finally, museum specimens can be measured to track evolutionary changes by measuring

363   the traits of specimen. This approach has been already used to investigate tongue length

364   [51] and body size [52] changes in response to climate and land-use change. Finally, plant

365   herbariums can also contain indirect evidence of pollinator and pollination declines [53], a

366   basic information for linking pollinator declines with its consequences for ecosystem

367   functioning.

368   **Conclusions**

369   Unleashing the power of museum collection data to answer pressing ecological and

370   evolutionary questions is at our hands, but requires the coordinated effort of many actors.

371   Using two case studies, we show that strong collaboration between museum curators and

372   ecologists is key to understanding data and treating it appropriately. To progress our

373   understanding of the global pollination crisis, researchers and curators must aim to digitize

374   museum collection data and make it readily available in a format that is widely accessible.

375   Centralization of regional and national museum collection data in existing global platforms,

376   such as GBIF, would facilitate free and widespread access. However, datasets could also be

377   stored in alternative webpages or database repositories (e.g., university and museum

378   webpages or Dryad) providing they are thoroughly documented and easily retrieved and

379   combined with other datasets using open science tools [54].

380   We must revolutionize the way that researchers collaborate with museums, in order to

381   foster healthy bidirectional relationships. For example, ecological researchers collect

382   massive amounts of specimens, but these are often inappropriately vouchered [55,56],

383   rendering them less useful for future research. To improve this process, strong

384   communication between museums and researchers is required. However, this can only be

385   achieved with adequate funding and recognition that accurate data recording and long-

386   term preservation are critical for research [57].

387   To identify global trends in pollinator declines we require robust data, collected from

388   diverse geographic regions. It is also crucial that these data are analysed appropriately.

389   This requires researches to identify biases and to any fill taxonomic and geographic gaps

390   where possible. We need to place increased emphasis on quantifying pollinator declines in

391   regions outside of the US and Europe, and for pollinator groups other than bees. For the US

392   and Europe, there have been few regional bee extinctions [17,22] but in disturbed

393   ecosystems, declines are widespread [15,18]. For most other pollinator taxa and regions

394   throughout the world we know almost nothing. Moving forward, the first step for many

395   taxa will be to identify and describe species. Only then can we begin to document pollinator

396   declines.

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